NOAA Technical Memorandum NMFS-NE-288

U.S. Atlantic and Gulf of Mexico Marine Mammal Stock Assessments 2021

US DEPARTMENT OF COMMERCE
National Oceanic and Atmospheric Administration
National Marine Fisheries Service
Northeast Fisheries Science Center
Woods Hole, Massachusetts
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U.S. Atlantic and Gulf of Mexico Marine Mammal Stock Assessments 2021

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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 three years for stocks determined to be non-strategic. Included in this report as appendices are: a summary of serious injury/mortality estimates of marine mammals in observed U.S. fisheries (Appendix I), a summary of NMFS records of large whale human-caused serious injury and mortality (Appendix II), detailed fisheries information (Appendix III), summary tables of abundance estimates generated over recent years and the surveys from which they are derived (Appendix IV), a summary of observed fisheries bycatch (Appendix V), and estimates of human-caused mortality resulting from the *Deepwater Horizon* oil spill (Appendix VI).

Table 1 contains a summary, by species, of the information included in the stock assessments, and also indicates those that have been revised since the 2020 publication. The 2021 revisions consist primarily of updated abundance estimates and/or revised human-caused mortality and serious injury (M/SI) estimates. A total of 23 Atlantic and Gulf of Mexico stock assessment reports were written for 2021. This year, the NEFSC revised 12 stock assessment reports, 3 were “strategic” and 9 were “non-strategic.” For 2021, the SEFSC revised 11 Atlantic and Gulf of Mexico stock assessment reports representing 33 stocks (one report, the “Common Bottlenose Dolphin, Northern Gulf of Mexico Bay, Sound and Estuary Stocks” covers 23 stocks). Of the 33 SEFSC stocks, 22 are strategic and 11 are non-strategic. No SEFSC stocks changed in status from “non-strategic” to “strategic.” Three Northern Gulf of Mexico bay, sound and estuary stocks changed from “strategic” to “non-strategic” (Galveston Bay, East Bay, Trinity Bay; Mississippi River Delta; and Sabine Lake). One new SAR was written for the Galveston Bay, East Bay, Trinity Bay stock of common bottlenose dolphins. Previously, information for this stock was contained within the report “Common Bottlenose Dolphin, Northern Gulf of Mexico Bay, Sound, and Estuary 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 2021 meeting of the Atlantic Scientific Review Group (ASRG), and subsequent revisions were based on their contributions and constructive criticism. This is a working document and individual stock assessment reports will be updated as new information becomes available and as changes to marine mammal stocks and fisheries occur. The authors solicit any new information or comments which would improve future stock assessment reports.

INTRODUCTION

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

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

Following enactment of the 1994 amendments, the NMFS and USFWS held a series of workshops to develop guidelines for preparing the SARs. The first set of stock assessments for the Atlantic Coast (including the Gulf of Mexico) were published in July 1995 in the *NOAA Technical Memorandum* series (Blaylock *et al.* 1995). In April 1996, NMFS held a workshop to review proposed additions and revisions to the guidelines for preparing SARs (Wade and Angliss 1997). Guidelines developed at the workshop were followed in preparing the 1996 through 2016 SARs. In 1997 and 2004 SARs were not produced. Guidelines for preparing SARs were revised again in 2016 based largely on recommendations of the 2011 GAMMS III workshop (NMFS 2016). The revised guidelines were followed in preparing the 2017 to 2021 SARs.

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 May 2022 date-stamp at the top right corner at the beginning of each report.

REFERENCES

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

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

Total annual mortality serious injury (M/SI) and annual fisheries M/SI are mean annual figures for the period 2015–2019. Nest = estimated abundance, CV = coefficient of variation, Nmin = minimum abundance estimate, Rmax = maximum productivity rate, Fr = recovery factor, PBR = potential biological removal, unk = unknown, and undet = undetermined (PBR for species with outdated abundance estimates is considered "undetermined").

ID	Species	Stock Area	Updated this Year	Nest	Nest CV	Nmin	Rmax	Fr	PBR	Total Annual M/SI	Annual Fish. M/SI (CV)	Strategic Status	SAR of Last Update	Last Survey Year	Comments	NMFS Ctr.
1	North Atlantic right whale	Western North Atlantic	Y	368	0	364	0.04	0.1	0.7	7.7	5.7	Y	2020	2019		NEC
2	Humpback whale	Gulf of Maine	N	1,396	0	1,380	0.065	0.5	22	12.15	7.75	N	2019	2016		NEC
3	Fin whale	Western North Atlantic	Y	6,802	0.24	5,573	0.04	0.1	11	1.8	1.4	Y	2020	2016		NEC
4	Sei whale	Nova Scotia	Y	6,292	1.02	3,098	0.04	0.1	6.2	0.8	0.4	Y	2020	2016		NEC
5	Minke whale	Canadian East Coast	Y	21,968	0.31	17,002	0.04	0.5	170	10.6	9.65	N	2020	2016		NEC
6	Blue whale	Western North Atlantic	N	unk	unk	402	0.04	0.1	0.8	0	0	Y	2019	1980–2008		NEC
7	Sperm whale	North Atlantic	N	4,349	0.28	3,451	0.04	0.1	3.9	0	0	Y	2019	2016		NEC
8	Dwarf sperm whale	Western North Atlantic	N	7,750	0.38	5,689	0.04	0.4	46	0	0	N	2019	2016	Estimate for *Kogia spp.* Only.	SEC
9	Pygmy sperm whale	Western North Atlantic	N	7,750	0.38	5,689	0.04	0.4	46	0	0	N	2019	2016	Estimate for *Kogia spp.* Only.	SEC
10	Killer whale	Western North Atlantic	N	unk	unk	unk	0.04	0.5	unk	0	0	N	2014	2016		NEC
11	Pygmy killer whale	Western North Atlantic	N	unk	unk	unk	0.04	0.5	unk	0	0	N	2019	2016		SEC
12	False killer whale	Western North Atlantic	N	1,791	0.56	1,154	0.04	0.5	12	0	0	N	2019	2016		SEC

ID	Species	Stock Area	Updated this Year	Nest	Nest CV	Nmin	Rmax	Fr	PBR	Total Annual M/SI	Annual Fish. M/SI (CV)	Strategic Status	SAR of Last Update	Last Survey Year	Comments	NMFS Ctr.
13	Northern bottlenose whale	Western North Atlantic	N	unk	unk	unk	0.04	0.5	unk	0	0	N	2014	2016		NEC
14	Cuvier's beaked whale	Western North Atlantic	N	5,744	0.36	4,282	0.04	0.5	43	0.2	0	N	2019	2016		NEC
15	Blainville’s beaked whale	Western North Atlantic	N	10,107	0.27	8,085	0.04	0.5	81	0.2	0	N	2019	2016	Estimates for *Mesoplodon spp.*	NEC
16	Gervais beaked whale	Western North Atlantic	N	10,107	0.27	8,085	0.04	0.5	81	0	0	N	2019	2016	Estimates for *Mesoplodon spp.*	NEC
17	Sowerby’s beaked whale	Western North Atlantic	N	10,107	0.27	8,085	0.04	0.5	81	0	0	N	2019	2016	Estimates for *Mesoplodon spp.*	NEC
18	True’s beaked whale	Western North Atlantic	N	10,107	0.27	8,085	0.04	0.5	81	0.2	0.2	N	2019	2016	Estimates for *Mesoplodon spp.*	NEC
19	Melon-headed whale	Western North Atlantic	N	unk	unk	unk	0.04	0.5	unk	0	0	N	2019	2016		SEC
20	Risso's dolphin	Western North Atlantic	Y	35,215	0.19	30,051	0.04	0.5	301	34	34 (0.09)	N	2019	2016		NEC
21	Pilot whale, long-finned	Western North Atlantic	Y	39,215	0.30	30,627	0.04	0.5	306	9	9 (0.4)	N	2019	2016		NEC
22	Pilot whale, short-finned	Western North Atlantic	Y	28,924	0.24	23,637	0.04	0.5	236	136	136 (0.14)	N	2019	2016		SEC
23	Atlantic white-sided dolphin	Western North Atlantic	Y	93,233	0.71	54,443	0.04	0.5	544	27	27 (0.21)	N	2020	2016		NEC
24	White-beaked dolphin	Western North Atlantic	N	536,016	0.31	415,344	0.04	0.5	4,153	0	0	N	2019	2016		NEC

ID	Species	Stock Area	Updated this Year	Nest	Nest CV	Nmin	Rmax	Fr	PBR	Total Annual M/SI	Annual Fish. M/SI (CV)	Strategic Status	SAR of Last Update	Last Survey Year	Comments	NMFS Ctr.
25	Common dolphin	Western North Atlantic	Y	172,974	0.21	145,216	0.04	0.5	1,452	390	390 (0.11)	N	2020	2016		NEC
26	Atlantic spotted dolphin	Western North Atlantic	N	39,921	0.27	32,032	0.04	0.5	320	0	0	N	2019	2016		SEC
27	Pantropical spotted dolphin	Western North Atlantic	N	6,593	0.52	4,367	0.04	0.5	44	0	0	N	2019	2016		SEC
28	Striped dolphin	Western North Atlantic	N	67,036	0.29	52,939	0.04	0.5	529	0	0	N	2019	2016		NEC
29	Fraser's dolphin	Western North Atlantic	N	unk	unk	unk	0.04	0.5	unk	0	0	N	2019	2016		SEC
30	Rough-toothed dolphin	Western North Atlantic	N	136	1.0	67	0.04	0.5	0.7	0	0	N	2018	2016		SEC
31	Clymene dolphin	Western North Atlantic	N	4,237	1.03	2,071	0.04	0.5	21	0	0	N	2019	2016		SEC
32	Spinner dolphin	Western North Atlantic	N	4,102	0.99	2,045	0.04	0.5	20	0	0	N	2019	2016		SEC
33	Common bottlenose dolphin	Western North Atlantic, Offshore	N	62,851	0.23	51,914	0.04	0.5	519	28	28 (0.34)	N	2019	2016	Estimates may include sightings of the coastal form.	SEC
34	Common bottlenose dolphin	Western North Atlantic, Northern Migratory Coastal	N	6,639	0.41	4,759	0.04	0.5	48	12.2–21.5	12.2–21.5	Y	2020	2016		SEC

ID	Species	Stock Area	Updated this Year	Nest	Nest CV	Nmin	Rmax	Fr	PBR	Total Annual M/SI	Annual Fish. M/SI (CV)	Strategic Status	SAR of Last Update	Last Survey Year	Comments	NMFS Ctr.
35	Common bottlenose dolphin	Western North Atlantic, Southern Migratory Coastal	N	3,751	0.60	2,353	0.04	0.5	24	0–18.3	0–18.3	Y	2020	2016		SEC
36	Common bottlenose dolphin	Western North Atlantic, S. Carolina, Georgia Coastal	N	6,027	0.34	4,569	0.04	0.5	46	1.4–1.6	1.0–1.2	Y	2017	2017		SEC
37	Common bottlenose dolphin	Western North Atlantic, Northern Florida Coastal	N	877	0.49	595	0.04	0.5	6.0	0.6	0	Y	2017	2017		SEC
38	Common bottlenose dolphin	Western North Atlantic, Central Florida Coastal	N	1,218	0.35	913	0.04	0.5	9.1	0.4	0.4	Y	2017	2017		SEC
39	Common bottlenose dolphin	Northern North Carolina Estuarine System	N	823	0.06	782	0.04	0.5	7.8	7.2–30	7.0–29.8	Y	2020	2017		SEC
40	Common bottlenose dolphin	Southern North Carolina Estuarine System	N	unk	unk	unk	0.04	0.5	undet	0.4	0.4	Y	2020	2017		SEC
41	Common bottlenose dolphin	Northern South Carolina Estuarine System	N	unk	unk	unk	0.04	0.5	unk	0.2	0.2	Y	2015	n/a		SEC

ID	Species	Stock Area	Updated this Year	Nest	Nest CV	Nmin	Rmax	Fr	PBR	Total Annual M/SI	Annual Fish. M/SI (CV)	Strategic Status	SAR of Last Update	Last Survey Year	Comments	NMFS Ctr.
42	Common bottlenose dolphin	Charleston Estuarine System	N	unk	unk	unk	0.04	0.5	undet	unk	unk	Y	2015	2005, 2006		SEC
43	Common bottlenose dolphin	Northern Georgia, Southern South Carolina Estuarine System	N	unk	unk	unk	0.04	0.5	unk	1.4	1.4	Y	2015	n/a		SEC
44	Common bottlenose dolphin	Central Georgia Estuarine System	N	192	0.04	185	0.04	0.5	1.9	unk	unk	Y	2015	2008, 2009		SEC
45	Common bottlenose dolphin	Southern Georgia Estuarine System	N	194	0.05	185	0.04	0.5	1.9	unk	unk	Y	2015	2008, 2009		SEC
46	Common bottlenose dolphin	Jacksonville Estuarine System	N	unk	unk	unk	0.04	0.5	unk	1.2	1.2	Y	2015	n/a		SEC
47	Common bottlenose dolphin	Indian River Lagoon Estuarine System	N	unk	unk	unk	0.04	0.5	unk	4.4	4.4	Y	2015	n/a		SEC
48	Common bottlenose dolphin	Biscayne Bay	N	unk	unk	unk	0.04	0.5	unk	unk	unk	Y	2013	n/a		SEC
49	Common bottlenose dolphin	Florida Bay	N	unk	unk	unk	0.04	0.5	undet	unk	unk	N	2013	2003		SEC
50	Harbor porpoise	Gulf of Maine, Bay of Fundy	Y	95,543	0.31	74,034	0.046	0.5	851	164	163 (0.13)	N	2020	2016		NEC
51	Harbor seal	Western North Atlantic	Y	61,336	0.08	57,637	0.12	0.5	1,729	339	334 (0.09)	N	2020	2018		NEC

ID	Species	Stock Area	Updated this Year	Nest	Nest CV	Nmin	Rmax	Fr	PBR	Total Annual M/SI	Annual Fish. M/SI (CV)	Strategic Status	SAR of Last Update	Last Survey Year	Comments	NMFS Ctr.
52	Gray seal	Western North Atlantic	Y	27,300	0.22	22,785	0.128	1.0	1,458	4,453	1,169 (0.10)	N	2020	2016		NEC
53	Harp seal	Western North Atlantic	Y	7.6M	unk	7.1M	0.12	1.0	426,000	178,573	86 (0.16)	N	2019	2019		NEC
54	Hooded seal	Western North Atlantic	N	unk	unk	unk	0.12	0.75	unk	1,680	0.6 (1.12)	N	2018	n/a		NEC
55	Sperm whale	Gulf of Mexico	N	1,180	0.22	983	0.04	0.1	2.0	9.6	0.2 (1.0)	Y	2020	2017, 2018		SEC
56	Bryde's whale	Gulf of Mexico	N	51	0.5	34	0.04	0.1	0.1	0.5	0	Y	2020	2017, 2018	Total M/SI is a minimum estimate and does not include Fisheries M/SI.	SEC
57	Cuvier's beaked whale	Gulf of Mexico	N	18	0.75	10	0.04	0.5	0.1	5.2	0	N	2020	2017, 2018		SEC
58	Blainville's beaked whale	Gulf of Mexico	N	98	0.46	68	0.04	0.5	0.7	5.2	0	N	2020	2017, 2018	Estimates for *Mesoplodon spp.*	SEC
59	Gervais' beaked whale	Gulf of Mexico	N	20	0.98	10	0.04	0.5	0.1	5.2	0	N	2020	2017, 2018		SEC
60	Common bottlenose dolphin	Gulf of Mexico, Continental Shelf	Y	63,280	0.11	57,917	0.04	0.48	556	65	64.6	N	2015	2017, 2018	M/S is a minimum count and does not include projected mortality estimates for 2015–2019 due to the DWH oil spill.	SEC
61	Common bottlenose dolphin	Gulf of Mexico, Eastern Coastal	Y	16,407	0.17	14,199	0.04	0.4	114	9.2	8.8	N	2015	2017, 2018		SEC
62	Common bottlenose dolphin	Gulf of Mexico, Northern Coastal	Y	11,543	0.19	9,881	0.04	0.45	89	28	7.9	N	2015	2017, 2018		SEC
63	Common bottlenose dolphin	Gulf of Mexico, Western Coastal	Y	20,759	0.13	18,585	0.04	0.45	167	36	32.4	N	2015	2017, 2018		SEC

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64	Common bottlenose dolphin	Gulf of Mexico, Oceanic	N	7,462	0.31	5,769	0.04	0.5	58	32	0	N	2020	2017, 2018		SEC
65	Common bottlenose dolphin	Laguna Madre	Y	80	1.57	unk	0.04	0.4	undet	0.8	0.2	Y	2018	1992	Details for this stock are included in the collective report: Common bottlenose dolphin (*Tursiops truncatus truncatus*), Northern Gulf of Mexico Bay, Sound, and Estuary Stocks.	SEC
66	Common bottlenose dolphin	Neuces Bay, Corpus Christi Bay	Y	58	0.61	unk	0.04	0.4	undet	0.2	0	Y	2018	1992	Details for this stock are included in the collective report: Common bottlenose dolphin (*Tursiops truncatus truncatus*), Northern Gulf of Mexico Bay, Sound, and Estuary Stocks.	SEC
67	Common bottlenose dolphin	Copano Bay, Aransas Bay, San Antonio Bay, Redfish Bay, Espiritu Santo Bay	Y	55	0.82	unk	0.04	0.4	undet	0.6	0	Y	2018	1992	Details for this stock are included in the collective report: Common bottlenose dolphin (*Tursiops truncatus truncatus*), Northern Gulf of Mexico Bay, Sound, and Estuary Stocks.	SEC
68	Common bottlenose dolphin	Matagorda Bay, Tres Palacios Bay, Lavaca Bay	Y	61	0.45	unk	0.04	0.4	undet	0.4	0	Y	2018	1992	Details for this stock are included in the collective report: Common bottlenose dolphin (*Tursiops truncatus truncatus*), Northern Gulf of Mexico Bay, Sound, and Estuary Stocks.	SEC
69	Common bottlenose dolphin	West Bay	Y	37	0.05	35	0.04	0.4	0.3	0	0	N	2019	2014, 2015		SEC

ID	Species	Stock Area	Updated this Year	Nest	Nest CV	Nmin	Rmax	Fr	PBR	Total Annual M/SI	Annual Fish. M/SI (CV)	Strategic Status	SAR of Last Update	Last Survey Year	Comments	NMFS Ctr.
70	Common bottlenose dolphin	Galveston Bay, East Bay, Trinity Bay	Y	842	0.08	787	0.04	0.4	6.3	1.0	0.4	N	2018	2016		SEC
71	Common bottlenose dolphin	Sabine Lake	Y	122	0.19	104	0.04	0.45	0.9	0	0	N	2018	2017	Details for this stock are included in the collective report: Common bottlenose dolphin (*Tursiops truncatus truncatus*), Northern Gulf of Mexico Bay, Sound, and Estuary Stocks.	SEC
72	Common bottlenose dolphin	Calcasieu Lake	Y	0	-	-	0.04	0.45	undet	0.2	0.2	Y	2018	1992	Details for this stock are included in the collective report: Common bottlenose dolphin (*Tursiops truncatus truncatus*), Northern Gulf of Mexico Bay, Sound, and Estuary Stocks.	SEC
73	Common bottlenose dolphin	Vermilion Bay, West Cote Blanche Bay, Atchafalaya Bay	Y	0	-	-	0.04	0.45	undet	0	0	Y	2018	1992	Details for this stock are included in the collective report: Common bottlenose dolphin (*Tursiops truncatus truncatus*), Northern Gulf of Mexico Bay, Sound, and Estuary Stocks.	SEC
74	Common bottlenose dolphin	Terrebonne, Timbalier Bay Estuarine System	N	3,870	0.15	3,426	0.04	0.4	27	0.2	0	N	2018	2016		SEC
75	Common bottlenose dolphin	Barataria Bay Estuarine System	Y	2,071	0.06	1,971	0.04	0.45	18	41	0	Y	2017	2019		SEC

ID	Species	Stock Area	Updated this Year	Nest	Nest CV	Nmin	Rmax	Fr	PBR	Total Annual M/SI	Annual Fish. M/SI (CV)	Strategic Status	SAR of Last Update	Last Survey Year	Comments	NMFS Ctr.
76	Common bottlenose dolphin	Mississippi River Delta	Y	1,446	0.19	1,238	0.04	0.4	11	9.2	0.2	N	2018	2017–2018	Details for this stock are included in the collective report: Common bottlenose dolphin (*Tursiops truncatus truncatus*), Northern Gulf of Mexico Bay, Sound, and Estuary Stocks.	SEC
77	Common bottlenose dolphin	Mississippi Sound, Lake Borgne, Bay Boudreau	Y	1,265	0.35	947	0.04	0.45	8.5	59	2.0	Y	2017	2018		SEC
78	Common bottlenose dolphin	Mobile Bay, Bonsecour Bay	Y	122	0.34	unk	0.04	0.45	undet	16.0	1.0	Y	2018	1993	Details for this stock are included in the collective report: Common bottlenose dolphin (*Tursiops truncatus truncatus*), Northern Gulf of Mexico Bay, Sound, and Estuary Stocks.	SEC
79	Common bottlenose dolphin	Perdido Bay	Y	0	-	-	0.04	0.4	undet	0.8	0.6	Y	2018	1993	Details for this stock are included in the collective report: Common bottlenose dolphin (*Tursiops truncatus truncatus*), Northern Gulf of Mexico Bay, Sound, and Estuary Stocks.	SEC
80	Common bottlenose dolphin	Pensacola Bay, East Bay	Y	33	0.80	unk	0.04	0.4	undet	0.4	0.2	Y	2018	1993	Details for this stock are included in the collective report: Common bottlenose dolphin (*Tursiops truncatus truncatus*), Northern Gulf of Mexico Bay, Sound, and Estuary Stocks.	SEC
81	Common bottlenose dolphin	Chocta-whatchee Bay	N	179	0.04	unk	0.04	0.5	undet	0.4	0	Y	2015	2007		SEC

ID	Species	Stock Area	Updated this Year	Nest	Nest CV	Nmin	Rmax	Fr	PBR	Total Annual M/SI	Annual Fish. M/SI (CV)	Strategic Status	SAR of Last Update	Last Survey Year	Comments	NMFS Ctr.
82	Common bottlenose dolphin	St. Andrew Bay	N	199	0.09	185	0.04	0.4	1.5	0.2	0.2	N	2019	2016		SEC
83	Common bottlenose dolphin	St. Joseph Bay	N	142	0.17	123	0.04	0.4	1.0	unk	unk	N	2019	2011		SEC
84	Common bottlenose dolphin	St. Vincent Sound, Apalachicola Bay, St. George Sound	Y	439	0.14	unk	0.04	0.4	undet	0.2	0.2	Y	2018	2007	Details for this stock are included in the collective report: Common bottlenose dolphin (*Tursiops truncatus truncatus*), Northern Gulf of Mexico Bay, Sound, and Estuary Stocks.	SEC
85	Common bottlenose dolphin	Apalachee Bay	Y	491	0.39	unk	0.04	0.4	undet	0	0	Y	2018	1993	Details for this stock are included in the collective report: Common bottlenose dolphin (*Tursiops truncatus truncatus*), Northern Gulf of Mexico Bay, Sound, and Estuary Stocks.	SEC
86	Common bottlenose dolphin	Waccasassa Bay, Withla-coochee Bay, Crystal Bay	Y	unk	-	unk	0.04	0.4	undet	0.4	0.4	Y	2018	n/a	Details for this stock are included in the collective report: Common bottlenose dolphin (*Tursiops truncatus truncatus*), Northern Gulf of Mexico Bay, Sound, and Estuary Stocks.	SEC
87	Common bottlenose dolphin	St. Joseph Sound, Clearwater Harbor	Y	unk	-	unk	0.04	0.4	undet	0.8	0.4	Y	2018	n/a	Details for this stock are included in the collective report: Common bottlenose dolphin (*Tursiops truncatus truncatus*), Northern Gulf of Mexico Bay, Sound, and Estuary Stocks.	SEC

ID	Species	Stock Area	Updated this Year	Nest	Nest CV	Nmin	Rmax	Fr	PBR	Total Annual M/SI	Annual Fish. M/SI (CV)	Strategic Status	SAR of Last Update	Last Survey Year	Comments	NMFS Ctr.
88	Common bottlenose dolphin	Tampa Bay	Y	unk	-	unk	0.04	0.4	undet	3.0	2.2	Y	2018	n/a	Details for this stock are included in the collective report: Common bottlenose dolphin (*Tursiops truncatus truncatus*), Northern Gulf of Mexico Bay, Sound, and Estuary Stocks.	SEC
89	Common bottlenose dolphin	Sarasota Bay, Little Sarasota Bay	Y	158	0.27	126	0.04	0.4	1.0	0.2	0.2	N	2018	2015	Details for this stock are included in the collective report: Common bottlenose dolphin (*Tursiops truncatus truncatus*), Northern Gulf of Mexico Bay, Sound, and Estuary Stocks.	SEC
90	Common bottlenose dolphin	Pine Island Sound, Charlotte Harbor, Gasparilla Sound, Lemon Bay	Y	826	0.09	unk	0.04	0.4	undet	1.0	0.6	Y	2018	2006	Details for this stock are included in the collective report: Common bottlenose dolphin (*Tursiops truncatus truncatus*), Northern Gulf of Mexico Bay, Sound, and Estuary Stocks.	SEC
91	Common bottlenose dolphin	Caloosa-hatchee River	Y	0	-	-	0.04	0.4	undet	0.4	0.2	Y	2018	1985	Details for this stock are included in the collective report: Common bottlenose dolphin (*Tursiops truncatus truncatus*), Northern Gulf of Mexico Bay, Sound, and Estuary Stocks.	SEC

ID	Species	Stock Area	Updated this Year	Nest	Nest CV	Nmin	Rmax	Fr	PBR	Total Annual M/SI	Annual Fish. M/SI (CV)	Strategic Status	SAR of Last Update	Last Survey Year	Comments	NMFS Ctr.
92	Common bottlenose dolphin	Estero Bay	Y	unk	-	unk	0.04	0.4	undet	0.4	0.2	Y	2018	n/a	Details for this stock are included in the collective report: Common bottlenose dolphin (*Tursiops truncatus truncatus*), Northern Gulf of Mexico Bay, Sound, and Estuary Stocks.	SEC
93	Common bottlenose dolphin	Chokoloskee Bay, Ten Thousand Islands, Gullivan Bay	Y	unk	-	unk	0.04	0.4	undet	0.2	0.2	Y	2018	n/a	Details for this stock are included in the collective report: Common bottlenose dolphin (*Tursiops truncatus truncatus*), Northern Gulf of Mexico Bay, Sound, and Estuary Stocks.	SEC
94	Common bottlenose dolphin	Whitewater Bay	Y	unk	-	unk	0.04	0.4	undet	0	0	Y	2018	n/a	Details for this stock are included in the collective report: Common bottlenose dolphin (*Tursiops truncatus truncatus*), Northern Gulf of Mexico Bay, Sound, and Estuary Stocks.	SEC
95	Common bottlenose dolphin	Florida Keys (Bahia Honda to Key West)	Y	unk	-	unk	0.04	0.4	undet	0.2	0.2	Y	2018	n/a	Details for this stock are included in the collective report: Common bottlenose dolphin (*Tursiops truncatus truncatus*), Northern Gulf of Mexico Bay, Sound, and Estuary Stocks.	SEC
96	Atlantic spotted dolphin	Gulf of Mexico	Y	21,506	0.26	17,339	0.04	0.48	166	36	36 (0.47)	N	2015	2017, 2018	M/S is a minimum count and does not include projected mortality estimates for 2015–2019 due to the DWH oil spill.	SEC
97	Pantropical spotted dolphin	Gulf of Mexico	N	37,195	0.24	30,377	0.04	0.5	304	241	0	N	2020	2017, 2018		SEC

ID	Species	Stock Area	Updated this Year	Nest	Nest CV	Nmin	Rmax	Fr	PBR	Total Annual M/SI	Annual Fish. M/SI (CV)	Strategic Status	SAR of Last Update	Last Survey Year	Comments	NMFS Ctr.
98	Striped dolphin	Gulf of Mexico	N	1,817	0.56	1,172	0.04	0.5	12	13	0	Y	2020	2017, 2018		SEC
99	Spinner dolphin	Gulf of Mexico	N	2,991	0.54	1,954	0.04	0.5	20	113	0	Y	2020	2017, 2018		SEC
100	Rough-toothed dolphin	Gulf of Mexico	N	unk	n/a	unk	0.04	0.4	undet	39	0.8 (1.00)	N	2020	2017, 2018		SEC
101	Clymene dolphin	Gulf of Mexico	N	513	1.03	250	0.04	0.5	2.5	8.4	0	Y	2020	2017, 2018		SEC
102	Fraser's dolphin	Gulf of Mexico	N	213	1.03	104	0.04	0.5	1.0	unk	0	N	2020	2017, 2018		SEC
103	Killer whale	Gulf of Mexico	N	267	0.75	152	0.04	0.5	1.5	unk	0	N	2020	2017, 2018		SEC
104	False killer whale	Gulf of Mexico	N	494	0.79	276	0.04	0.5	2.8	2.2	0	N	2020	2017, 2018		SEC
105	Pygmy killer whale	Gulf of Mexico	N	613	1.15	283	0.04	0.5	2.8	1.6	0	N	2020	2017, 2018		SEC
106	Dwarf sperm whale	Gulf of Mexico	N	336	0.35	253	0.04	0.5	2.5	31	0	N	2020	2017, 2018	Estimate for *Kogia spp.* only.	SEC
107	Pygmy sperm whale	Gulf of Mexico	N	336	0.35	253	0.04	0.5	2.5	31	0	N	2020	2017, 2018	Estimate for *Kogia spp.* only.	SEC
108	Melon-headed whale	Gulf of Mexico	N	1,749	0.68	1,039	0.04	0.5	10	9.5	0	N	2020	2017, 2018		SEC
109	Risso's dolphin	Gulf of Mexico	N	1,974	0.46	1,368	0.04	0.5	14	5.3	0	N	2020	2017, 2018		SEC
110	Pilot whale, short-finned	Gulf of Mexico	N	1,321	0.43	934	0.04	0.4	7.5	3.9	0.4 (1.00)	N	2020	2017, 2018	Nbest includes all *Globicephala sp.*, though it is presumed that only short-finned pilot whales are present in the Gulf of Mexico.	SEC
111	Sperm Whale	Puerto Rico and U.S. Virgin Islands	N	unk	unk	unk	0.04	0.1	unk	unk	unk	Y	2010	n/a		SEC
112	Common bottlenose dolphin	Puerto Rico and U.S. Virgin Islands	N	unk	unk	unk	0.04	0.5	unk	unk	unk	Y	2011	n/a		SEC

ID	Species	Stock Area	Updated this Year	Nest	Nest CV	Nmin	Rmax	Fr	PBR	Total Annual M/SI	Annual Fish. M/SI (CV)	Strategic Status	SAR of Last Update	Last Survey Year	Comments	NMFS Ctr.
113	Cuvier’s beaked whale	Puerto Rico and U.S. Virgin Islands	N	unk	unk	unk	0.04	0.5	unk	unk	unk	Y	2011	n/a		SEC
114	Pilot whale, short-finned	Puerto Rico and U.S. Virgin Islands	N	unk	unk	unk	0.04	0.5	unk	unk	unk	Y	2011	n/a		SEC
115	Spinner dolphin	Puerto Rico and U.S. Virgin Islands	N	unk	unk	unk	0.04	0.5	unk	unk	unk	Y	2011	n/a		SEC
116	Atlantic spotted dolphin	Puerto Rico and U.S. Virgin Islands	N	unk	unk	unk	0.04	0.5	unk	unk	unk	Y	2011	n/a		SEC

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

STOCK DEFINITION AND GEOGRAPHIC RANGE

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

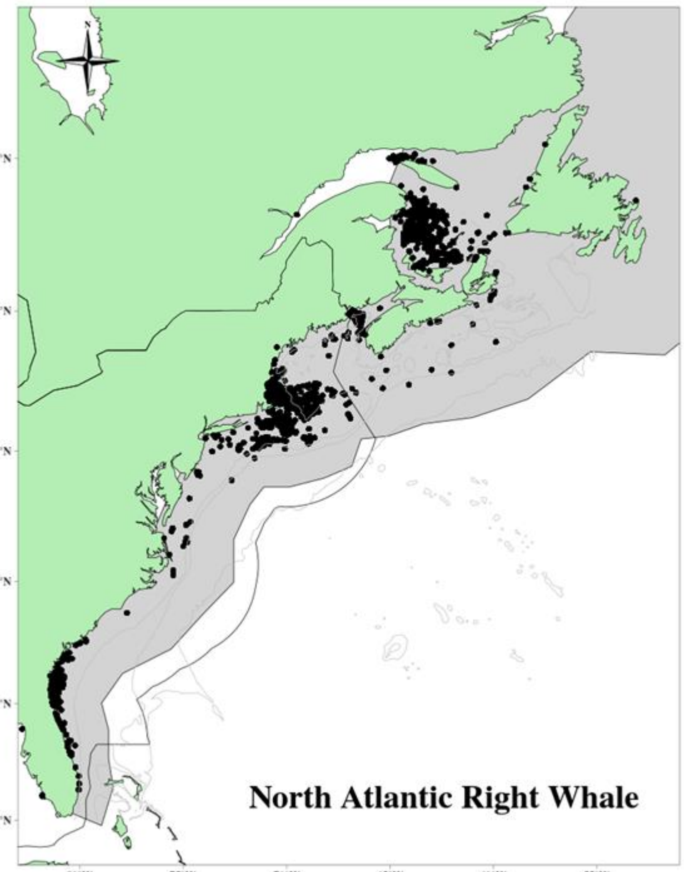

Figure 1. Approximate range (shaded area) and distribution of sightings (dots) of known North Atlantic right whales 2015–2019.

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). Davis *et al*. (2017) recently pooled together detections from a large number of passive acoustic devices and documented broad-scale use of the U.S. eastern seaboard during much of the year. In Canada, Simard *et al*. (2019) documented the frequency of right whale contact calls in the Gulf of St. Lawrence from June 2010 to November 2018 using a year-round passive acoustic network. Acoustic detections indicated right whale presence every year. The earliest detections were at the end of April and the latest in mid-January, with peak occurrence between August and the end of October. Detections were focused in the southern Gulf, and daily detection rates quadrupled at listening stations off the Gaspé Peninsula beginning in 2015.

Individuals' movements within and between habitats across the range are extensive. 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 a few weeks in the same area should not necessarily be assumed to indicate a stationary or resident animal. Instead, telemetry data have shown lengthy excursions, including into deep water off the continental shelf over short timeframes (Mate *et al*. 1997; Baumgartner and Mate 2005).

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

New England and Canadian 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). The characteristics of acceptable prey distribution in these areas are summarized in Baumgartner *et al*. (2003), and Baumgartner and Mate (2003). In 2016, the Northeastern U.S. Foraging Area Critical Habitat was expanded to include nearly all U.S. waters of the Gulf of Maine (81 FR 4837, 26 February 2016).

An important shift in habitat use patterns in 2010 was highlighted in an analysis of right whale acoustic presence in the western North Atlantic from 2004 to 2014 (Davis *et al*. 2017). This shift was also reflected in visual survey data in the greater Gulf of Maine region. Between 2012 and 2016, visual surveys detected fewer individuals in the Great South Channel (NMFS unpublished data) and the Bay of Fundy (Davies *et al*. 2019), while the number of individuals using Cape Cod Bay in spring increased (Mayo *et al*. 2018; Ganley *et al*. 2019). In addition, right whales apparently abandoned the central Gulf of Maine in winter (see Cole *et al*. 2013), but have since been seen in large numbers in a region south of Martha's Vineyard and Nantucket Islands (Leiter *et al*. 2017; Stone *et al*. 2017), an area outside of the 2016 Northeastern U.S. Foraging Area Critical Habitat. Since 2015, increased acoustic detections and survey effort in the Gulf of St. Lawrence have documented right whale presence there from late spring through the fall (Cole *et al*. 2016; DFO 2020; Simard *et al*. 2019). Photographic captures of right whales in the Gulf of St. Lawrence during the summers of 2015–2019 documented 48, 50, 133, 132 and 135 unique individuals using the region, respectively, with a total of 187 unique individuals documented over the five summers (Crowe *et al*. 2021).

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

High-resolution (i.e., using 35 microsatellite loci) genetic profiling improved our understanding of genetic variability, the number of reproductively active individuals, reproductive fitness, parentage, and relatedness of individuals (Frasier *et al*. 2007, 2009). One finding of the genetic studies is the importance of obtaining biopsy samples from calves on the calving grounds. Between 1990 and 2010, only about 60% of all known calves were seen with their mothers in summering areas when their callosity patterns are stable enough to reliably make a photo-ID match later in life. The remaining 40% were not seen on a known summering ground. Because the calf's genetic profile is the only reliable way to establish parentage, if the calf is not sampled when associated with its mother early on, then it is not possible to link it with a calving event or to its mother, and information such as age and familial relationships is lost. From 1980 to 2001, there were 64 calves born that were not sighted later with their mothers and thus unavailable to provide age-specific mortality information (Frasier *et al*. 2007).

An additional interpretation of paternity analysis was that the population size may be larger than was previously thought. Fathers for only 45% of known calves have been genetically determined; yet, genetic profiles were available for 69% of all photo-identified males (Frasier 2005). The conclusion was that the majority of these calves must have different fathers that cannot be accounted by the unsampled males, therefore the population of males must be larger (Frasier 2005). However, a recent study comparing photo-identification and pedigree genetic data for animals known or presumed to be alive during 1980–2016 found that the presumed alive estimate is similar to the actual abundance of this population, which indicates that the majority of the animals have been photo-identified (Fitzgerald 2018).

POPULATION SIZE

The western North Atlantic right whale stock size is based on a published state-space model of the sighting histories of individual whales identified using photo-identification techniques (Pace *et al*. 2017; Pace 2021). Sightings histories were constructed from the photo-ID recapture database as it existed in January 2021, and included photographic information up through November 2019. Using a hierarchical, state-space Bayesian open population model of these histories produced a median abundance value (Nest) as of 30 November 2019 of 368 individuals (95%CI: 356–378; Table 1). As with any statistically-based estimation process, uncertainties exist in the estimation of abundance because it is based on a probabilistic model that makes certain assumptions about the structure of the data. Because the statistically-based uncertainty is asymmetric about N, the credible interval is used to characterize that uncertainty (as opposed to a CV that may appear in other stock assessment reports).

Table 1. Best and minimum abundance estimates as of 30 November 2019 for the western North Atlantic right whale (Eubalaena glacialis) with Maximum Productivity Rate (R_{max}), Recovery Factor (F_r) and PBR.

N_{est}	95% Credible Interval	60% Credible Interval	N_{min}	F_r	R_{max}	PBR
368	356–378	364–373	364	0.1	0.04	0.7

Historical Abundance

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

Minimum Population Estimate

The minimum population estimate is the lower limit of the two-tailed 60% credible interval about the median of the posterior abundance estimates using the methods of Pace *et al*. (2017) and refinements of Pace (2021). 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 368. The minimum population estimate as of 30 November 2019 is 364 individuals (Table 1).

Current Population Trend

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

Examination of the abundance estimates for the years 1990–2011 (Figures 2a, 2b) suggests that abundance increased at about 2.8% per annum from posterior median point estimates of 270 individuals in 1990 to 481 in 2011, but that there was a 100% chance that abundance declined from 2011 to 2019 when the final estimate was 368 individuals. The overall abundance decline between 2011 and 2019 was 23.5% (CI=21.4% to 26.0%). There has been a considerable change in right whale habitat use patterns in areas where most of the population had been observed in previous years (e.g., Davies *et al*. 2017), exposing the population to new anthropogenic threats (Hayes *et al.* 2018). Pace (2021) found a significant decrease in mean survival rates since 2010, correlating with the observed change in area-use patterns (Figure 2c). This apparent change in habitat use also had the effect that, despite relatively constant effort to find whales in traditional areas, the chance of photographically capturing individuals decreased (Figure 3). However, the methods in Pace *et al.* (2017) and Pace (2021) account for changes in capture probability.

There were 17 right whale mortalities reported in 2017 (Daoust *et al.* 2017). This number exceeds the largest estimated mortality rate during the past 25 years. Further, despite high survey effort, only 5 and 0 calves were detected in 2017 and 2018, respectively. In 2019, 7 calves were identified (Pettis *et al.* 2021).

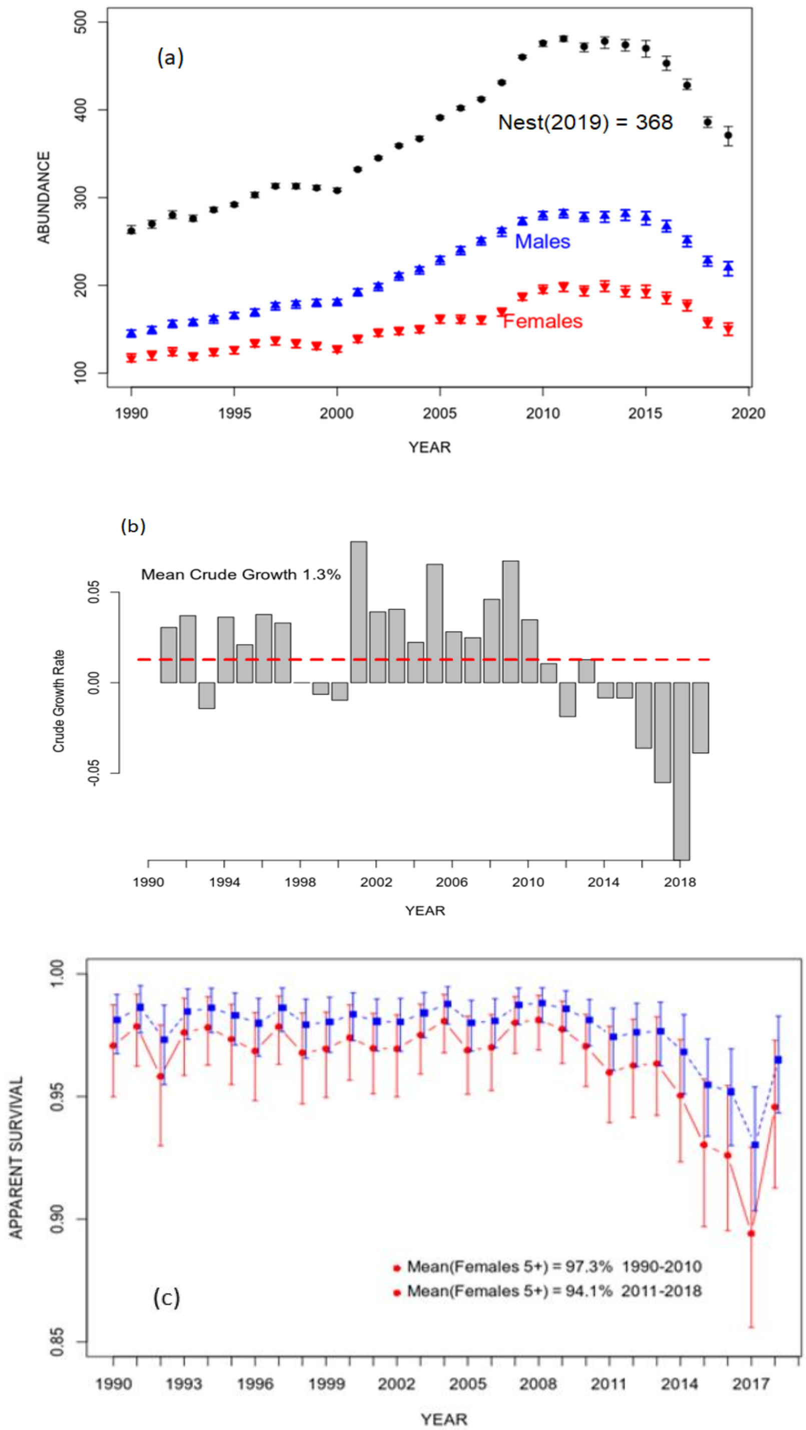

Figure 2. (a) Abundance estimates for North Atlantic right whales. Estimates are the median values of a posterior distribution from modeled capture histories. Also shown are sex-specific abundance estimates. Cataloged whales may include some but not all calves produced each year. (b) Crude annual growth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

(c) Sex-specific survival rate estimates.

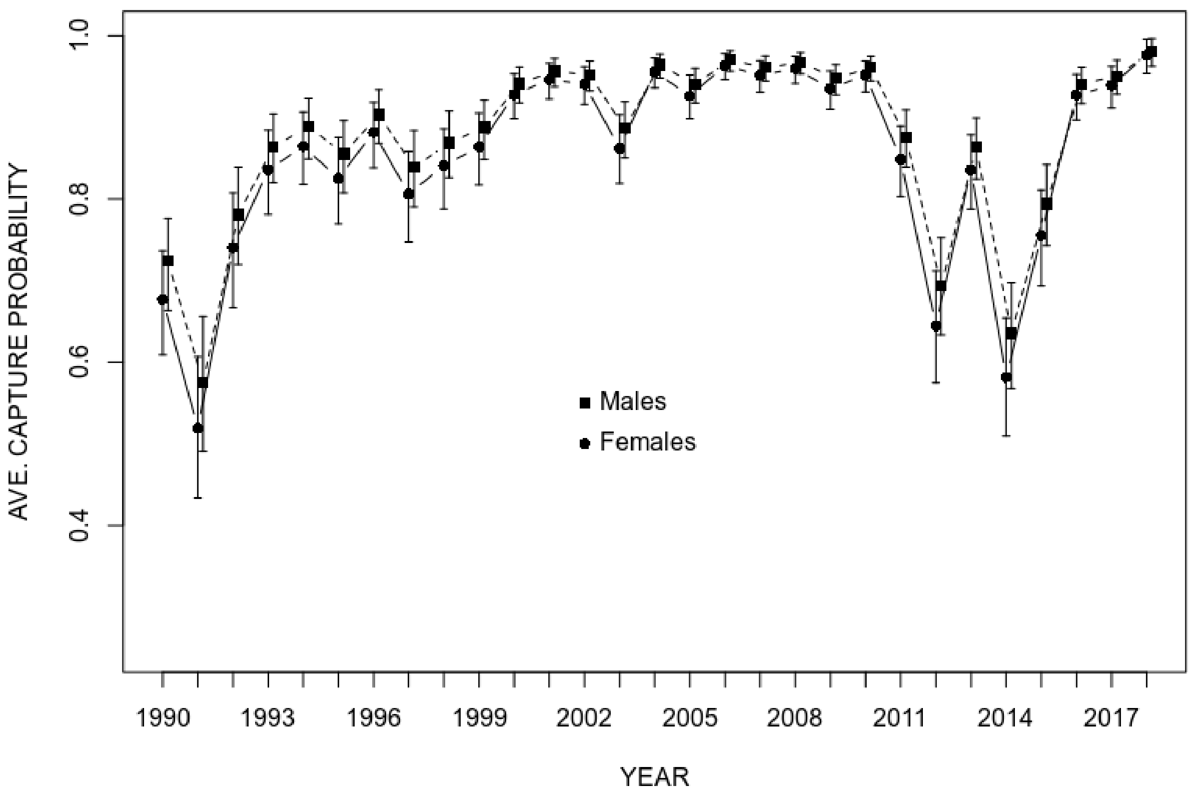

Figure 3. Estimated recapture probability and associated 95% credible intervals of North Atlantic right whales 1990–2018 based on a Bayesian mark-resight/recapture model allowing random fluctuation among years for survival rates, treating capture rates as fixed effects over time, and using both observed and known states as data (from Pace et al. 2017).

CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

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

During 1990–2019, at least 461 calves were born into the population. The number of calves born annually ranged from 0 to 39, and averaged 15 but was highly variable (SD=9.1). No calves were born in the winter of 2017–2018. The fluctuating abundance observed from 1990 to 2019 makes interpreting a count of calves by year less clear than measuring population productivity, which we index by dividing the number of detected calves by the estimated size of the population each year (Apparent Productivity Index or API). Productivity for this stock has been highly variable over time and has been characterized by periodic swings in per capita birth rates (Figure 4). Notwithstanding the high variability observed, as expected for a small population, productivity in North Atlantic right whales lacks a definitive trend. Corkeron *et al.* (2018) found that during 1990–2016, calf count rate increased at 1.98% per year with outlying years of very high and low calf production. This is approximately a third of that found for three different southern right whale (*Eubalaena australis)* populations during the same time period (5.3–7.2%).

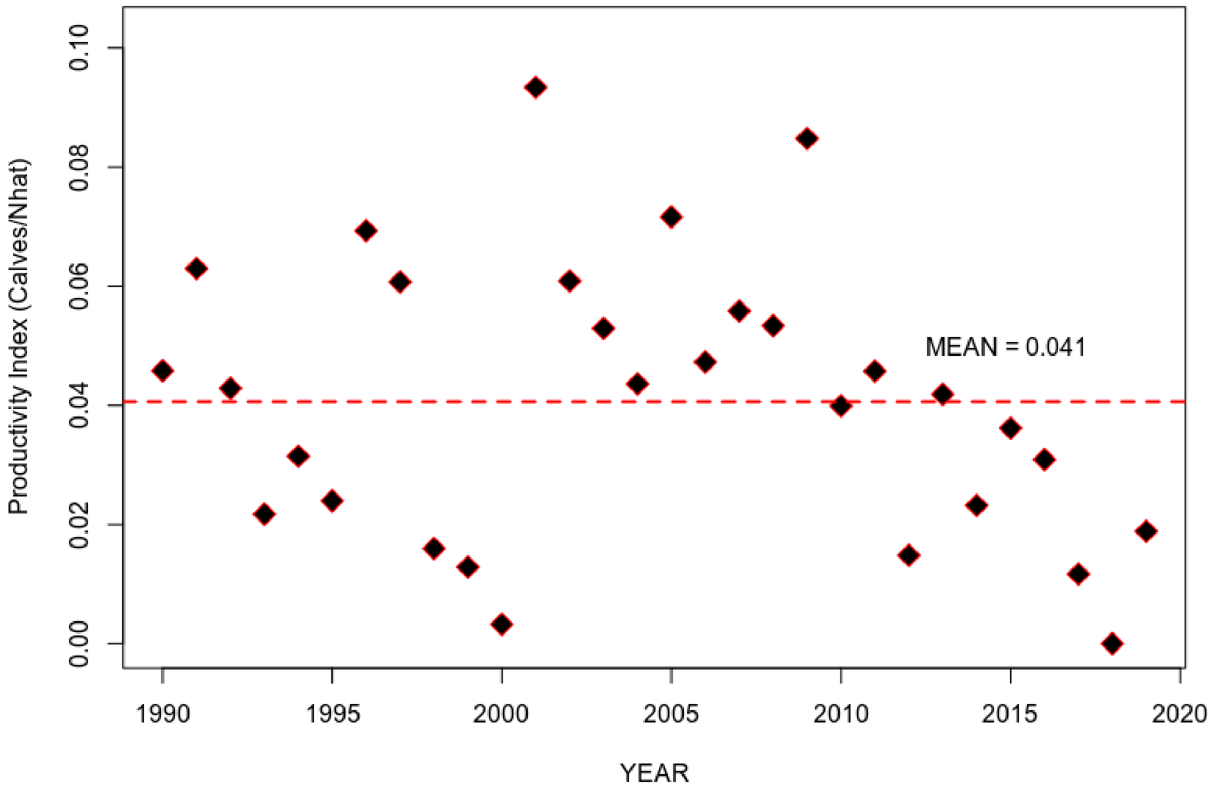

Figure 4. Productivity in the North Atlantic right whale population as characterized by calves detected divided by the estimated population size for each year.

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

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

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

POTENTIAL BIOLOGICAL REMOVAL

Potential biological removal (PBR) is the product of minimum population size, one-half the maximum net productivity rate and a recovery factor for endangered, depleted, threatened stocks, or stocks of unknown status relative to Optimum Sustainable Population (OSP; MMPA Sec. 3. 16 U.S.C. 1362; Wade and Angliss 1997). The recovery factor for right whales is 0.1 because this species is listed as endangered under the Endangered Species Act (ESA). The minimum population size is 364. The maximum productivity rate is 0.04, the default value for cetaceans. PBR for the western North Atlantic stock of the North Atlantic right whale is 0.7 (Table 1).

ANNUAL HUMAN-CAUSED SERIOUS INJURY AND MORTALITY

For the period 2015 through 2019, the annual detected (i.e. observed) human-caused mortality and serious injury to right whales averaged 7.7 (Table 2). This is derived from two components: 1) incidental fishery entanglement records at 5.7 per year, and 2) vessel strike records averaging 2.0 per year.

Injury determinations are made based upon the best available information; these determinations may change with the availability of new information (Henry *et al.* 2022). Only records considered to be confirmed human-caused mortalities or serious injuries are reported in the observed mortality and serious injury (M/SI) rows of Table 2.

Annual rates calculated from detected mortalities are a negatively-biased accounting of human-caused mortality; they represent a definitive lower bound. Detections are irregular, incomplete, and not the result of a designed sampling scheme. Research on other cetaceans has shown the actual number of deaths can be several times higher than observed (Wells *et al.* 2015; Williams *et al*. 2011). The hierarchical Bayesian, state-space model used to estimate North Atlantic right whale abundance (Pace *et al.* 2017) can also be used to estimate total mortality. The estimated annual rate of total mortality using this modeling approach is 27.4 animals for the period 2014–2018 (Pace *et al.* 2021). This estimated total mortality accounts for detected mortality and serious injury (injuries likely to lead to death), as well as undetected (cryptic) mortality within the population. Figure 5 shows the estimates of total mortality for 1990–2018 from the state-space model. Using the methods of Pace *et al.* 2021, the detection rate of mortality and serious injury for the 5-year period 2014–2018 was 29.7% of the model's annual mortality estimates, which is 3.4 times larger than the 8.15 total detected mortalities and serious injuries during 2014–2018. The estimated mortality for 2019 is not yet available because it is derived from a comparison with the population estimate for 2020, which, in turn, is contingent on the processing of all photographs collected through 2020 for incorporation into the state-space model of the sighting histories of individual whales. An analysis of right whale mortalities between 2003 and 2018 found that of the examined non-calf carcasses for which cause of death could be determined, all mortality was human-caused (Sharpe *et al.* 2019). Based on these findings, 100% of the estimated mortality of 27.4 animals per year is assumed to be human-caused. This estimate of total annual human-caused mortality may be somewhat positively biased (i.e., a slight overestimate) given that some calf mortality is likely not human-caused.

There is currently insufficient information to apportion the estimated total right whale mortality by country, e.g., occurring in U.S. versus Canadian waters. Apportioning the estimated total right whale mortality by cause, e.g., entanglement versus vessel collision, also remains uncertain at this time. Pace *et al.* (2021) suggest that entanglements account for more than twice the number of cryptic deaths compared to vessel collisions based on the preponderance of entanglement serious injuries; from 1990 to 2017, NMFS determined a total of 62 right whales were seriously injured, and of these 54 (87%) were due to entanglement. However, during the same period, of 41 right whale carcasses examined for cause of death, 21 (51%) were attributed to vessel collision and 20 (49%) to entanglement. Moore *et al.* (2004) and Sharpe *et al.* (2019) suggest that the underrepresentation of entanglement deaths in examined carcasses may be the result of weight loss in chronically entangled whales, who can become negatively buoyant and sink at the time of death, whereas whales killed instantly by vessel collision may remain available for detection for a longer period and are more likely to be recovered for examination. Both Pace *et al*. (2021) and Moore *et al*. (2020) recommend continued research into the potential mechanisms creating the disparity between apparent causes of serious injuries and necropsy results.

Table 2. Average annual observed and estimated human-caused mortality and serious injury for the North Atlantic right whale (Eubalaena glacialis) from 2015 through 2019. Observed values are from confirmed interactions. Estimated total mortality is model-derived (Pace et al. 2017; Pace et al. 2021). Fishery related serious injuries prevented are a result of successful disentanglement efforts. *The observed incidental fishery interaction count does not include fishery related serious injuries that were prevented by disentanglement.

Years	Source	Annual Average
2015–2019	Observed incidental fishery-related M/SI	5.7*
2015–2019	Observed vessel collisions	2.0
2015–2019	Observed total human-caused M/SI	7.7
2014–2018	Estimated total mortality	27.4
2015–2019	Fishery-related SI prevented	1.6

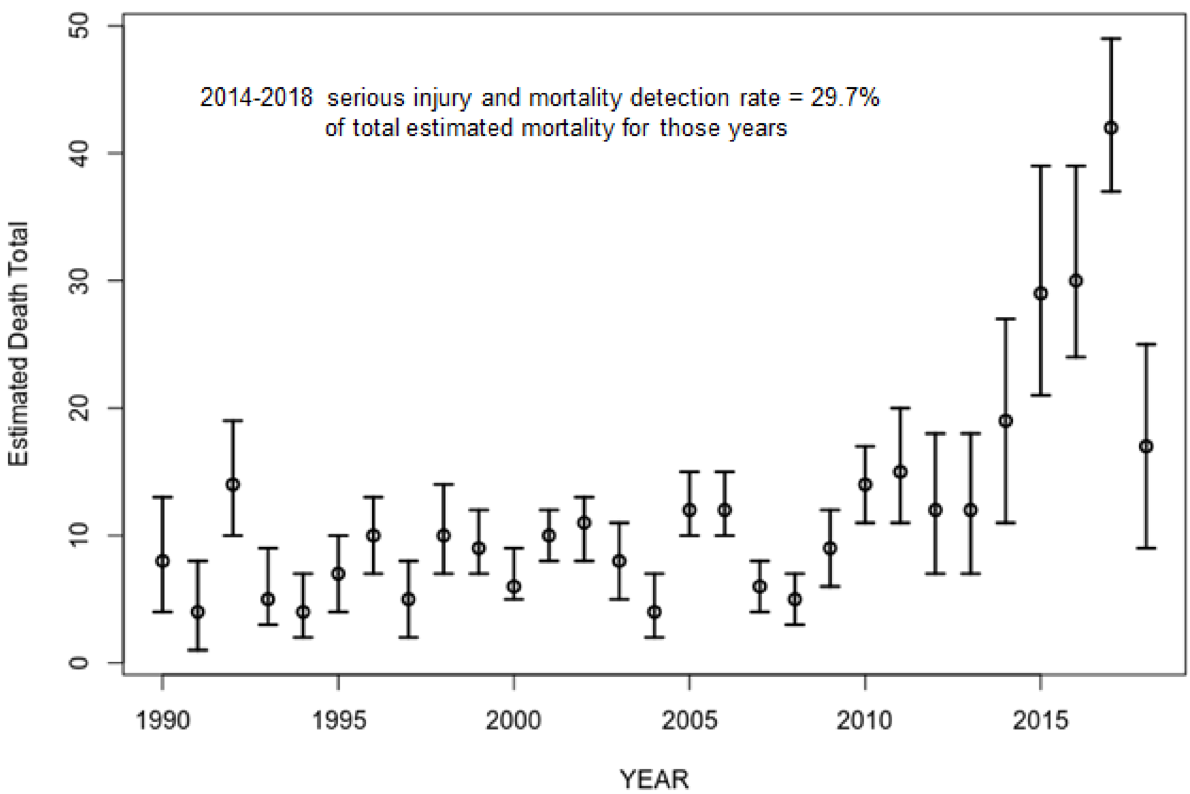

Figure 5. Time series of estimated total mortalities.

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

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

It should be noted that entanglement and vessel collisions may not seriously injure or kill an animal directly, but may weaken or otherwise affect a whale's reproductive success (van der Hoop *et al.* 2017; Corkeron *et al.* 2018). The NMFS serious injury determinations for large whales commonly include animals carrying gear when these entanglements are constricting or are determined to interfere with foraging (Henry *et al.* 2022). Successful disentanglement and subsequent resightings of these individuals in apparent good health are criteria for downgrading an injury to non-serious. However, these and other non-serious injury determinations should be considered to fully understand anthropogenic impacts to the population, especially in cases where females' fecundity may be affected.

Fishery-Related Mortality and Serious Injury

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

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

Whales often free themselves of gear following an entanglement event, and as such scarring may be a better indicator of fisheries interaction rates than entanglement records. Scarring rates suggest that entanglements occur at about an order of magnitude more often than detected from observations of whales with gear on them. A review of scars detected on identified individual right whales over a period of 30 years (1980–2009) documented 1,032 definite, unique entanglement events on the 626 individual whales identified (Knowlton *et al*. 2012). Most individual whales (83%) were entangled at least once, and over half of them (59%) were entangled more than once. About a quarter of the individuals identified in each year (26%) were entangled in that year. Juveniles and calves were entangled at higher rates than were adults. 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 from 1980–2009 that efforts of the prior decade to reduce right whale entanglement had not worked. Using a completely different data source (observed mortalities of eight large whale species, 1970–2009), van der Hoop *et al*. (2012) arrived at a similar conclusion. Similarly, Pace *et al.* (2015), analyzing entanglement rates and serious injuries due to entanglement during 1999–2009, found no support that mitigation measures implemented prior to 2009 had been effective at reducing takes due to commercial fishing. Since 2009, new entanglement mitigation measures (72 FR 193, 05 October 2007; 79 FR 124, 27 June 2014) have been implemented as part of the Atlantic Large Whale Take Reduction Plan, but their effectiveness has yet to be evaluated. One difficulty in assessing mitigation measures is the need for a statistically-significant time series to determine effectiveness.

Other Mortality

Vessel strikes are a major cause of mortality and injury to right whales (Kraus 1990; Knowlton and Kraus 2001, van der Hoop *et al.* 2012). Records from 2015 through 2019 have been summarized in Table 3. Early analyses of the effectiveness of the vessel-strike rule were reported by Silber and Bettridge (2012). van der Hoop *et al.* (2015) concluded that large whale mortalities due to vessel strikes decreased inside active seasonal management areas (SMAs) and increased outside inactive SMAs. Analysis by Laist *et al.* (2014) incorporated an adjustment for drift around areas regulated under the vessel-strike rule and produced weak evidence that the rule was effective inside the SMAs. When simple logistic regression models fit using maximum likelihood-based estimation procedures were applied to previously reported vessel strikes between 2000 and 2017, there was no apparent trend (Hayes *et al.* 2018).

An Unusual Mortality Event was established for North Atlantic right whales in June 2017 due to elevated strandings along the Northwest Atlantic Ocean coast, especially in the Gulf of St. Lawrence region of Canada (https://www.fisheries.noaa.gov/national/marine-life-distress/2017-2020-north-atlantic-right-whale-unusual-mortality-event). There were 30 dead whales documented through December 2019, with 17 whales having evidence of vessel strike or entanglement as the preliminary cause of death. Additionally, eight free-swimming whales were documented as being seriously injured due to entanglements during the time period. Therefore, through December 2019, the number of whales included in the UME was 38, including 30 dead and 8 seriously injured free-swimming whales.

Table 3. Confirmed human-caused mortality and serious injury records of right whales: 2015–2019[a].

Date[b]	Fate	ID	Location[b]	Assigned Cause	Value against PBR[c]	Country[d]	Gear Type[e]	Description
04/06/2015	Serious Injury	C4370	Cape Cod Bay, MA	EN	1	XU	NP	Encircling laceration at fluke insertion with potential to affect major artery. Source of injury likely constricting entanglement. No gear present. Evidence of health decline. No resights.
06/13/2015	Prorated Injury	-	off Westport, NS	EN	.75	XC	NR	Line through mouth, trailing 300-400m ending in 2 balloon-type buoys. Full entanglement configuration unknown. No resights.
09/28/2015	Prorated Injury	-	off Cape Elizabeth, ME	EN	.75	XU	NR	Unknown amount of line trailing from flukes. Attachment point(s) and configuration unknown. No resights.
11/29/2015	Serious Injury	3140	off Truro, MA	EN	1	XU	NR	New, significant ent. injuries indicating constricting wraps. No gear visible. In poor cond. with grey skin and heavy cyamid coverage. No resights.
01/29/2016	Serious Injury	1968	off Jupiter Inlet, FL	EN	1	XU	NP	No gear present, but evidence of recent entanglement of unknown configuration. Significant health decline: emaciated, heavy cyamid coverage, damaged baleen. Resighted in April 2017 still in poor cond.
05/19/2016	Serious Injury	3791	off Chatham, MA	EN	1	XU	NP	New entanglement injuries on peduncle. Left pectoral appears compromised. No gear seen. Significant health decline: emaciated with heavy cyamid coverage. No resights post Aug 2016.
05/03/2016	Mortality	4681	Morris Island, MA	VS	1	US	-	Fresh carcass with 9 deep ventral lacerations. Multiple shorn and/or fractured vertebral and skull bones. Destabilized thorax. Edema, blood clots, and hemorrhage associated with injuries. Proximate COD=sharp trauma. Ultimate COD= exsanguination.

07/26/2016	Serious Injury	1427	Gulf of St Lawrence, QC	EN	1	XC	NP	No gear present, but new entanglement injuries on peduncle and fluke insertions. No gear present. Resights show subsequent health decline: gray skin, rake marks, cyamids.
08/1/2016	Serious Injury	3323	Bay of Fundy, NS	EN	1	XC	NP	No gear present, but new, severe entanglement injuries on peduncle, fluke insertions, and leading edges of flukes. Significant health decline: emaciated, cyamids patches, peeling skin. No resights.
08/13/2016	Serious Injury	4057	Bay of Fundy, NS	EN	1	CN	PT	Free-swimming with extensive entanglement. Two heavy lines through mouth, multiple loose body wraps, multiple constricting wraps on both pectorals with lines across the chest, jumble of gear by left shoulder. Partially disentangled: left with line through mouth and loose wraps at right flipper that are expected to shed. Significant health decline: extensive cyamid coverage. Current entanglement appears to have exacerbated injuries from previous entanglement (see 16Feb2014 event). No resights.
08/16/2016	Prorated Injury	1152	off Baccaro, NS	EN	0.75	XC	NR	Free-swimming with line and buoy trailing from unknown attachment point(s). No resights.
08/28/2016	Serious Injury	2608	off Brier Island, NS	EN	1	XC	NR	Free-swimming with constricting wraps around rostrum and right pectoral. Line trails 50 ft aft of flukes. Significant health decline: heavy cyamid coverage and indication of fluke deformity. No resights.
08/31/2016	Mortality	4320	Sable Island, NS	EN	1	CN	PT	Decomposed carcass with multiple constricting wraps on pectoral with associated bone damage consistent with chronic entanglement.
09/23/2016	Mortality	3694	off Seguin Island, MA	EN	1	CN	PT	Fresh, floating carcass with extensive, constricting entanglement. Thin blubber layer and other findings consistent with prolonged stress due to chronic entanglement. Gear previously reported as unknown.
12/04/2016	Prorated Injury	3405	off Sandy Hook, NJ	EN	0.75	XU	NE	Lactating female. Free-swimming with netting crossing over blowholes and one line over back. Full configuration unknown. Calf not present, possibly already weaned. No resights. Gear type previously reported as NR.
04/13/2017	Mortality	4694	Cape Cod Bay, MA	VS	1	US	-	Carcass with deep hemorrhaging and muscle tearing consistent with blunt force trauma.
06/19/2017	Mortality	1402	Gulf of St Lawrence, QC	VS	1	CN	-	Carcass with acute internal hemorrhaging consistent with blunt force trauma.

06/21/2017	Mortality	3603	Gulf of St Lawrence, QC	EN	1	CN	PT	Fresh carcass found anchored in at least 2 sets of gear. Multiple lines through mouth and constricting wraps on left pectoral. Glucorticoid levels support acute entanglement as COD.
06/23/2017	Mortality	1207	Gulf of St Lawrence, QC	VS	1	CN	-	Carcass with acute internal hemorrhaging consistent with blunt force trauma.
07/04/2017	Serious Injury	3139	off Nantucket, MA	EN	1	XU	NP	No gear present, but evidence of recent extensive, constricting entanglement and health decline. No resights.
07/06/2017	Mortality	-	Gulf of St Lawrence, QC	VS	1	CN	-	Carcass with fractured skull and associated hemorrhaging. Glucorticoid levels support acute blunt force trauma as COD.
07/19/2017	Serious Injury	4094	Gulf of St Lawrence, QC	EN	1	CN	PT	Line exiting right mouth, crossing over back, ending at buoys aft of flukes. Non-constricting configuration, but evidence of significant health decline. No resights.
07/19/2017	Mortality	2140	Gulf of St Lawrence, QC	VS	1	CN	-	Fresh carcass with acute internal hemorrhaging. Glucorticoid levels support acute blunt force trauma as COD.
08/06/2017	Mortality	-	Martha's Vineyard, MA	EN	1	XU	NP	No gear present, but evidence of constricting wraps around both pectorals and flukes with associated tissue reaction. Histopathology results support entanglement as COD.
09/15/2017	Mortality	4504	Gulf of St Lawrence, QC	EN	1	CN	PT	Anchored in gear with extensive constricting wraps with associated hemorrhaging.
10/23/2017	Mortality	-	Nashawena Island, MA	EN	1	XU	NP	No gear present, but evidence of extensive ent involving pectorals, mouth, and body. Hemorrhaging associated with body and right pectoral injuries. Histo results support entanglement as COD.
01/22/2018	Mortality	3893	55 nm E of Virginia Beach, VA	EN	1	CN	PT	Extensive, severe constricting entanglement including partial amputation of right pectoral accompanied by severe proliferative bone growth. COD - chronic entanglement.
02/15/2018	Serious Injury	3296	33 nm E of Jekyll Island, GA	EN	1	XU	NP	No gear present, but extensive recent injuries consistent with constricting gear on right flipper, peduncle, and leading fluke edges. Large portion of right lip missing. Extremely poor condition - emaciated with heavy cyamid load. No resights.
07/13/2018	Prorated Injury	3312	25.6 nm E of Miscou Island, NB	EN	0.75	CN	NR	Free swimming with line through mouth and trailing both sides. Full configuration unknown - unable to confirm extent of flipper involvement. No resights.

07/30/2018	Prorated Injury	3843	13 nm E of Grand Manan, NB	EN	0.75	XC	GU	Free-swimming with buoy trailing 70ft behind whale. Attachment point(s) unknown. Severe, deep, raw injuries on peduncle & head. Partial disentanglement. Resighted with line exiting left mouth and no trailing gear. Possible rostrum and left pectoral wraps, but unable to confirm. Improved health, but final configuration unclear. No additional resights.
08/25/2018	Mortality	4505	Martha's Vineyard, MA	EN	1	XU	NP	No gear present. Evidence of constricting pectoral wraps with associated hemorrhaging. COD - acute entanglement
10/14/2018	Mortality	3515	134 nm E of Nantucket, MA	EN	1	XU	NP	No gear present, but evidence of constricting wraps across ventral surface and at pectorals. COD - acute, severe entanglement.
12/20/2018	Prorated Injury	2310	Nantucket, MA	EN	0.75	XU	NR	Free-swimming with open bridle through mouth. Resight in Apr2019 shows configuration changed, but unable to determine full configuration. Health appears stable.No additional resights
12/1/2018	Serious Injury	3208	South of Nantucket, MA	EN	1	XU	NP	No gear present. Evidence of new, healed, constricting body wrap. Health decline evident - grey, lesions, thin. Previously reported as 24Dec2018
6/4/2019	Mortality	4023	46.4 nm ESE of Perce, QC	VS	1	CN	-	Abrasion, blubber hemorrhage, and muscle contusion caudal to blowholes consistent with pre-mortem vessel strike
6/20/2019	Mortality	1281	27.3 nm E of Magdalen Islands, QC	VS	1	CN	-	Sharp trauma penetrating body cavity consistent with vessel strike. Vessel >65ft based on laceration dimensions.
6/25/2019	Mortality	1514	20.3 nm E of Miscou Island, QC	VS	1	CN	-	Fractured ear bones, skull hemorrhaging, and jaw contusion consistent with blunt trauma from vessel strike.
6/27/2019	Mortality	3450	37.4 nm E of Perce, QC	VS	1	CN	-	Hemothorax consistent with blunt force trauma.
7/4/2019	Serious Injury	3125	35.2 nm E of Perce, QC	EN	1	CN	PT	Free-swimming with extensive entanglement involving embedded head wraps, flipper wraps, and trailing gear. Baleen damaged and protruding from mouth. Partially disentangled: 200-300ft of line removed. Embedded rostrum and blowhole wraps remain, but now able to open mouth. Significant health decline. No resights.
8/6/2019	Mortality	1226	36.4 nm NW of Iles de la Madeleine, NS	EN	1	CN	NR	Constricting rostrum wraps, in anchored or weighted gear. Carcass found with no gear present but evidence of extensive constricting entanglement involving rostrum, gape, both flippers. COD = probable acute entanglement
Assigned Cause					**Five-year mean (US/CN/XU/XC)**			

Vessel strike	2.0 (0.4/1.6/0/0)
Entanglement	5.7 (0/1.95/2.65/1.05)

a. For more details on events please see Henry *et al.* 2022
b. The date sighted and location provided in the table are not necessarily when or where the serious injury or mortality occurred; rather, this information indicates when and where the whale was first reported beached, entangled, or injured.
c. Mortality events are counted as 1 against PBR. Serious injury events have been evaluated using NMFS guidelines (NOAA 2012).
d. CN=Canada, US=United States, XC=Unassigned 1st sight in CN, XU=Unassigned 1st sight in US.
e. H=hook, GN=gillnet, GU=gear unidentifiable, MF=monofilament, NP=none present, NR=none recovered/received, PT=pot/trap, WE=weir.

HABITAT ISSUES

Baumgartner *et al.* (2017) discuss that ongoing and future environmental and ecosystem changes may displace *C. finmarchicus*, or disrupt the mechanisms that create very dense copepod patches upon which right whales depend. One of the consequences of this may be a shift of right whales into different areas with additional anthropogenic impacts to the species. Record *et al.* (2019) described the effects of a changing oceanographic climatology in the Gulf of Maine on the distribution of right whales and their prey. The warming conditions in the Gulf have altered the availability of late stage *C. finmarchicus* to right whales, resulting in a sharp decline in sightings in the Bay of Fundy and Great South Channel over the last decade (Record *et al.* 2019; Davies *et al.* 2019), and an increase in sightings in Cape Cod Bay (Ganley *et al.* 2019). Gavrilchuk *et al*. (2021) suggest that ocean warming in the Gulf of St. Lawrence may eventually compromise the suitability of this foraging area for right whales, potentially displacing them further to the shelf waters east of Newfoundland and Labrador in pursuit of dense *Calanus* patches.

In addition, construction noise and vessel traffic from planned development of offshore wind in southern New England and the mid-Atlantic could result in communication masking, increased risk of vessel strike, or avoidance of wind energy areas. Offshore wind turbines could also influence the hydrodynamics of seasonal stratification and ocean mixing, which, in turn, could influence shelf-wide primary production and copepod distribution (Broström 2008; Paskyabi and Fer 2012; Paskyabi 2015, Carpenter *et al.* 2016, Afsharian *et al*. 2020).

STATUS OF STOCK

The size of this stock is considered to be extremely low relative to OSP in the U.S. Atlantic EEZ. This species is listed as endangered under the ESA and has been declining since 2011 (see Pace *et al.* 2017). The North Atlantic right whale is considered one of the most critically endangered populations of large whales in the world (Clapham *et al.* 1999; NMFS 2017). The observed (and clearly biased low) human-caused mortality and serious injury was 7.7 right whales per year from 2015 through 2019. Using the refined methods of Pace *et al.* (2021), the estimated annual rate of total mortality for the period 2014–2018 was 27.4, which is 3.4 times larger than the 8.15 total derived from reported mortality and serious injury for the same period. Given that PBR has been calculated as 0.7, human-caused mortality or serious injury for this stock must be considered significant. This is a strategic stock because the average annual human-related mortality and serious injury exceeds PBR, and also because the North Atlantic right whale is an endangered species. All ESA-listed species are classified as strategic by definition; therefore, any uncertainties discussed above will not affect the status of stock.

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

STOCK DEFINITION AND GEOGRAPHIC RANGE

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

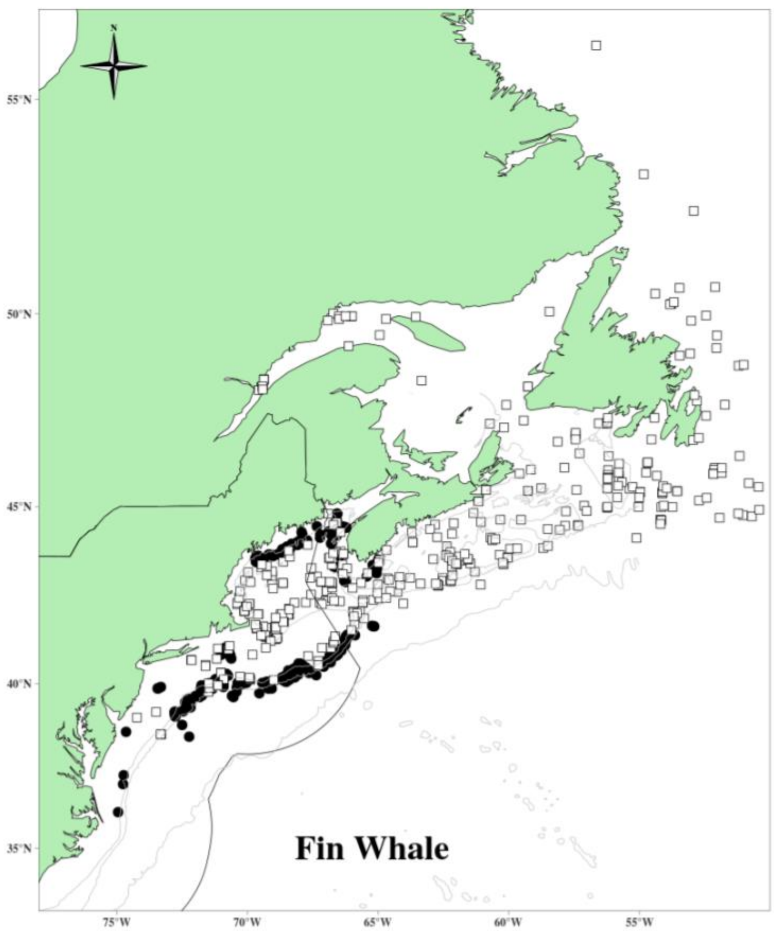

Figure 1. Distribution of fin whale sightings from NEFSC and SEFSC shipboard and aerial surveys during the summers of 1995, 1998, 1999, 2002, 2004, 2006, 2007, 2008, 2010, 2011 and 2016 and DFO's 2007 TNASS and 2016 NAISS surveys. Isobaths are the 100-m, 1,000-m and 4000-m depth contours. Circle symbols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

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

Fin whales are common in waters of the U.S. Atlantic Exclusive Economic Zone (EEZ), principally from Cape Hatteras northward (Figure 1). In a globally-scaled review of sightings data, Edwards *et al.* (2015) found evidence to confirm the presence of fin whales in every season throughout much of the U.S. EEZ north of 30º N; however, densities vary seasonally. Fin whales accounted for 46% of the large whales and 24% of all cetaceans sighted over the continental shelf during aerial surveys (CETAP 1982) between Cape Hatteras and Nova Scotia during 1978–1982. While much remains unknown, the magnitude of the ecological role of the fin whale is impressive. In this region fin whales are the dominant large cetacean species during all seasons, having the largest standing stock, the largest food requirements, and therefore the largest influence on ecosystem processes of any cetacean species (Hain *et al.* 1992; Kenney *et al.* 1997). Acoustic detections of fin whale singers augment and confirm these visual sighting conclusions for males. Recordings from the Atlantic Continental Shelf, and deep-ocean areas detected some level of fin whale singing year round (Watkins *et al*. 1987; Clark and Gagnon 2002; Morano *et al*. 2012; Davis *et al* 2020). These acoustic observations from both coastal and deep-ocean regions support the conclusion that male fin whales are broadly distributed throughout the western North Atlantic for most of the year.

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

POPULATION SIZE

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

Earlier Abundance Estimates

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

Recent Surveys and Abundance Estimates

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

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

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

Month/Year	Area	N_{est}	CV
Jun−Sep 2016	Florida to lower Bay of Fundy	2,390	0.40
Aug−Sep 2016	Bay of Fundy/Scotian Shelf	2,235	0.413
Aug−Sep 2016	Newfoundland/Labrador	2,177	0.465
Jun−Sep 2016	**Florida to Newfoundland/Labrador (COMBINED)**	**6,802**	**0.24**

Minimum Population Estimate

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

Current Population Trend

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

CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

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

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

POTENTIAL BIOLOGICAL REMOVAL

Potential Biological Removal (PBR) is the product of minimum population size, one-half the maximum productivity rate, and a recovery factor (MMPA Sec. 3. 16 U.S.C. 1362; Wade and Angliss 1997). The minimum population size is 65,573. The maximum productivity rate is 0.04, the default value for cetaceans. The recovery factor is 0.10 because the fin whale is listed as endangered under the Endangered Species Act (ESA). PBR for the western North Atlantic fin whale is 11.

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

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

ANNUAL HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

The total annual estimated average human-caused mortality and serious injury for the western North Atlantic fin whale for the period 2015–2019 is presented in Table 3 (Henry *et al.* 2022). Annual rates calculated from detected mortalities should not be considered an unbiased representation of human-caused mortality, but they represent a definitive lower bound. Detections are haphazard and not the result of a designed sampling scheme. As such they represent a minimum estimate of human-caused mortality which is almost certainly biased low. The size of this bias is uncertain.

Table 3. The total annual observed average human-caused mortality and serious injury for the western North Atlantic fin whale (Balaenoptera physalus).

Years	Source	Annual Avg.
2015–2019	Incidental fishery interactions	1.45
2015–2019	Vessel collisions	0.40
TOTAL		1.85

Fishery-Related Serious Injury and Mortality

United States

U.S. fishery interaction records for large whales come through two main sources—dedicated fishery observer data and opportunistic reports collected in the Greater Atlantic Regional Fisheries Office/NMFS entanglement/stranding database. No confirmed fishery-related mortalities or serious injuries of fin whales have been reported in the NMFS Sea Sampling bycatch database (fishery observers) during this reporting period. Records of stranded, floating, or injured fin whales for the reporting period in the Greater Atlantic Regional Fisheries Office/NMFS entanglement/stranding database with substantial evidence of fishery interactions causing injury or mortality are presented in Table 4 (Henry *et al.* 2022). These records are not statistically quantifiable in the same way as the observer fishery records, and they almost surely undercount entanglements for the stock.

Canada

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

Table 4. Confirmed human-caused mortality and serious injury records of fin whales (Balaenoptera physalus) where the cause was assigned as either an entanglement (EN) or a vessel strike (VS): 2015–2019[a].

Date[b]	Fate	ID	Location[b]	Assigned Cause	Value against PBR[c]	Country[d]	Gear Type[e]	Description
06Jun15	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.
06Jul16	Prorated Injury	-	off Truro, MA	EN	0.75	XU	NR	Free-swimming with line trailing 60-70 ft aft of flukes. Attachment point(s) and configuration unknown. No resights.
08Jul16	Prorated Injury	-	off Virginia Beach, VA	EN	0.75	XU	H/MF	Free-swimming with lures in tow along left flipper area. Attachment point(s) and configuration unknown. No resights.
14Dec16	Prorated Injury	-	off Provincetown, MA	EN	0.75	XU	NR	Free-swimming with buoy trailing 6-8ft aft of flukes. Attachment point(s) and configuration unknown. No resights.
30May17	Mortality	-	Port Newark, NJ	VS	1	US	-	Fresh carcass on bow of 656 ft vessel. Speed at strike unknown.
25Aug17	Mortality	-	off Miscou Island, QC	EN	1	CN	PT	Fisher found fresh carcass when hauling gear. Entangled at 78m depth, 51m from trap. Full configuration unknown, but unlikely to have drifted post-mortem into gear.
22Jun18	Mortality	-	16.5 nm E of Gaspe, QC	EN	1	CN	NP	No gear present. Fresh carcass with evidence of constricting entanglement across ventral pleats and peduncle with raw injuries to fluke. Evidence of associated bruising. No necropsy, but COD due to entanglement most parsimonious.
14Oct18	Mortality	Ladders	Cape Cod Bay	VS	1	US	-	Floating carcass with great white shark actively scavenging. Landed on 18 Oct. Necropsied on 19 Oct. Left side not examined due to remote location & no heavy equipment. Additional exam conducted on 30 Oct. Evidence of blunt force trauma - fractured mandibles and rostrum with associated hemorrhaging. Histopathology results support findings.
19Jun19	Mortality	-	20nm E of Miscou, QC	EN	1	CN	NR	No necropsy and no gear present but evidence of extensive constricting entanglement injuries across ventral surface, peduncle and fluke insertion. Entanglement as COD is most parsimonious.
18Jul19	Mortality	-	Portugal Cove South, Avalon, NL	EN	1	CN	PT	Carcass anchored in gear with line through mouth. No necropsy but COD from entanglement is most parsimonious.

Assigned Cause	5-Year mean (US/CN/XU/XC)
Vessel Strike	0.4 (0.4/0/0/0)
Entanglement	1.45 (0/0.8/0.65/0)

a. For more details on events please see Henry *et al.* 2022.

b. The date sighted and location provided in the table are not necessarily when or where the serious injury or mortality occurred; rather, this

information indicates when and where the whale was first reported beached, entangled, or injured.
c. Mortality events are counted as 1 against PBR. Serious injury events have been evaluated using NMFS guidelines (NOAA 2012).
d. US=United States, XU=Unassigned 1st sight in US, CN=Canada, XC=Unassigned 1st sight in CN**.**
e. H=hook, GN=gillnet, GU=gear unidentifiable, MF=monofilament, NP=none present, NR=none recovered/received, PT=pot/trap, WE=weir.

Other Mortality

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

HABITAT ISSUES

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

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

STATUS OF STOCK

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

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

STOCK DEFINITION AND GEOGRAPHIC RANGE

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

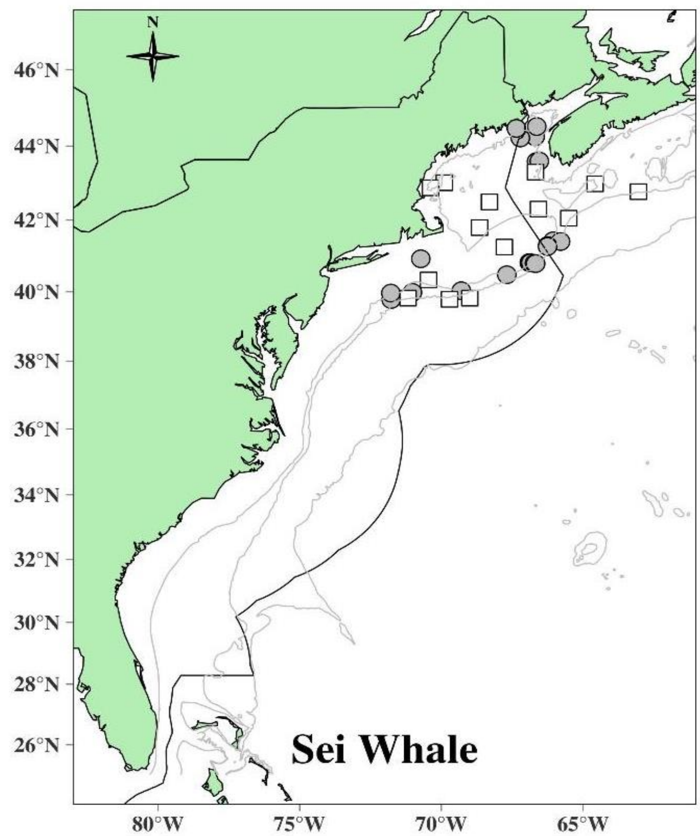

Figure 1. Distribution of sei whale sightings from NEFSC and SEFSC shipboard and aerial surveys during the summers of 1995, 1998, 1999, 2002, 2004, 2006, 2007, 2008, 2010, 2011, and 2016 and DFO's 2007 TNASS and 2016 NAISS surveys. Isobaths are the 100-m, 200-m, 1000-m and 4000-m depth contours.

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

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

Passive acoustic monitoring (PAM) conducted along the Atlantic Continental Shelf and Slope in 2004–2014, detected sei whales calls from south of Cape Hatteras to the Davis Strait with evidence of distinct seasonal and geographic patterns. Davis *et al* 2020 detected peak call occurrence in northern latitudes during summer indicating feeding grounds ranging from Southern New England through the Scotian Shelf. Sei whales were recorded in the southeast on Blake's Plateau in the winter months, but only on the offshore recorders indicating a more pelagic distribution in this region. Persistent year-round detections in Southern New England and the New York Bight highlight this as an important region for the species. The general offshore pattern of sei whale distribution is disrupted during episodic incursions into shallower, more inshore waters. Although known to eat fish in other oceans (Flinn *et al.* 2002), North Atlantic sei whales are largely planktivorous, feeding primarily on euphausiids and copepods (Flinn *et al.* 2002). A review of prey preferences by Horwood (1987) showed that, in the North Atlantic, sei whales seem to prefer copepods over all other prey species. In Nova Scotia, sampled stomachs from captured sei whales showed a clear preference for copepods between June and October, and euphausiids were taken only in May and November (Mitchell 1975). Sei whales are reported in some years in more inshore locations, such as the Great South Channel (in 1987 and 1989) and Stellwagen Bank (in 1986) areas (Payne *et al.* 1990). An influx of sei whales into the southern Gulf of Maine occurred in the summer of 1986 (Schilling *et al.* 1993). Such episodes, often punctuated by years or even decades of absence from an area, have been reported for sei whales from various places worldwide (Jonsgård and Darling 1977).

POPULATION SIZE

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

Earlier Abundance Estimates

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

Recent Surveys and Abundance Estimates

An estimate of 6,292 (CV=1.02) was the springtime (March–May) average abundance estimate generated from spatially- and temporally-explicit density models derived from visual two-team abundance survey data collected between 2010 and 2013 (Table 1; Palka *et al*. 2017). This estimate is for waters between Halifax, Nova Scotia and Florida, where the highest densities of animals were predicted to be on the Scotia shelf outside of U.S. waters. Over 25,000 km of shipboard and over 99,000 km of aerial visual line-transect survey data collected in all seasons in Atlantic waters from Florida to Nova Scotia during 2010–2014 were divided into 10x10 km spatial grid cells and 8-day temporal time periods. Mark-recapture covariate Distance sampling was used to estimate abundance in each spatial-temporal cell which was corrected for perception bias. These density estimates and spatially- and temporally-explicit static and dynamic environmental data were used in Generalized Additive Models (GAMs) to develop spatially- and temporally-explicit animal density-habitat statistical models. These estimates were also corrected by platform- and species-specific availability bias correction factors that were based on dive time patterns.

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

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

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

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

Month/Year	Area	N_{est}	CV
Apr−Jun 1999−2013	Maine to Florida in U.S. waters only	627	0.14
Jul−Sep 1995−2013	Gulf of St Lawrence entrance to Florida	717	0.30
Mar−May 2010−2013	**Halifax, Nova Scotia to Florida**	**6,292**	**1.02**
Jun−Aug 2016	Continental shelf break waters from New Jersey to south of Nova Scotia	28	0.55

Minimum Population Estimate

The minimum population estimate is the lower limit of the two-tailed 60% confidence interval of the log-normally distributed best abundance estimate. This is equivalent to the 20th percentile of the log-normal distribution as specified by (Wade and Angliss 1997). The best estimate of abundance for the Nova Scotia stock sei whales is 6,292 (CV=1.02). The minimum population estimate is 3,098.

Current Population Trend

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

CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

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

POTENTIAL BIOLOGICAL REMOVAL

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

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

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

ANNUAL HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

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

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

Years	Source	Annual Avg.
2015−2019	Incidental fishery interactions	0.40
2015−2019	Vessel collisions	0.20
2015−2019	Other human-caused mortality	0.20
TOTAL		0.80

Fishery-Related Serious Injury and Mortality

No confirmed fishery-related mortalities or serious injuries of sei whales have been reported in the NMFS Sea Sampling bycatch database. A review of the records of stranded, floating, or injured sei whales for the period 2015 through 2019 on file at NMFS found 3 records with substantial evidence of fishery interaction causing serious injury or mortality (Table 4), which results in an annual serious injury and mortality rate of 0.55 sei whales from fishery interactions.

Table 4. Confirmed human-caused mortality and serious injury records of sei whales (Balaenoptera borealis borealis) where the cause was assigned as either an entanglement (EN) or a vessel strike (VS): 2015–2019 [a].

Date[b]	Injury Determination	ID	Location[b]	Assigned Cause	Value against PBR[c]	Country[d]	Gear Type[e]	Description
25Jul16	Mortality	-	Hudson River, Newark, NJ	VS	1	US	-	Fresh carcass on bow of ship (>65 ft). Speed at strike unknown.
11May17	Serious Injury	-	Cape Lookout Bight, NC	EN	1	XU	-	Free-swimming, emaciated, and carrying a large mass of heavily fouled gear consisting of line & buoys crossing over back. Full configuration unknown, but evidence of significant health decline.

12Mar18	Mortality	-	Fanny Keys, FL	EN	1	XU	NR	Carcass with line exiting left side of mouth, across rostrum, and entering right side. Bundle of frayed line lodged in baleen mid-rostrum. Severely emaciated, extensive scavenging. Partial necropsy conducted. Partial healing of lesions + epibiotic growth on line + emaciation = chronic entanglement. Gear not recovered
Assigned Cause				**Five-year Mean (US/CN/XU/XC)**				
Vessel Strike				0.20 (0.20/0/0/0)				
Entanglement				0.40 (0/0/0.40/0)				

a. For more details on events please see Henry *et al.* 2022.
b. The date sighted and location provided in the table are not necessarily when or where the serious injury or mortality occurred; rather, this information indicates when and where the whale was first reported beached, entangled, or injured.
c. Mortality events are counted as 1 against PBR. Serious injury events have been evaluated using NMFS guidelines (NOAA 2012).
d. US=United States, XU=Unassigned 1st sight in US, CN=Canada, XC=Unassigned 1st sight in CN.
e. H=hook, GN=gillnet, GU=gear unidentifiable, MF=monofilament, NP=none present, NR=none recovered/received, PT=pot/trap, WE=weir.

Other Mortality

Records with substantial evidence of vessel collision causing serious injury or mortality are presented in Table 4. One sei whale in 2019 was reported with cause of death as starvation due to plastic ingestion (see Table 3 - other mortality).

HABITAT ISSUES

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

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

STATUS OF STOCK

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

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

STOCK DEFINITION AND GEOGRAPHIC RANGE

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

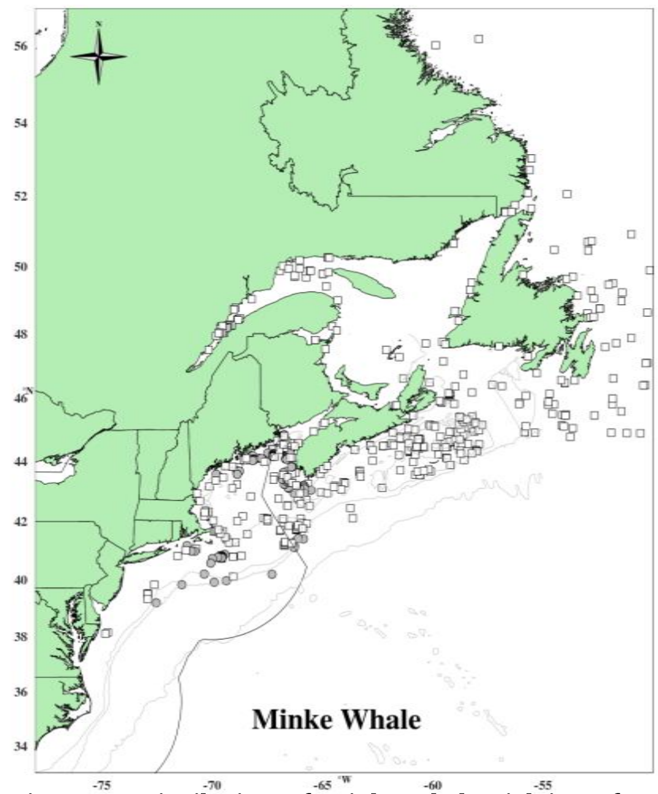

Figure 1. Distribution of minke whale sightings from NEFSC and SEFSC shipboard and aerial surveys during the summers of 1995, 1998, 1999, 2002, 2004, 2006, 2007, 2008, 2010, 2011 and 2016 and DFO's 2007 TNASS and 2016 NAISS surveys. Isobaths are the 100-m, 200-mm 1000-m and 4000-m depth contours. Circle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

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

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

POPULATION SIZE

The best available current abundance estimate for common minke whales in the Canadian East Coast stock is the sum of the 2016 NEFSC and Department of Fisheries and Oceans Canada (DFO) surveys: 21,968 (CV=0.31). Because the survey areas did not overlap, the estimates from the two surveys were added together and the CVs pooled using a delta method to produce a species abundance estimate for the stock area. This is assumed to be the majority of the Canadian East Coast stock. The 2016 estimate is larger than those from 2011 because the 2016 estimate is derived from a survey area extending from Newfoundland to Florida, which is about 1,300,000 km^2 larger than the 2011 survey area. In addition, some of the 2016 survey estimates in U.S. waters were corrected for availability bias (due to diving behavior), whereas the 2011 estimates were not corrected.

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

Earlier Abundance Estimates

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

Recent Surveys and Abundance Estimates

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

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

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

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

Minimum Population Estimate

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

Current Population Trend

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

CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

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

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

POTENTIAL BIOLOGICAL REMOVAL

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

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

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

ANNUAL HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

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

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

Years	Source	Annual Avg.
2015−2019	Incidental fishery interactions non- observed	9.55
2015−2019	U.S. fisheries using observer data	0.2
2015−2019	Vessel collisions	0.8
TOTAL		10.55

Fishery-Related Serious Injury and Mortality

United States

U.S. fishery interaction records for large whales come through 2 main sources – dedicated fishery observer data and opportunistic reports collected in the Greater Atlantic Regional Fisheries Office/NMFS entanglement/stranding database. One confirmed fishery-related mortality or serious injury of minke whales has been reported in the NMFS Sea Sampling bycatch database (fishery observers) during this reporting period (Table 4). A review of the records of stranded, floating, or injured minke whales for the reporting period 2015 through 2019 on file at NMFS, found records in the audited Greater Atlantic Regional Fisheries Office/NMFS entanglement/stranding database with substantial evidence of fishery interactions causing injury or mortality (presented in Table 5; Henry *et al. 2022*). These records are not statistically quantifiable in the same way as the observer fishery records, and they almost surely undercount entanglements for the stock.

Mid-Atlantic Gillnet

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

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

Fishery	Years	Data Type[a]	Observer Coverage[b]	Observed Serious Injury[c]	Observed Mortality	Estimated Serious Injury[c]	Est. Mort.	Est. Combined Mortality	Est. CVs	Mean Combined Annual Mortality	CV of Mean
Mid-Atl. Gillnet	2015	Obs. Data, Weighout	0.06	0	0	0	0	0	0	0.2	0
	2016		0.08	0	1	0	1	1	0		
	2017		0.09	0	0	0	0	0	0		
	2018		0.09	0	0	0	0	0	0		
	2019		0.12	0	0	0	0	0	0		
TOTAL										0.2	0

a. Observer data (Obs. Data) are used to measure bycatch rates and the data are collected within the Northeast Fisheries Observer Program. NEFSC collects Weighout (Weighout) landings data that are used as a measure of total effort for the U.S. gillnet fisheries. Mandatory vessel trip report (VTR; Trip Logbook) data are used to determine the spatial distribution of fishing effort in the Northeast sink gillnet fishery.
b. Observer coverage for the U.S. Northeast gillnet fisheries is based on tons of fish landed.
c. Serious injuries were evaluated for the current period and include both at-sea monitor and traditional observer data (Josephson *et al.* 2022).

Other Fisheries

Confirmed mortalities and serious injuries of common minke whales in the last five years as recorded in the audited Greater Atlantic Regional Office/NMFS entanglement/stranding database are reported in Table 5. Most cases in which gear was recovered and identified involved gillnet or pot/trap gear.

Canada

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

Other Fisheries

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

Table 5. Confirmed human-caused mortality and serious injury records of common minke whales (Balaenoptera acutorostrata acutorostrata): 2015–2019[a].

Date[b]	Injury determination	ID	Location[b]	Assigned Cause[c]	Value against PBR[d]	Country[e]	Gear Type[f]	Description
26Mar15	Serious Injury	-	off Cape Canaveral, FL	EN	1	XU	NR	Evidence of constricting rostrum wrap, but unable to determine if gear still present. Emaciated.
16Apr15	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.
09May15	Mortality	-	Duck, NC	EN	1	XU	GU	Live stranded and euthanized. Embedded gear cutting into bone of mandible. Emaciated.
06Jun15	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.
14Jun15	Prorated Injury	-	off Chatham, MA	EN	0.75	XU	NR	Free-swimming with acorn buoy trailing 20–30 ft. Attachment point(s) and configuration unknown.
23Jun15	Prorated Injury	-	off Ingonish, NS	EN	0.75	CN	PT	Entangled in traps and buoys. Partially disentangled by fisherman. Original and final configuration unknown.
07Jul15	Mortality		off Funk Island, NL	EN	1	CN	PT	Found at 340m depth in between two pots. Gear through mouth and wrapped around peduncle.
18Aug15	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.
01Sept15	Mortality	-	Gloucester, MA	EN	1	US	NP	Evidence of extensive, constricting gear with associated hemorrhaging. No gear present.
21Sept15	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.
06Dec15	Mortality		off Port Joli, NS	EN	1	CN	PT	Live animal anchored in gear. Carcass recovered 4 days later.
03May16	Mortality		Biddeford, ME	EN	1	US	PT	Line through mouth with evidence of constriction across ventral pleats and at peduncle. Hemorrhaging associated with these lesions.
21Jul16	Serious Injury	-	Digby, NS	EN	1	XC	GU	Free-swimming with netting deeply embedded in rostrum. Disentangled, but significant health decline.
15Aug16	Mortality	-	off Seguin Island, ME	EN	1	US	NR	Line exiting mouth leading to weighted/anchored gear.
30Aug16	Mortality		3.1 nm SW of Matinicus Island, ME	EN	1	US	PT	Fresh carcass anchored in gear with evidence of constricting wraps at peduncle and fluke insertions
02Nov16	Prorated Injury	-	Bonne Bay, Gros Morne National Park, NL	EN	0.75	XC	NR	Free-swimming and towing gear. Attachment point(s) and configuration unknown. No resights post 06Nov2016.
27Apr17	Mortality	-	Staten Island, NY	VS	1	US	-	Evidence of bruising on dorsal and right scapular region. Histopathology

								results support blunt trauma from vessel strike most parsimonious as COD.
06Jul17	Mortality	-	Manomet Point, MA	EN	1	US	PT	Live animal anchored in gear. Witnessed becoming entangled in second set. Gear hauled and animal found deceased with line through mouth and constricting wraps on peduncle.
22Jul17	Mortality	-	Piscataqua River, NH	EN	1	US	NP	Evidence of multiple constricting wraps on lower jaw and ventral pleats with associated hemorrhaging. No gear present.
09Aug17	Mortality	-	off Plymouth, MA	EN	1	US	NP	Evidence of constricting entanglement at fluke insertion, across fluke blades and ventral pleats. No necropsy but fresh carcass with extensive injuries supports COD of entanglement as most parsimonious.
11Aug17	Prorated Injury	-	off York, ME	EN	0.75	US	NR	Partially disentangled from anchoring gear. Final configuration unknown.
12Aug17	Mortality	-	off Tremont, ME	EN	1	US	GU	Fresh carcass of a pregnant female in gear. Constricting wrap injuries with associated hemorrhaging on dorsal and ventral surfaces and flukes.
14Aug17	Mortality	-	Pt. Judith, RI	EN	1	US	NP	Evidence of constricting entanglement along left side with associated hemorrhaging. Found floating in stationary offshore fishing trap, but not entangled in trap gear. No gear present on animal.
17Aug17	Mortality	-	Rye, NH	EN	1	US	NR	Evidence of constricting wraps on fluke blades and peduncle. Documented with line in baleen, but not present at time of necropsy. Limited necropsy, but extent of injuries and robust animal with evidence of recent feeding supports COD of entanglement as most parsimonious.
28Aug17	Mortality	-	off Portland, ME	EN	1	US	PT	Fresh carcass anchored in gear. Endline wrapped around mouth and laceration from constricting gear on peduncle. Mud on flippers and mouth.
30Aug17	Mortality	-	off North Cape, PEI	EN	1	CN	NR	Fresh carcass in gear. Full configuration unclear, but complex enough to not have drifted into post-mortem.
04Sept17	Mortality	-	St. Carroll's, NL	EN	1	CN	NE	Alive in herring net. Found dead the next day. Fisher pulled carcass ashore and removed the net.
06Sept17	Mortality		Newport, RI	VS	1	US	-	Hemorrhaging at left pectoral, left body, and aft of blowholes. Histopathology results support blunt trauma from vessel strike as COD.
17Sept17	Mortality	-	Henry Island, NS	EN	1	CN	NR	Fresh carcass with gear in mouth and around flukes. Evidence of constricting wrap on dorsum. No necropsy, but configuration complex enough that unlikely to have drifted into gear post-mortem.
26Sept17	Prorated Injury	-	off Richbuctou, NB	EN	0.75	CN	NR	Animal initially anchored in gear then not resighted. Unable to confirm if gear free, partially entangled, or drowned.

27Sept17	Mortality	-	5.7nm NE of Richbuctou, NB	EN	1	CN	NP	No gear present. Fresh carcass with evidence of constricting wraps.
10Oct17	Mortality		off Rockland, ME	EN	1	US	PT	Entangled in 2 different sets of gear. Constricting wrap around lower jaw. Found at depth when fisher hauled gear.
09Feb18	Mortality	-	Tiverton, Long Island, NS	EN	1	XC	NP	No gear present. Evidence of constricting body, flipper, and peduncle wraps. No necropsy conducted, but COD from entanglement most parsimonious.
25May18	Mortality	-	Digby, NS	VS	1	CN	-	Fresh carcass in harbor with large area of hemorrhage aft of blowholes. Necropsy did not state COD, but blunt trauma from vessel strike most parsimonious.
11Jun18	Mortality	-	Cape Dauphin, NS	EN	1	CN	PT	Fresh, pregnant carcass anchored in gear.
19Jun18	Mortality	-	East Point, PEI	EN	1	CN	NP	No gear present. Fresh, pregnant carcass with evidence of extensive constricting body and peduncle wraps with associated hemorrhaging.
22Jun18	Prorated Injury	-	4.5 nm N of Grand Manan, NB	EN	0.75	XC	NR	Full configuration unclear - line across back, one buoy under left pectoral and another trailing 30–40ft aft. Reported as anchored but unable to confirm. Response team was not able to relocate.
24Jun18	Mortality	-	Wellfleet, MA	EN	1	XU	GN	Evidence of extensive constricting body and mouth wraps with associated hemorrhaging. Deep lacerations at fluke insertion from constricting gear. COD - peracute underwater entrapment.
07Jul18	Mortality	-	1.6 nm E of Newcastle, NH	EN	1	US	PT	Anchored in gear with line through mouth and wrapping around body. Associated bruising at right corner of mouth. COD - peracute underwater entrapment.
22Jul18	Mortality	-	Cape Neddick, ME	EN	1	XU	NP	No necropsy, but evidence of constricting wrap at fluke insertion with associated hemorrhaging. Histopathology confirms pre-mortem human-induced trauma.
28Jul18	Mortality	-	Biddeford, ME	EN	1	XU	NP	No gear present, but evidence of constricting gear with associated bruising at mouth, around body and peduncle.
06Aug18	Prorated Injury	-	Fish Cove Point, NL	EN	0.75	CN	NE	Free-swimming towing net with float attached. Member of public cut off float. Original and final configuration unknown.
29Aug18	Prorated Injury	-	7.5 nm SE of Chatham, MA	EN	0.75	XU	NR	Free-swimming with buoy near flukes, full configuration unknown.
03Sep18	Mortality	-	Nancy Head, Campobello, NB	EN	1	CN	WE, SE	Live animal entrapped. Failed attempt by fisher to remove animal with seine. Animal became entangled in seine and drowned.
16Sep18	Mortality	-	0.7 nm SSE of Rye, NH	EN	1	US	PT	Fresh carcass anchored in gear. Constricting body, jaw, peduncle, and fluke wraps with associated hemorrhaging.
07Nov18	Mortality	-	Tangier Island, VA	EN	1	XU	NE	Constricting gear with associated hemorrhaging partly amputating tip of

								rostrum. Poor body condition. COD - chronic entanglement.
25Dec18	Mortality	-	Yarmouth Bar, NS	EN	1	XC	NP	No gear present. Evidence of constricting entanglement on head, ventral pleats, peduncle and flukes. No necropsy, but COD from entanglement most parsimonious.
27Mar19	Mortality	-	Duxbury, MA	EN	1	US	NR	Carcass with line through mouth when first documented, but not present at exam. No COD determined, but mouth abrasion with associated hemorrhaging in muscle and staining of bone is consistent with pre-mortem entanglement.
05Jun19	Mortality	-	Queensland Beach, NS	EN	1	CN	NP	No necropsy, but evidence of multiple constricting body and peduncle wraps. Fluke cleanly severed. Likely removed post-mortem. COD = EN most parsimonious.
04Aug19	Prorated Injury	-	6.0 nm E of Montauk, NY	EN	0.75	XU	NR	Free-swimming with line crossing over back just in front of dorsal fin. Line fouled with growth. Attachment point(s) and full configuration unknown.
09Aug19	Prorated Injury	-	Rigolet, Labrador	EN	0.75	CN	NE	Anchored with line around rostrum and constricting peduncle wraps. Partially disentangled. Final configuration unknown.
21Aug19	Prorated Injury	-	Mer et Monde, QC	EN	0.75	XC	NR	Free-swimming with line over back and possibly through mouth. Full configuration and attachment point(s) unknown.
01Sep19	Prorated Injury	-	31.3 nm SE of Chatham, MA	EN	0.75	XU	NR	Free-swimming with buoy trailing from fluke area. Attachment point(s) and full configuration unknown.
10Sep19	Prorated Injury	-	0.1 nm N of Mattinicus Rock, ME	EN	0.75	XU	NR	Unable to confirm if anchored or free-swimming. Full configuration and attachment point(s) unknown.
19Sep19	Mortality	-	off Burnt Island, ME	EN	1	US	-	No gear present, but evidence of constricting body, peduncle, and fluke wraps. No necropsy, but COD due to EN is most parsimonious.

Assigned Cause	**5-Year mean (US//CN/XU/XC)**
Vessel strike	0.8 (0.6/ 0.2/0/0)
Entanglement	9.55 (2.95/ 3.2/2.35/1.05)

a. For more details on events please see Henry *et al.* 2022.
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. Assigned cause: EN=entanglement, VS=vessel strike, ET=entrapment (summed with entanglement).
d. Mortality events are counted as 1 against PBR. Serious injury events have been evaluated using NMFS guidelines (NOAA 2012).
e. US=United States, XU=Unassigned 1st sight in U.S., CN=Canada, XC=Unassigned 1st sight in CN.
f. H=hook, GN=gillnet, GU=gear unidentifiable, MF=monofilament, NP=none present, NR=none recovered/received, PT=pot/trap, WE=weir.

Other Mortality

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

United States

Common minke whales inhabit coastal waters during much of the year and are thus susceptible to collision with vessels. Vessel strike interactions in U.S. and Canadian waters are reported in Table 5. In January 2017, a minke whale Unusual Mortality Event (UME) was declared for the U.S. Atlantic coast due to elevated numbers of mortalities. From January 2017 to December 2019, 79 minke whales stranded between Maine and South Carolina. Preliminary findings in several of the whales have shown evidence of human interactions or infectious disease. This most recent UME is ongoing (https://www.fisheries.noaa.gov/national/marine-life-distress/2017-2021-minke-whale-unusual-mortality-event-along-atlantic-coast#minke-whale-strandings; accessed 27Jan2021). Anthropogenic mortalities and serious injuries that occurred in 2017–2019 as part of this UME are included in Table 5.

Canada

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

The Whale Release and Strandings program reports common minke whale stranding mortalities in Newfoundland and Labrador (Ledwell and Huntington 2015, 2016, 2017, 2018, 2019). Those that have been determined to be human-caused serious injury or mortality are included in Table 5.

HABITAT ISSUES

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

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

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

STATUS OF STOCK

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

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

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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 (Figure 1; CETAP 1982; Payne *et al*. 1984). In winter, the range is in the mid-Atlantic Bight and extends outward into oceanic waters (Payne *et al*. 1984). In general, the population occupies the mid-Atlantic continental shelf edge year round, and is rarely seen in the Gulf of Maine (Payne *et al*. 1984). During 1990, 1991 and 1993, spring/summer surveys conducted along the continental shelf edge and in deeper oceanic waters sighted Risso's dolphins associated with strong bathymetric features, Gulf Stream warm-core rings, and the Gulf Stream north wall (Waring *et al*. 1992, 1993; Hamazaki 2002). Sightings during 2016 surveys were concentrated along the shelf break (Figure 1; NEFSC and SEFSC 2018).

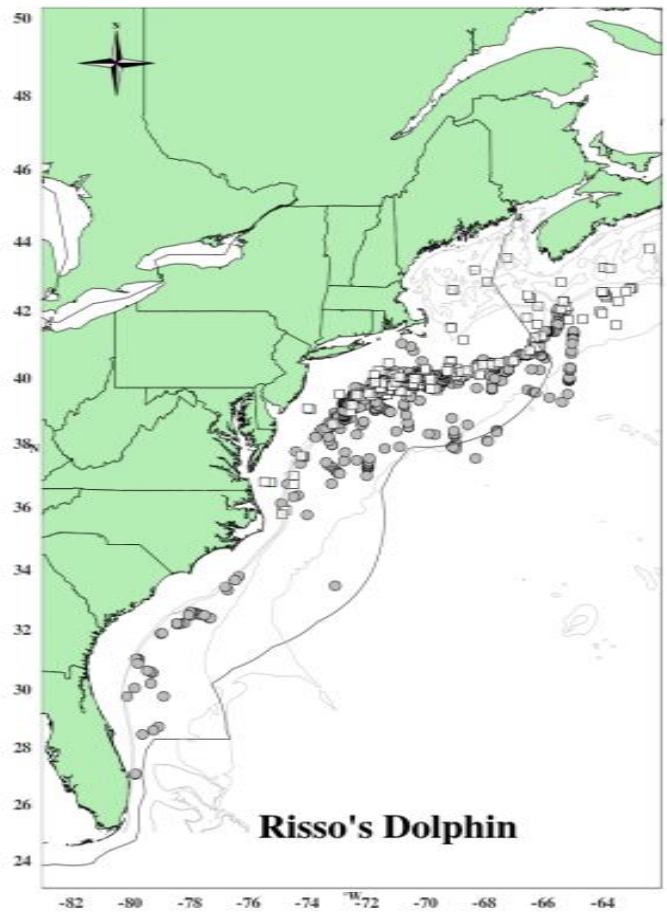

Figure 1. Distribution of Risso's dolphin sightings from NEFSC and SEFSC shipboard and aerial surveys during the summers of 1995, 1998, 1999, 2002, 2004, 2006, 2007, 2008, 2010, 2011 and 2016 and Department of Fisheries and Oceans Canada 2007 TNASS and 2016 NAISS surveys. Isobaths are the 100-m, 1000-m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

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

POPULATION SIZE

The best abundance estimate for Risso's dolphins is the sum of the estimates from the 2016 NEFSC and Department of Fisheries and Oceans Canada (DFO) surveys—35,215 (CV=0.19; Table 1). Because the survey areas did not overlap, the estimates from the two surveys were added together and the CVs pooled using a delta method to produce a species abundance estimate for the stock area.

Earlier Abundance Estimates

Please see Appendix IV for 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

The Department of Fisheries and Oceans, Canada (DFO) generated Risso's dolphin estimates from a large-scale aerial survey of Atlantic Canadian shelf and shelf break habitats extending from the northern tip of Labrador to the U.S. border off southern Nova Scotia in August and September of 2016 (Lawson and Gosselin 2018). A total of 29,123 km of effort were flown over the Gulf of St. Lawrence/Bay of Fundy/Scotian Shelf strata and 21,037 over the Newfound/Labrador strata. The Bay of Fundy/Scotian shelf portion of the Risso's dolphin population was estimated as 6,073 (CV=0.445).

Abundance estimates of 21,897 (CV=0.23) and 7,245 (CV=0.44) Risso's dolphins were generated from vessel surveys conducted in U.S. waters of the western North Atlantic during the summer of 2016 (Table 1; Garrison 2020; Palka 2020). One survey was conducted from 27 June to 25 August in waters north of 38ºN latitude and consisted of 5,354 km of on-effort trackline along the shelf break and offshore to the outer limit of the U.S. EEZ (NEFSC and SEFSC 2018). The second vessel survey covered waters from Central Florida to approximately 38ºN latitude between the 100-m isobaths and the outer limit of the U.S. EEZ during 30 June–19 August. A total of 4,399 km of trackline was covered on effort (NEFSC and SEFSC 2018). Both surveys utilized two visual teams and an independent observer approach to estimate detection probability on the trackline (Laake and Borchers 2004). Mark-recapture distance sampling was used to estimate abundance. Estimates from the two surveys were combined and CVs pooled to produce a species abundance estimate for the stock area.

Table 1. Summary of recent abundance estimates for the western North Atlantic Risso's dolphin (Grampus griseus), by month, year, and area covered during each abundance survey, resulting abundance estimate (N_{est}) and coefficient of variation (CV).

Month/Year	Area	N_{est}	CV
Jun–Sep 2016	Central Florida to Central Virginia	7,245	0.44
Jun–Sep 2016	Central Virginia to lower Bay of Fundy	21,897	0.23
Aug–Sep 2016	Gulf of St. Lawrence/Bay of Fundy/Scotian Shelf	6,073	0.445
Jun–Sep 2016	Central Florida to Gulf of St. Lawrence/Bay of Fundy/Scotian Shelf - COMBINED	35,215	0.19

Minimum Population Estimate

The minimum population estimate is the lower limit of the two-tailed 60% confidence interval of the log-normally distributed best abundance estimate. This is equivalent to the 20th percentile of the log-normal distribution as specified by Wade and Angliss (1997). The best estimate of abundance for Risso's dolphins is 35,215 (CV=0.19), obtained from the 2016 surveys. The minimum population estimate for the western North Atlantic Risso's dolphin is 30,051.

Current Population Trend

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

CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

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

POTENTIAL BIOLOGICAL REMOVAL

Potential Biological Removal (PBR) is the product of minimum population size, one-half the maximum productivity rate, and a recovery factor (MMPA Sec. 3. 16 U.S.C. 1362; Wade and Angliss 1997). The minimum population size is 30,051. 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 Optimum Sustainable Population (OSP), and the CV of the average mortality estimate is less than 0.3 (Wade and Angliss 1997). PBR for the western North Atlantic stock of Risso's dolphin is 301 (Table 2).

Table 2. Best and minimum abundance estimates for the western North Atlantic Risso's dolphin (Grampus griseus) with Maximum Productivity Rate (R_{max}), Recovery Factor (F_r) and PBR.

N_{est}	CV	N_{min}	F_r	R_{max}	PBR
35,215	0.19	30,051	0.5	0.04	301

ANNUAL HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

Total annual estimated average human-caused mortality or serious injury to this stock during 2015–2019 was 35 Risso's dolphins, derived from estimated mortalities and serious injuries in observed fisheries (CV=0.09; Tables 3, 4). Key uncertainties include the potential that the observer coverage was not representative of the fishery during all times and places.

Table 3. Total annual estimated average human-caused mortality and serious injury for the western North Atlantic Risso's dolphin (Grampus griseus).

Years	Source	Annual Avg.	CV
2015–2019	U.S. fisheries using observer data	34	0.09
2015–2019	Non-fishery human caused stranding mortalities	0	-
TOTAL		34	0.09

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 2015–2019 are documented in Garrison and Stokes (2017, 2019, 2020a, 2020b, 2021). 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 4 for bycatch estimates and observed mortality and serious injury for the current 5-year period, and Appendix V for historical bycatch information.

Northeast Bottom Trawl

Two Risso's dolphins were observed taken in northeast bottom trawl fisheries in 2016 (Table 4). Annual Risso's dolphin mortalities were estimated using annual stratified ratio-estimator methods (Lyssikatos and Chavez-Rosales.

2021). See Table 4 for bycatch estimates and observed mortality and serious injury for the current 5-year period, and Appendix V for historical bycatch information.

Mid-Atlantic Bottom Trawl

Risso's dolphins have been observed taken in mid-Atlantic bottom trawl fisheries (Table 4). Annual Risso's dolphin mortalities were estimated using annual stratified ratio-estimator methods (Lyssikatos and Chavez-Rosales 2021). See Table 4 for bycatch estimates and observed mortality and serious injury for the current 5-year period, and Appendix V for historical bycatch information.

Northeast Sink Gillnet

In the northeast sink gillnet fishery, Risso's dolphin interactions have historically been rare, but in 2019 one animal was observed in the waters south of Massachusetts (2016; Orphanides 2019, 2020, 2021; Precoda and Orphanides 2022). See Table 4 for bycatch estimates and observed mortality and serious injury for the current 5-year period, and Appendix V for historical bycatch information.

Table 4. 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	2015	Obs. Data, Logbook	0.12	2	0	8.4	0	8.4	0.71	5.0 (0.44)
	2016		0.15	1	1	10.5	5.6	16.1	0.57	
	2017		0.12	1	0	0.2	0	0.2	1	
	2018		0.10	1	0	0.2	0	0.2	0.94	
	2019		0.10	0	0	0	0	0	0	
Northeast Sink Gillnet	2015	Obs. Data, Trip Logbook, Allocated Dealer Data	0.14	0	0	0	0	0	0	1.1 (0.7)
	2016		0.10	0	0	0	0	0	0	
	2017		0.12	0	0	0	0	0	0	
	2018		0.11	0	0	0	0	0	0	
	2019		0.13	0	1	0	5.3	5.3	0.7	
Northeast Bottom Trawl	2015	Obs. Data, Weighout	0.19	0	0	0	0	0	0	3.4 (0.88)
	2016		0.12	0	2	0	17	17	0.88	
	2017		0.16	0	0	0	0	0	0	
	2018		0.12	0	0	0	0	0	0	
	2019		0.16	0	0	0	0	0	0	
Mid-Atlantic Bottom Trawl	2015	Obs. Data, Dealer Data	0.09	2	1	27	13	40	0.63	25 (0.33)
	2016		0.10	0	4	0	39	39	0.56	
	2017		2.10	2	5	12	31	43	0.51	
	2018		0.12	0	0	0	0	0	0	
	2019		0.12	0	0	0	0	0	0	
TOTAL										35 (0.254)

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 2015–2019 period and include both at-sea monitor and traditional observer data (Josephson *et al*. 2022).

Other Mortality

From 2015 to 2019, 31 Risso's dolphin strandings were recorded along the U.S. Atlantic coast (NOAA National

Marine Mammal Health and Stranding Response Database unpublished data, accessed 17 November 2020). None of the animals had indications of human interaction.

Table 3. Risso's dolphin (Grampus griseus) reported strandings along the U.S. Atlantic coast and Puerto Rico, 2015–2019.

STATE	2015	2016	2017	2018	2019	TOTALS
Massachusetts	1	2	14	0	0	17
Rhode Island	0	0	1	0	0	1
New York[a]	2	0	0	0	3	5
New Jersey	0	0	1	0	0	1
Maryland[b]	0	0	0	0	1	1
North Carolina	0	0	1	1	1	3
Florida	0	2	1	0	0	4
TOTAL	4	4	4	1	5	31

a. One animal in 2019 released alive.
b. One animal in 2019 alive, left at site.

Stranding data probably underestimate the extent of mortality and serious injury because all of the marine mammals that die or are seriously injured may not wash ashore, nor will all of those that do wash ashore necessarily show signs of entanglement or other fishery-interaction. Finally, the level of technical expertise among stranding network personnel varies widely as does the ability to recognize signs of fishery interaction.

HABITAT ISSUES

The chronic impacts of contaminants (polychlorinated biphenyls [PCBs] and chlorinated pesticides [DDT, DDE, dieldrin, etc.]) on marine mammal reproduction and health are of concern (e.g., Storelli and Macrotrigiano 2000; Pierce *et al.* 2008; Jepson *et al.* 2016; Hall *et al.* 2018; Murphy *et al.* 2018), but research on contaminant levels for the western north Atlantic stock of Risso's dolphins is lacking.

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

STATUS OF STOCK

Risso's dolphins are not listed as threatened or endangered under the Endangered Species Act and the Western North Atlantic stock is not considered strategic under the Marine Mammal Protection Act. The 2015–2019 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 is expected that these uncertainties will have little effect on the designation of the status of this stock.

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

STOCK DEFINITION AND GEOGRAPHIC RANGE

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

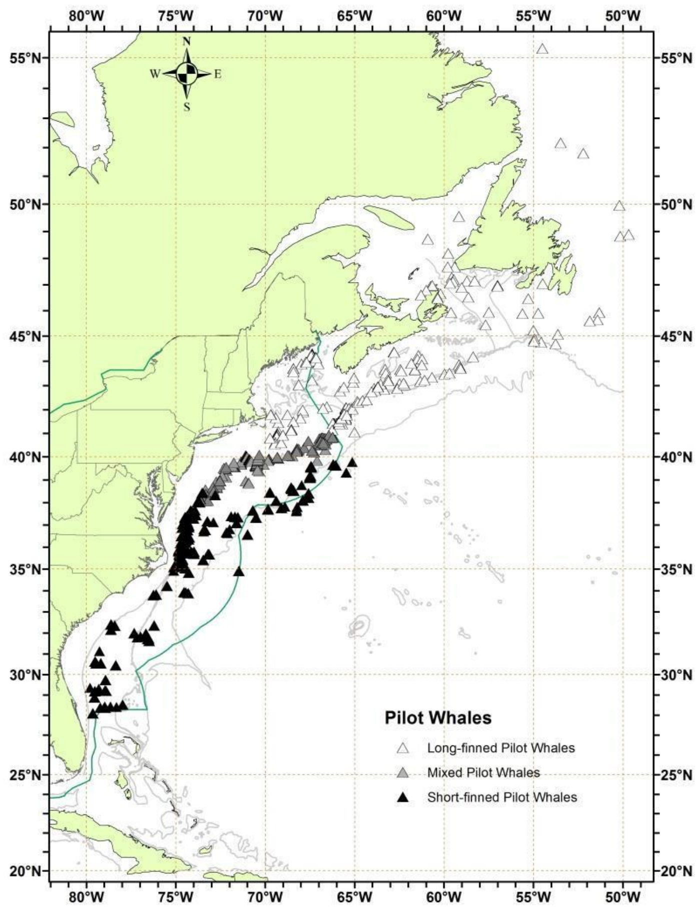

Figure 1. Distribution of long-finned (open symbols), short-finned (black symbols), and possibly mixed (gray symbols; could be either species) pilot whale sightings from NEFSC and SEFSC shipboard and aerial surveys during the summers of 1998, 1999, 2002, 2004, 2006, 2007, 2011, and 2016 and the Department of Fisheries and Oceans Canada 2007 TNASS and 2016 NAISS surveys. The inferred distribution of the two species is preliminary and is valid for June–August only. Isobaths are the 1000-m and 3000-m depth contours. The U.S. EEZ is also displayed in green.

In U.S. Atlantic waters, pilot whales (*Globicephala* sp.) are distributed principally along the continental shelf edge off the northeastern U.S. coast in winter and early spring (CETAP 1982; Payne and Heinemann 1993; Abend and Smith 1999; Hamazaki 2002). In late spring, pilot whales move onto Georges Bank and into the Gulf of Maine and more northern waters, and remain in these areas through late autumn (CETAP 1982; Payne and Heinemann 1993). Pilot whales tend to occupy areas of high relief or submerged banks. They are also associated with the Gulf Stream wall and thermal fronts along the continental shelf edge (Waring *et al.* 1992). Long-finned and short-finned pilot whales overlap spatially along the mid-Atlantic shelf break between Delaware and the southern flank of Georges Bank (Payne and Heinemann 1993; Rone and Pace 2012). Long-finned pilot whales have occasionally been observed stranded as far south as Florida, and short-finned pilot whales have occasionally been observed stranded as far north as Massachusetts. The exact latitudinal ranges of the two species therefore remain uncertain, although south of Cape Hatteras, most pilot whale sightings are expected to be short-finned pilot whales, while north of ~42°N most pilot whale sightings are expected to be long-finned pilot whales (Figure 1; Garrison and Rosel 2017).

POPULATION SIZE

The best available estimate for long-finned pilot whales in the western North Atlantic is 39,215 (CV=0.30; Table 1; Garrison 2020; Palka 2020; Lawson and Gosselin 2018). This estimate is the sum of the estimates generated from the northeast U.S. summer 2016 surveys covering U.S. waters from central Virginia to Maine and the Department of Fisheries and Oceans Canada summer 2016 survey covering Canadian waters from the U.S. to Labrador. Because the survey areas did not overlap, the estimates from the two surveys were added together and the CVs pooled using a delta method to produce a species abundance estimate for the stock area. The 2016 estimate is larger than those from 2011 because the 2016 estimate is derived from a survey area extending from Newfoundland to Florida, which is about 1,300,000 km^2 larger than the 2011 survey area. In addition, the 2016 survey estimates in U.S. waters were corrected for availability bias (due to diving behavior), whereas the 2011 estimates were not corrected. These survey data have been combined with an analysis of the spatial distribution of the 2 species based on genetic analyses of biopsy samples to derive separate abundance estimates (Garrison and Rosel 2017).

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

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

Recent Surveys and Abundance Estimates for *Globicephala* sp.

Abundance estimates of 8,166 (CV=0.31) and 25,114 (CV=0.27) *Globicephala* sp. were generated from vessel surveys conducted in the northeast and southeast U.S., respectively, during the summer of 2016. The Northeast survey was conducted during 27 June–25 August and consisted of 5,354 km of on-effort trackline. The majority of the survey was conducted in waters north of 38ºN latitude and included trackline along the shelf break and offshore to the U.S. EEZ. Pilot whale sightings were concentrated along the shelf-break between the 1,000-m and 2,000-m isobaths and along Georges Bank (NMFS 2017). The Southeast vessel survey covered waters from Central Florida to approximately 38ºN latitude between the 100-m isobaths and the U.S. EEZ during 30 June–19 August. A total of 4,399 km of trackline was covered on effort. Pilot whales were observed in high densities along the shelf-break between Cape Hatteras and New Jersey and also in waters further offshore in the mid-Atlantic and off the coast of Florida (NMFS 2017; Garrison and Palka 2018). Both the Northeast and Southeast surveys utilized two visual teams and an independent observer approach to estimate detection probability on the trackline (Laake and Borchers 2004). Mark-recapture distance sampling was used to estimate abundance. A logistic regression model was used to estimate the abundance of long-finned pilot whales from these surveys. For the northeast survey, this resulted in an abundance estimate of 10,997 (CV=0.51) long-finned pilot whales. In the southeast, the model indicated that this survey included habitats expected to exclusively contain short-finned pilot whales so no estimate for long-finned pilot whales was generated.

An abundance estimate of 28,218 (CV=0.36) long-finned pilot whales from the Newfoundland/Labrador region was generated from an aerial survey conducted by the Department of Fisheries and Oceans, Canada (DFO). This survey covered Atlantic Canadian shelf and shelf break waters extending from the northern tip of Labrador to the U.S border off southern Nova Scotia in August and September of 2016 (Lawson and Gosselin 2018). A total of 29,123 km were flown over the Gulf of St. Lawrence/Bay of Fundy/Scotian Shelf stratum using two Cessna Skymaster 337s and 21,037 km were flown over the Newfoundland/Labrador stratum using a DeHavilland Twin Otter. The Newfoundland estimate was derived from the Twin Otter data using two-team mark-recapture multi-covariate distance sampling methods. An availability bias correction factor, which was based on the cetaceans' surface intervals, was also applied. The Gulf of St. Lawrence/Bay of Fundy/Scotian Shelf survey detected 10 pilot whale groups, however, no abundance estimate was produced.

Spatial Distribution and Abundance Estimates for *Globicephala melas*

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

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

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

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

Minimum Population Estimate

The minimum population estimate is the lower limit of the two-tailed 60% confidence interval of the log-normally distributed best abundance estimate. This is equivalent to the 20th percentile of the log-normal distribution as specified by Wade and Angliss (1997). The best estimate of abundance for western North Atlantic long-finned pilot whales is 39,215 animals (CV=0.30). The minimum population estimate for long-finned pilot whales is 30,627.

Current Population Trend

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

CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

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

POTENTIAL BIOLOGICAL REMOVAL

Potential Biological Removal (PBR) is the product of minimum population size, one-half the maximum productivity rate, and a "recovery" factor (MMPA Sec. 3. 16 U.S.C. 1362; Wade and Angliss 1997). The minimum population size for long-finned pilot whales is 30,627. The maximum productivity rate is 0.04, the default value for cetaceans. The "recovery" factor is 0.5 because this stock is of unknown status relative to optimum sustainable population (OSP) and the CV of the average mortality estimate is less than 0.3 (Wade and Angliss 1997). PBR for the western North Atlantic long-finned pilot whale is 306 (Table 2).

Table 2. Best and minimum abundance estimates for western North Atlantic long-finned pilot whale (Globicephala melas melas) with Maximum Productivity Rate (R_{max}), Recovery Factor (F_r) and PBR.

N_{est}	CV	N_{min}	F_r	R_{max}	PBR
39,215	0.30	30,627	0.5	0.04	306

ANNUAL HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

Total annual observed average fishery-related mortality or serious injury during 2015–2019 was 9.0 long-finned pilot whales (CV=0.4; Table 3). In bottom trawls and mid-water trawls and in the gillnet fisheries, mortalities were more generally observed north of 40°N latitude and in areas expected to have only long-finned pilot whales. Takes in these fisheries were therefore attributed to the long-finned pilot whales. Takes in the pelagic longline fishery were partitioned according to a logistic regression model (Garrison and Rosel 2017).

Fishery Information

Detailed fishery information is reported in Appendix III.

Earlier Interactions

See Appendix V for more information on historical takes.

United States

Longline

During 2015–2019, pilot whale interactions (all serious injuries) were apportioned between the short-finned and long-finned pilot whale stocks according to a logistic regression model (Garrison and Rosel 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 Bottom Trawl

Fishery-related bycatch rates for years 2015–2019 were estimated using an annual stratified ratio-estimator (Lyssikatos and Chavez-Rosales 2022). 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 (Including Pair Trawl)

Three pilot whales were taken in the northeast mid-water trawl fishery in 2016. Using model-based predictions and at-sea identification, these takes have all been assigned as long-finned pilot whales. Expanded estimates of fishery mortality for 2015–2019 are not available, and so for those years the raw number is provided. 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

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

Table 3. Summary of the incidental mortality and serious injury of long-finned pilot whales (Globicephala melas melas) by U.S. commercial fisheries including the years sampled (Years), the type of data used (Data Type), the annual observer coverage coverage (Observer Coverage), the observed mortalities and serious injuries recorded by on-board observers, the estimated annual mortality and serious injury, the combined annual estimates of mortality and serious injury (Estimated Combined Mortality), the estimated CV of the combined estimates (Est. CVs) and the

mean of the combined estimates (CV in parentheses). These are minimum observed counts as expanded estimates are not available.

Fishery	Years	Data Type [a]	Observer Coverage [b]	Observed Serious Injury [c]	Observed Mortality	Estimated Serious Injury [e]	Estimated Mortality	Estimated Combined Mortality	Estimated CVs	Mean Combined Annual Mortality
Northeast Bottom Trawl	2015 2016 2017 2018 2019	Obs. Data, Logbook	0.19 0.12 0.12 0.16	0 0 0 0 0	0 4 0 0 1	0 0 0 0 0	0 29 0 0 5.39	0 29 0 0 5.39	na 0.58 na na 0.88	6.9 (0.51)
Northeast Mid-Water Trawl - Including Pair Trawl [c]	2015 2016 2017 2018 2019	Obs. Data, Dealer Data, VTR Data	0.08 0.27 0.16 0.14 0.28	0 0 0 0 0	0 3 0 0 0	0 0 0 0 0	0 3 0 0 0	0 3 0 0 0	na na na na na	0.6 (na)
Pelagic Longline Fishery	2015 2016 2017 2018 2019	Obs. Data, Logbook Data	0.12 0.15 0.12 0.10 0.10	1 1 1 1 1	0 0 0 0 0	2.2 1.1 3.3 0.4 0.4	0 0 0 0 0	2.2 1.1 3.3 0.4 0.4	0.49 1.0 0.98 0.93 1.0	1.5 (0.49)
TOTAL										9.0 (0.4)

a. Observer data (Obs. Data) are used to measure bycatch rates and the data are collected within the Northeast Fisheries Observer Program (NEFOP). NEFSC collects landings data (unallocated Dealer Data and Allocated Dealer Data) which are used as a measure of total landings. Mandatory Vessel Trip Reports (VTR; Trip Logbook) are used to determine the spatial distribution of landings and fishing effort. Total landings are used as a measure of total effort for the coastal gillnet fishery.
b. The observer coverages for the northeast sink gillnet fishery are ratios based on tons of fish landed. Northeast bottom trawl and northeast mid-water trawl fishery coverages are ratios based on trips.
c. Expanded estimates are not available for this fishery.
d. Serious injuries were evaluated for the period and include both at-sea monitor and traditional observer data (Josephson *et al*. 2022).

Other Mortality

Pilot whales have a propensity to mass strand throughout their range, but the role of human activity in these events is unknown. From 2015 to 2019, 7 long-finned pilot whales (*Globicephala melas melas*) were reported stranded between Maine and Florida, including the EEZ (Table 4; NOAA National Marine Mammal Health and Stranding Response Database, accessed 17 November 2020). None of the animals had indications of human interaction.

Table 4. Pilot whale (Globicephala melas melas) strandings along the Atlantic coast, 2015–2019. The level of technical expertise among stranding network personnel varies, and given the potential difficulty in correctly identifying stranded pilot whales to species, reports to specific species should be viewed with caution.

State	2015	2016	2017	2018	2019	Total
Nova Scotia [a]	21	12	12	3	2	50
Newfoundland and Labrador [b]	0	0	1	0	2	3
Maine [c]	0	1	1	3	0	5
Massachusetts	0	1	1	0	0	2
Total U.S.	0	2	2	3	0	7

a. Data supplied by Nova Scotia Marine Animal Response Society (pers. comm.).
b. See Ledwell and Huntington 2015, 2017, 2018, 2019, 2020.
c. 2016 animal released alive.

Stranding data probably underestimate the extent of human and fishery-related mortality and serious injury, particularly for offshore species such as pilot whales, because not all of the whales that die or are seriously injured in human interactions wash ashore, or, if they do, they are not all recovered (Peltier *et al*. 2012; Wells *et al*. 2015).

Additionally, not all carcasses will show evidence of human interaction, entanglement or other fishery-related interaction due to decomposition, scavenger damage, etc. (Byrd *et al*. 2014). Finally, the level of technical expertise among stranding network personnel varies widely as does the ability to recognize signs of human interaction.

HABITAT ISSUES

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

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

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

STATUS OF STOCK

The long-finned pilot whale is not listed as threatened or endangered under the Endangered Species Act, and the western North Atlantic stock is not considered strategic under the MMPA because the mean annual human-caused mortality and serious injury does not exceed PBR. Total U.S. fishery-related mortality and serious injury for long-finned pilot whales is less than 10% of the calculated PBR and, therefore, can be considered to be insignificant and approaching zero mortality and serious injury rate. The status of this stock relative to OSP in the U.S. Atlantic EEZ is unknown. A population trend analysis for this stock has not been conducted.

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

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

STOCK DEFINITION AND GEOGRAPHIC RANGE

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

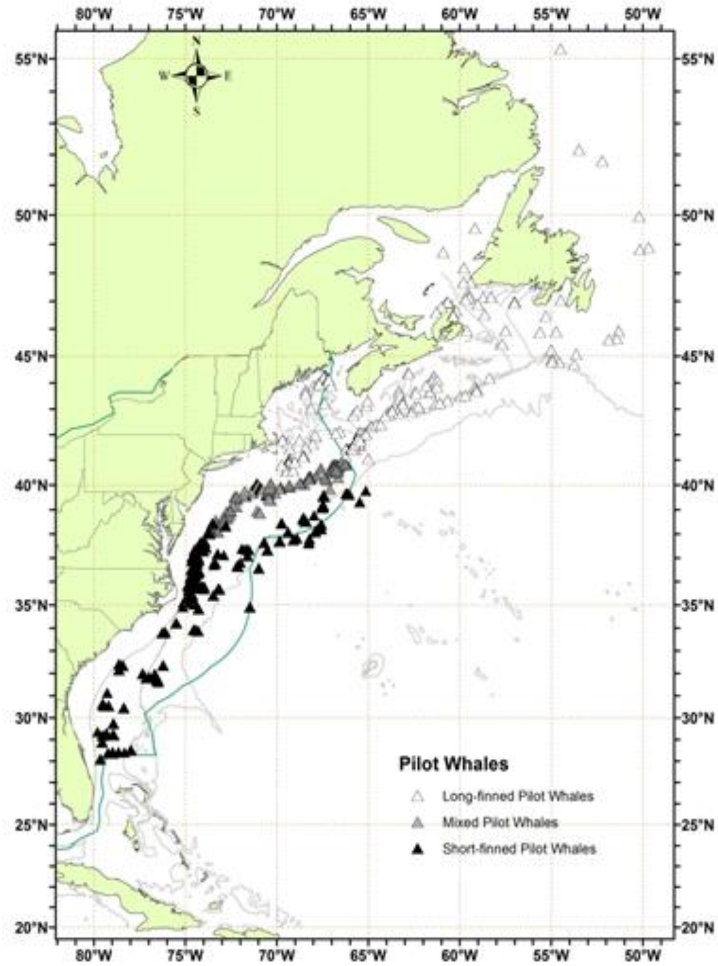

Figure 1. Distribution of long-finned (open symbols), short-finned (black symbols), and possibly mixed (gray symbols; could be either species) pilot whale sightings from NEFSC and SEFSC shipboard and aerial surveys during 1995, 1998, 1999, 2002, 2004, 2006, 2007, 2008, 2010, 2011 and 2016, and DFO's 2007 TNASS and 2016 NAISS surveys. The inferred distribution of the two species is preliminary and is valid for June–August only. Isobaths are the 200-m, 1000-m and 4000-m depth contours. The green line indicates the U.S. EEZ.

Thorne *et al.* (2017) tracked 33 short-finned pilot whales off Cape Hatteras in 2014 and 2015 using satellite-linked telemetry tags. Kernel density estimates of habitat use by whales during tracking were concentrated along the continental shelf break from Cape Hatteras north to Hudson Canyon, but whale distribution also included shelf break waters south of Cape Lookout, shelf break waters off Nantucket Shoals, and deeper offshore waters of the Gulf Stream east and north of Cape Hatteras, reinforcing that the continental shelf break is an important foraging habitat for short-finned pilot whales in the western North Atlantic. Finally, short-finned pilot whales that have stranded alive along the U.S. Atlantic coast and subsequently were released and tracked via satellite telemetry have travelled hundreds of kilometers from their release sites to other areas of the U.S. Atlantic and to the Caribbean (e.g., Irvine *et al.* 1979; Wells *et al.* 2013). Whether these movements are representative of normal species' patterns is unknown because they were generated from stranded animals.

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

POPULATION SIZE

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

Earlier Abundance Estimates

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

Recent Surveys and Abundance Estimates for *Globicephala* sp.

Abundance estimates of 3,810 (CV=0.42) and 25,114 (CV=0.27) *Globicephala* sp. were generated from vessel surveys conducted in the northeast and southeast U.S., respectively, during the summer of 2016. The northeast survey was conducted during 27 June – 25 August and consisted of 5,354 km of on-effort trackline. The majority of the survey was conducted in waters north of 38ºN latitude and included trackline along the shelf break and offshore to the U.S. EEZ. Pilot whale sightings were concentrated along the shelf-break between the 1,000-m and 2,000-m isobaths and along Georges Bank (NMFS 2017). The southeast vessel survey covered waters from Central Florida to approximately 38ºN latitude between the 100-m isobaths and the U.S. EEZ during 30 June – 19 August. A total of 4,399 km of trackline was covered on effort. Pilot whales were observed in high densities along the shelf-break between Cape Hatteras and New Jersey and also in waters further offshore in the mid-Atlantic and off the coast of Florida (NMFS 2017; Garrison and Palka 2018). Both the northeast and southeast surveys utilized two visual teams and an independent observer approach to estimate detection probability on the trackline (Laake and Borchers 2004). Mark-recapture distance sampling was used to estimate abundance. A logistic regression model (see next section) was used to estimate the abundance of short-finned pilot whales from these surveys. For the northeast survey, this resulted in an abundance estimate of 3,810 (CV=0.42) short-finned pilot whales. In the southeast, the model indicated that this survey included habitats expected to exclusively contain short-finned pilot whales resulting in an abundance estimate of 25,114 (CV=0.27).

Spatial Distribution and Abundance Estimates for *Globicephala macrorhynchus*

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

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

Month/Year	Area	N_{est}	CV
Jun–Aug 2016	New Jersey to lower Bay of Fundy	3,810	0.42
Jun–Aug 2016	Central Florida to New Jersey	25,114	0.27
Jun–Aug 2016	Central Florida to lower Bay of Fundy (COMBINED)	28,924	0.24

Minimum Population Estimate

The minimum population estimate is the lower limit of the two-tailed 60% confidence interval of the log-normally distributed best abundance estimate. This is equivalent to the 20th percentile of the log-normal distribution as specified by Wade and Angliss (1997). The best estimate of abundance for western North Atlantic short-finned pilot whale is 28,924 animals (CV=0.24). The minimum population estimate is 23,637 (Table 2).

Current Population Trend

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

CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

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

POTENTIAL BIOLOGICAL REMOVAL

Potential Biological Removal (PBR) is the product of minimum population size, one-half the maximum productivity rate, and a “recovery” factor (MMPA Sec. 3. 16 U.S.C. 1362; Wade and Angliss 1997). The minimum population size for short-finned pilot whales is 23,637. The maximum productivity rate is 0.04, the default value for cetaceans. The "recovery" factor is 0.5 because the stock's status relative to optimum sustainable population (OSP) is unknown and the CV of the average mortality estimate is less than 0.3 (Wade and Angliss 1997). PBR for the western North Atlantic short-finned pilot whale is 236 (Table 2).

Table 2. Best and minimum abundance estimates for the Western North Atlantic short-finned pilot whale with Maximum Productivity Rate (R_{max}), Recovery Factor (F_r) and PBR.

N_{est}	N_{est} CV	N_{min}	F_r	R_{max}	PBR
28,924	0.24	23,637	0.5	0.04	236

ANNUAL HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

The estimated mean annual fishery-related mortality and serious injury during 2015–2019 due to the large pelagics longline fishery was 136 short-finned pilot whales (CV=0.14; Table 3). Uncertainty in this estimate arises because it incorporates a logistic regression model to predict the species of origin (long-finned or short-finned pilot whale) for each bycaught whale. The statistical uncertainty in the assignment to species is incorporated into the abundance estimates; however, the analysis assumes that the collected biopsy samples adequately represent the distribution of the two species and that the resulting model correctly predicts shifts in distribution in response to changes in environmental conditions.

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

Fishery Information

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

Earlier Interactions

See Appendix V for information on historical takes.

Pelagic Longline

The Atlantic Ocean, Caribbean, Gulf of Mexico large pelagics longline fishery operates in the U.S. Atlantic (including Caribbean) and Gulf of Mexico EEZ, and pelagic swordfish, tunas and billfish are the target species. The estimated annual average serious injury and mortality attributable to the Atlantic Ocean large pelagics longline fishery for the five-year period from 2015 to 2019 was 136 short-finned pilot whales (CV=0.14; Table 3). During 2015–2019, 77 serious injuries were observed in the following fishing areas of the North Atlantic: Florida East Coast, Mid-Atlantic Bight, Northeast Coastal, and South Atlantic Bight. During 2015–2019, one mortality was observed (in 2016) in the Florida East Coast fishing area (Garrison and Stokes 2017; 2019; 2020a; 2020b; 2021).

Prior to 2014, estimated bycatch in the pelagic longline fishery was assigned to the short-finned pilot whale stock because the observed interactions all occurred at times and locations where available data indicated that long-finned pilot whales were very unlikely to occur. Specifically, the highest bycatch rates of undifferentiated pilot whales were observed during September–November along the mid-Atlantic coast (south of 38°N; Garrison 2007), and biopsy data collected in this area during October–November 2011 indicated that only short-finned pilot whales occurred in this region (Garrison and Rosel 2017). Similarly, all genetic data collected from interactions in the pelagic longline fishery have indicated interactions with short-finned pilot whales. However, in recent years, pilot whale interactions (including serious injuries) were observed farther north and along the southern flank of Georges Bank. Therefore, the logistic regression model (described above in 'Spatial Distribution and Abundance Estimates for *Globicephala macrorhynchus*) was applied using contemporaneous sea surface temperature data to estimate the probability that these interactions were from short-finned vs. long-finned pilot whales (Garrison and Rosel 2017). Due to high water temperatures (ranging from 22 to 25ºC) at the time of the observed takes, these interactions were estimated to have a >90% probability of coming from short-finned pilot whales. The estimated probability was used to apportion the estimated mortality and serious injury in the pelagic longline fishery between the short-finned and long-finned pilot whale stocks (Garrison and Stokes 2016; 2017; 2019; 2020a; 2020b; 2021).

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

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

See Table 3 for bycatch estimates and observed mortality and serious injury for the current five-year period, and Appendix V for historical estimates of annual mortality and serious injury.

Table 3. Summary of the incidental mortality and serious injury of short-finned pilot whales (Globicephala macrorhynchus) by the pelagic longline commercial fishery including the years sampled (Years), the number of vessels active within the fishery (Vessels), the type of data used (Data Type), the annual observer coverage (Observer Coverage), the annual observed serious injury and mortality recorded by on-board observers, the annual estimated serious injury and mortality, the combined annual estimates of serious injury and mortality (Estimated Combined Mortality), the estimated CV of the combined annual mortality estimates (Est. CVs) and the mean of the combined mortality estimates (CV in parentheses).

Fishery	Years	Vessels[a]	Data Type[b]	Percent Observer Coverage[c]	Observed Serious Injury	Observed Mortality	Estimated Serious Injury	Estimated Mortality	Estimated Combined Mortality	Est. CVs	Mean Annual Mortality
Pelagic Longline	2015	74	Obs. Data, Logbook	12	32	0	200	0	200	0.24	136 (0.14)
	2016	60		15	14	1	106	5.1	111	0.31	
	2017	65		12	14	0	133	0	133	0.29	
	2018	57		10	7	0	102	0	102	0.39	
	2019	50		10	10	0	131	0	131	0.37	

a. Number of vessels in the fishery is based on vessels reporting effort to the pelagic longline logbook.
b. Observer data (Obs. Data) are used to measure bycatch rates and the data are collected within the Northeast Fisheries Observer Program (NEFOP) and the Southeast Pelagic Longline Observer Program.
c. Percentage of sets observed

Hook and Line

During 2015–2019, there were no documented takes by this fishery. The most recent take occurred in 2013. 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

Pilot whales have a propensity to mass strand throughout their range, but the role of human activity in these events is unknown. Between two and 168 pilot whales have stranded annually, either individually or in groups, along the eastern U.S. seaboard since 1980 (NMFS 1993; NOAA National Marine Mammal Health and Stranding Response Database unpublished data). During 2015–2019, 47 short-finned pilot whales were reported stranded between Massachusetts and Florida (Table 4; Northeast Regional Marine Mammal Stranding Network; Southeast Regional Marine Mammal Stranding Network; NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 25 August 2020 (SER) and 23 July 2020 (NER)). These strandings included two mass stranding events of live animals in 2019. Evidence of human interaction was detected for two animals (one animal pushed out to sea by the public and one with ingested plastic debris; neither interaction was believed to be the cause of the stranding). No evidence of human interaction was detected for 15 strandings, and for the remaining 30 strandings, it could not be determined if there was evidence of human interaction. It should be noted that evidence of human interaction does not necessarily mean the interaction caused the animal's stranding or death.

Table 4. Short-finned pilot whale (Globicephala macrorhynchus) strandings along the Atlantic coast, 2015–2019. Data are from the NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 25 August 2020 (SER) and 23 July 2020 (NER). EEZ=U.S. Exclusive Economic Zone (offshore U.S. waters).

State	2015	2016	2017	2018	2019	TOTALS
Massachusetts	0	0	0	0	3[a]	3
New York	0	0	0	4	0	4
Maryland	0	0	0	0	1	1
Virginia	0	0	0	0	1	1
North Carolina	2	0	1	2	2	7
South Carolina	0	0	0	0	5	5
Georgia	1	0	1	0	21[b]	23
Florida	2	0	0	1	0	3
TOTALS	5	0	2	7	33	47

a. These 3 animals were a live mass stranding event.
b. These 21 animals were part of a mass stranding event of ~50 live whales.

There are a number of difficulties associated with the interpretation of stranding data. Stranding data underestimate the extent of human and fishery-related mortality and serious injury, particularly for offshore species such as pilot whales, because not all of the whales that die or are seriously injured in human interactions wash ashore, or, if they do, they are not all recovered (Peltier *et al*. 2012; Wells *et al*. 2015; Carretta *et al.* 2016). In particular, shelf and slope stocks in the western North Atlantic are less likely to strand than nearshore coastal stocks. Additionally, not all carcasses will show evidence of human interaction, entanglement or other fishery-related interaction due to decomposition, scavenger damage, etc. (Byrd *et al*. 2014). Finally, the level of technical expertise among stranding network personnel varies widely as does the ability to recognize signs of human interaction.

HABITAT ISSUES

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

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

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

STATUS OF STOCK

The short-finned pilot whale is not listed as threatened or endangered under the Endangered Species Act, and the western North Atlantic stock is not a strategic stock under the MMPA because the mean annual human-caused mortality and serious injury does not exceed PBR. The status of this stock relative to OSP in the U.S. Atlantic EEZ is unknown. Total U.S. fishery-related mortality and serious injury attributed to short-finned pilot whales exceeds 10% of the calculated PBR and therefore cannot be considered to be insignificant and approaching zero mortality and serious injury rate. There is no evidence for a trend in population size for this stock.

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

STOCK DEFINITION AND GEOGRAPHIC RANGE

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

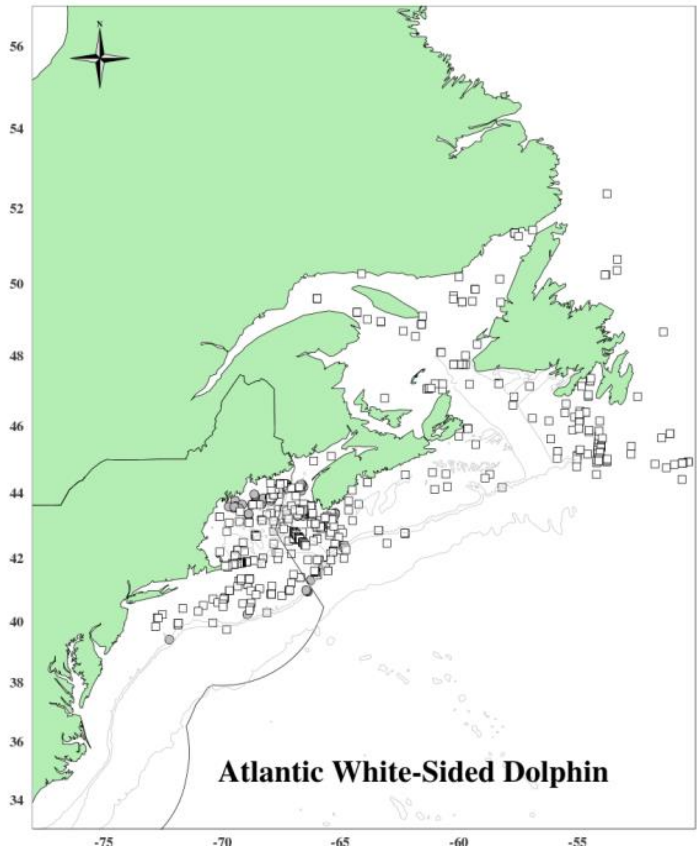

Figure 1. Distribution of white-sided dolphin sightings from NEFSC and SEFSC shipboard and aerial surveys during the summers of 1995, 1998, 1999, 2002, 2004, 2006, 2007, 2008, 2010, 2011, 2016 and Department of Fisheries and Oceans Canada 2007 TNASS and 2016 NAISS surveys. Isobaths are the 200-m, 1000-m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

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

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

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

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

POPULATION SIZE

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

Earlier Abundance Estimates

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

Recent Surveys and Abundance Estimates

An abundance estimate of 31,912 (CV=0.61) U.S. Gulf of Maine white-sided dolphins was generated from a shipboard and aerial survey conducted during 27 June–28 September 2016 (Palka 2020) in a region covering 425,192 km^2 (Table 1). The aerial portion included 11,782 km of tracklines that were over waters north of New Jersey from the coastline to the 100-m depth contour, throughout the U.S. waters. The shipboard portion included 4,351 km of tracklines that were in waters offshore of central Virginia to Massachusetts (waters that were deeper than the 100-m depth contour out to beyond the U.S. EEZ). Both sighting platforms used a two-team data-collection procedure, which allows estimation of abundance to correct for perception bias of the detected species (Laake and Borchers 2004). The estimates were also corrected for availability bias.

An abundance estimate of 61,321 (CV=1.04) white-sided dolphins from the Canadian side of the Gulf of Maine population and the entire Gulf of St. Lawrence population was generated from an aerial survey conducted by the Department of Fisheries and Oceans, Canada (DFO, Table 1). No white-sided dolphins were detected on the east side of Labrador in the Labrador population. This survey covered Atlantic Canadian shelf and shelf break waters extending from the northern tip of Labrador to the U.S. border off southern Nova Scotia in August and September of 2016 (Lawson and Gosselin 2018). A total of 29,123 km was flown over the Gulf of St. Lawrence/Bay of Fundy/Scotian Shelf stratum using two Cessna Skymaster 337s, and 21,037 km were flown over the Newfound/Labrador stratum using a DeHavilland Twin Otter. The estimate was derived from the Skymaster data using single-team multi-covariate distance sampling with left truncation (to accommodate the obscured area under the plane) where size-bias was also investigated. The Otter-based perception bias correction, which used double-platform mark-recapture methods, was applied. An availability bias correction factor, which was based on the cetaceans' surface intervals, was also applied.

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

Month/Year	Area	N_{est}	CV
Jun–Sep 2016	Central Virginia to Maine (US part of Gulf of Maine population)	31,912	0.61
Aug–Sep 2016	Bay of Fundy to Gulf of St. Lawrence (Canadian part of Gulf of Maine and all of Gulf of St. Lawrence population)	61,321	1.04
Aug–Sep 2016	Newfoundland and Labrador (part of the Labrador population)	0	0
Jun–Sep 2016	Central Virginia to Labrador – COMBINED	93,233	0.71

Minimum Population Estimate

The minimum population estimate is the lower limit of the two-tailed 60% confidence interval of the log-normally distributed best abundance estimate. This is equivalent to the 20th percentile of the log-normal distribution as specified by (Wade and Angliss 1997). The best estimate of abundance for the western North Atlantic stock of white-sided dolphins is 93,233 (CV=0.71). The minimum population estimate for these white-sided dolphins is 54,443.

Current Population Trend

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

CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

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

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

POTENTIAL BIOLOGICAL REMOVAL

Potential Biological Removal (PBR) is the product of minimum population size, one-half the maximum productivity rate, and a recovery factor (MMPA Sec. 3. 16 U.S.C. 1362; Wade and Angliss 1997). The minimum population size is 54,443. The maximum productivity rate is 0.04, the default value for cetaceans. The recovery factor is 0.5, the default value for stocks of unknown status relative to Optimum Sustainable Population (OSP), and the CV of the average mortality estimate is less than 0.3 (Wade and Angliss 1997). PBR for the western North Atlantic stock of white-sided dolphin is 544 (Table 2).

Table 2. Best and minimum abundance estimates for the western North Atlantic stock of white-sided dolphins (Lagenorhynchus acutus), with Maximum Productivity Rate (R_{max}), Recovery Factor (F_r) and PBR.

N_{est}	CV	N_{min}	F_r	R_{max}	PBR
93,233	0.71	54,443	0.5	0.04	544

ANNUAL HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

Total annual estimated average fishery-related mortality or serious injury to this stock during 2015–2019 was 27 (CV=0.21) white-sided dolphins from fisheries observer data and 0.2 from non-fishery stranding data (Table 3).

Table 3. Total annual estimated average human-caused mortality and serious injury for the North Atlantic stock of white-sided dolphins (Lagenorhynchus acutus).

Years	Source	Annual Avg.	CV
2015–2019	U.S. fisheries using observer data	27	0.21
2015–2019	Possible non-fishery human-caused stranding mortalities	0.2	
TOTAL		27.2	0.21

Key uncertainties include the potential that the observer coverage in the Mid-Atlantic gillnet may not be representative of the fishery during all times and places, since the observer coverage was relatively low in some times and areas (0.02–0.10). The effect of this is unknown.

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

Fishery Information

Detailed fishery information is reported in Appendix III.

Earlier Interactions

See Appendix V for more information on historical takes.

United States

Northeast Bottom Trawl

White-sided dolphins have been bycaught year-round in the Gulf of Maine, where most occurred outside of summer (May–August) and offshore near the outer edge of the EEZ. Fishery-related bycatch rates were estimated using an annual stratified ratio-estimator (Lyssikatos and Chavez-Rosales 2022). See Table 4 for bycatch estimates and observed mortality and serious injury for the current 5-year period, and Appendix V for long-term bycatch information.

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

Fishery	Years	Data Type [a]	Observer Coverage [b]	Observed Serious Injury[c]	Observed Mortality	Estimated Serious Injury	Estimated Mortality	Estimated Combined Mortality	Estimated CVs	Mean Combined Annual Mortality
Northeast Bottom Trawl	2015	Obs. Data, Trip Logbook	0.19	0	3	0	15	15	0.52	27 (0.21)
	2016		0.12	0	3	0	28	28	0.46	
	2017		0.12	1	1	7.4	7.4	14.8	0.64	
	2018		0.12	0	0	0	0	0	na	
	2019		0.16	0	14	0	79	79	0.28	
TOTAL										27 (0.21)

a. Observer data (Obs. Data), used to measure bycatch rates, are collected within the Northeast Observer Program and At-sea Monitoring Program. NEFSC collects landings data (unallocated Dealer Data or Allocated Dealer Data) which are used as a measure of total landings. Mandatory Vessel Trip Reports (VTR; Trip Logbook) are used to determine the spatial distribution of landings and fishing effort in the sink gillnet, bottom trawl and mid-water trawl fisheries. In addition, the Trip Logbooks are the primary source of the measure of total effort (tow duration) in the mid-water and bottom trawl fisheries.
b. Observer coverage is defined as the ratio of observed to total metric tons of fish landed for the gillnet fisheries, and the ratio of observed to total trips for bottom trawl and Mid-Atlantic mid-water trawl (including pair trawl) fisheries. Total observer coverage reported for bottom trawl and gillnet gear includes samples collected from the at-sea monitoring program in addition to traditional observer coverage through the Northeast Fisheries Observer Program (NEFOP).
c. Serious injuries were evaluated for the 2015–2019 period and include both at-sea monitor and traditional observer data (Josephson *et al.* 2022).

Canada

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

Hooker *et al.* (1997) summarized bycatch data from a Canadian fisheries observer program that placed observers on all foreign fishing vessels operating in Canadian waters, on 25–40% of large Canadian fishing vessels (greater than 100 feet long), and on approximately 5% of smaller Canadian fishing vessels. Bycaught marine mammals were noted as weight in kilos rather than by the numbers of animals caught. Thus the number of individuals was estimated by dividing the total weight per species per trip by the maximum recorded weight of each species. During 1991 through 1996, an estimated 6 white-sided dolphins were observed taken. One animal was from a longline trip south of the Grand Banks (43º 10'N 53º 08'W) in November 1996 and the other 5 were taken in the bottom trawl fishery off Nova Scotia in the Atlantic Ocean; 1 in July 1991, 1 in April 1992, 1 in May 1992, 1 in April 1993, 1 in June 1993 and 0 in 1994 to 1996.

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

Other Mortality

United States

Recent Atlantic white-sided dolphin strandings on the U.S. Atlantic coast are documented in Table 5 (NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 17 October 2020). Sixteen of these animals were released alive. Human Interaction (HI) was indicated in 4 records during this period, though in only one of these was the HI a possible contributor to the mortality (signs of an entanglement wound). 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.

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, including post-stranding. Stranding data probably underestimate the extent of mortality and serious injury because all of the marine mammals that die or are seriously injured may not wash ashore, nor will all of those that do wash ashore necessarily show signs of entanglement or other fishery-interaction. Finally, the level of technical expertise among stranding network personnel varies widely as does the ability to recognize signs of fishery interaction.

Canada

The Nova Scotia Stranding Network documented whales and dolphins stranded on the coast of Nova Scotia during 1991 to 1996 (Hooker *et al.* 1997). Researchers with the 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 5; Ledwell and Huntington 2015, 2017, 2018, 2019, 2020).

Table 5. Atlantic white-sided dolphin (Lagenorhynchus acutus) reported strandings along the U.S. and Canadian Atlantic coast, 2015–2019.

Area	2015	2016	2017	2018	2019	Total
Maine[b]	1	0	0	6	5	12
New Hampshire	0	0	0	0	2	2
Massachusetts[a, b, c, d]	3	27	8	41	65	144
Connecticut	0	1	1	0	0	2
TOTAL US	4	28	9	47	72	160
Nova Scotia[e]	11	11	8	0	0	30
Newfoundland and Labrador[f]	0	13	1	0	0	14
TOTAL US & CANADA	15	38	38	47	72	204

a. Records of mass strandings in Massachusetts during this period are: March 2016 - 2 animals (1 released alive), July 2016 - 2 animals (1 released alive), 3 animals (all released alive); September 2016 - 17 animals (all released alive).
b. In 2016, 2 animals (one of which was released alive) in Massachusetts were classified as human interaction due to intervention on the beach.
c. In 2018, 1 white-sided dolphin mortality had signs of human interaction indicated due to entanglement wounds found on tailstock and beach-protection mesh wrapped on torso.
d. In 2019, 2 white-sided dolphin mortalities had signs of human interaction indicated, although neither of these likely contributed to mortality. One was coded as HI due to public attempts to refloat, and the other due to tag applied by standing responders.
e. Data supplied by Nova Scotia Marine Animal Response Society (pers. comm.). 2015 data include a mass stranding of 5 animals.
f. Ledwell and Huntington (2015, 2017, 2018, 2019, 2020).

HABITAT ISSUES

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

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

STATUS OF STOCK

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

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

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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 exhibit seasonal movements, where they are found from Cape Hatteras northeast to Georges Bank (35˚ to 42˚N) during mid-January to May (Hain *et al.* 1981; CETAP 1982; Payne *et al.* 1984), although some animals tagged and released after stranding in winters of 2010–2012 used habitat in the Gulf of Maine north to almost 44˚N (Sharp *et al*. 2016). Common dolphins move onto Georges Bank, Gulf of Maine, and the Scotian Shelf from mid-summer to autumn. Selzer and Payne (1988) reported very large aggregations (greater than 3,000 animals) on Georges Bank in autumn. Migration onto the Scotian Shelf and continental shelf off Newfoundland occurs during summer and autumn when water temperatures exceed 11ºC (Sergeant *et al.* 1970; Gowans and Whitehead 1995).

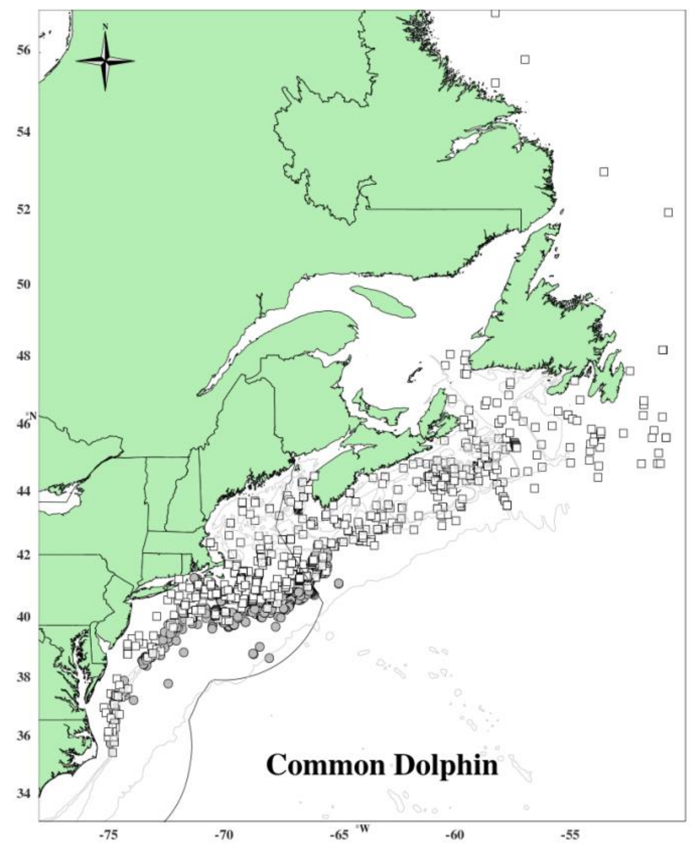

Figure 1. Distribution of common dolphin sightings from NEFSC and SEFSC shipboard and aerial surveys during the ummers of 1998, 1999, 2002, 2004, 2006, 2007, 2010, 2011, 2016 and Department of Fisheries and Oceans Canada 2007 TNASS and 2016 NAISS surveys. Isobaths are the 100-m, 1000-m and 4000-m depth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

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

POPULATION SIZE

The current best abundance estimate for Western North Atlantic stock of common dolphins is 172,947 (CV=0.21) which is the total of Canadian and U.S. surveys conducted in 2016 (Table 1). This estimate, derived from shipboard and aerial surveys, covers most of this stock's known range. Because the survey areas did not overlap, the estimates from the three surveys were added together and the CVs pooled using a delta method to produce a species abundance estimate for the stock area.

Earlier Abundance Estimates

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

Recent Surveys and Abundance Estimates

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

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

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

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

Minimum Population Estimate

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

Current Population Trend

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

CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

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

POTENTIAL BIOLOGICAL REMOVAL

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

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

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

ANNUAL HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

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

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

Years	Source	Annual Avg.	CV
2015–2019	U.S. fisheries using observer data	390	0.11
2015–2019	Research mortalities	0.2	
2015–2019	Non-fishery stranding mortalities	0.2	
TOTAL		390.4	

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

Northeast Sink Gillnet

Annual common dolphin mortalities were estimated using annual ratio-estimator methods (Orphanides and Hatch 2017; Orphanides 2019, 2020, 2021; Precoda and Orphanides 2022). See Table 4 for bycatch estimates and observed mortality and serious injury for the current 5-year period, and Appendix V for historical bycatch information.

Mid-Atlantic Gillnet

Common dolphins were taken in observed trips during most years. Annual common dolphin mortalities were estimated using annual ratio-estimator methods (Orphanides and Hatch 2017; Orphanides 2019, 2020, 2021; Precoda and Orphanides 2022). See Table 4 for bycatch estimates and observed mortality and serious injury for the current 5-year period, and Appendix V for historical bycatch information.

Northeast Bottom Trawl

This fishery is active in New England waters in all seasons. Annual common dolphin mortalities were estimated using annual stratified ratio-estimator methods (Lyssikatos and Chavez-Rosales 2022). See Table 4 for bycatch estimates and observed mortality and serious injury for the current 5-year period, and Appendix V for historical bycatch information.

Mid-Atlantic Bottom Trawl

Annual common dolphin mortalities were estimated using annual stratified ratio-estimator methods (Lyssikatos and Chavez-Rosales 2022). See Table 4 for bycatch estimates and observed mortality and serious injury for the current 5-year period, and Appendix V for historical bycatch information.

Pelagic Longline

Pelagic longline bycatch estimates of common dolphins for 2015–2019 were documented in Garrison and Stokes (2017, 2020a, 2020b, 2021). There is a high likelihood that dolphins released alive with ingested gear or gear wrapped around appendages will not survive (Wells *et al.* 2008). See Table 4 for bycatch estimates and observed mortality and serious injury for the current 5-year period, and Appendix V for historical bycatch information.

Research Takes

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

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

Fishery	Years	Data Type [a]	Observer Coverage [b]	Observed Serious Injury[d]	Observed Mortality	Estimated Serious Injury[d]	Estimated Mortality	Estimated Combined Mortality	Estimated CVs	Mean Combined Annual Mortality
Northeast Sink Gillnet	2015 2016 2017 2018 2019	Obs. Data, Trip Logbook, Allocated Dealer Data	0.14 0.10 0.12 0.11 0.13	0 0 0 0 0	3 8 20 10 1	0 0 0 0 0	55 80 133 93 5.0	55 80 133 93 5.0	0.54 0.38 0.28 0.45 0.68	73 (0.19)
Mid-Atlantic Gillnet	2015 2016 2017 2018 2019	Obs. Data, Weighout	0.06 0.08 0.09 0.09 0.13	0 0 1 0 0	3 1 1 1 3	0 0 11 1 0	30 7 11 7.7 20	30 7 22 7.7 20	0.55 0.97 0.71 0.91 0.56	17 (0.31)
Northeast Bottom Trawl [c]	2015 2016 2017 2018 2019	Obs. Data, Logbook	0.19 0.12 0.16 0.12 0.16	0 0 0 0 0	4 2 0 4 2	0 0 0 0 0	22 16 0 28 10	22 16 0 28 10	0.45 0.46 0 0.54 0.62	15 (0.27)
Mid-Atlantic	2015 2016	Obs. Data,	0.09 0.10	0 0	26 22	0 0	250 177	250 177	0.32 0.33	281 (0.12)

Bottom Trawl [c]	2017	Dealer Data	0.10	0	66	0	380	380	0.23	
	2018		0.12	1	34	5	200	205	0.54	
	2019		0.12	2	52	15	395	395	0.23	
Pelagic Longline	2015	Obs. Data, Logbook Data	0.12	1	0	9.05	0	9.05	1	3.1 (0.67)
	2016		0.15	0	0	0	0	0	0	
	2017		0.12	1	0	4.92	0	4.92	1	
	2018		0.10	1	0	1.44	0	1.44	1	
	2019		0.10	0	0	0	0	0	0	
TOTAL										390 (0.11)

a. Observer data (Obs. Data), used to measure bycatch rates, are collected within the Northeast Fisheries Observer Program and At-sea Monitoring Program. NEFSC collects landings data (unallocated Dealer Data or Allocated Dealer Data) which are used as a measure of total landings and mandatory Vessel Trip Reports (VTR; Trip Logbook) are used to determine the spatial distribution of landings and fishing effort.
b. Observer coverage is defined as the ratio of observed to total metric tons of fish landed for the gillnet fisheries and the ratio of observed to total trips for bottom trawl and Mid-Atlantic mid-water trawl (including pair trawl) fisheries.
c. Fishery related bycatch rates were estimated using an annual stratified ratio-estimator (Lyssikatos and Chavez-Rosales 2022).
d. Serious injuries were evaluated for the period and include both at-sea monitor and traditional observer data (Josephson *et al.* 2022)

Other Mortality

Common dolphins reported stranded between Maine and Florida are reported in Table 5 (NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 17 November 2020). The total includes mass-stranded common dolphins in Massachusetts during 2015 (a total of 37 in 13 events), 2016 (a total of 35 animals in 9 events), 2017 (over 90 animals in 20 events), and 2018 (a total of 28 animals in 9 events) and 2019 (28 animals in 9 events). Animals released or last sighted alive include 9 in 2015, 23 in 2016, 70 in 2017, 18 in 2018 and 4 in 2019. In 2015, 2 cases were classified as human interactions, both in Rhode Island, and both related to mutilation likely to be post-mortem. Seven cases in 2016 were coded as human interaction. All but 2 of these were released alive. One of the 2 was a fishery interaction and the other was coded HI (Human Interaction) due to a beachgoer intervention. Six cases in 2017 were coded as human interaction, 2 of which were classified as fishery interactions, 1 classified as a possible boat collision, and 1 released alive. Another dolphin was euthanized after multiple restrandings and another was HI due to beachgoer intervention. In 2018, 5 cases were coded as human interactions. Two were public harassment and 3 involved fishing gear, though only one was classified as a fishery interaction. Eight stranding mortalities in Massachusetts in 2019 were classified as human interactions and one each in New York and Rhode Island. The New York case was a fishery interaction. All were either coded as unlikely or undetermined that the HI contributed to the stranding. In this 5-year period, only 1 interaction (boat strike in 2017) was likely a non-fishery human-caused mortality. 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 stranded in 2015, 5 in 2016, 5 in 2017, 5 in 2018, and 4 in 2019 (Tonya Wimmer/Andrew Reid, pers. comm.).

Table 5. Common dolphin (Delphinus delphis delphis) reported strandings along the U.S. Atlantic coast, 2015–2019.

STATE	2015	2016	2017	2018	2019	TOTALS
New Hampshire	1	1	2	0	0	4
Massachusetts[a]	40	67	166	61	95	429
Rhode Island[b]	7	4	5	4	5	25
Connecticut	2	1	1	0	0	4
New York	3	3	15	11	9	41
New Jersey	3	5	0	2	4	14
Delaware	2	0	0	0	1	3

Maryland	1	0	0	0	2	3
Virginia	2	0	1	3	5	11
North Carolina	4	1	0	3	4	12
TOTALS	65	82	190	84	125	546

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

HABITAT ISSUES

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

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

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

STATUS OF STOCK

Common dolphins are not listed as threatened or endangered under the Endangered Species Act, and the Western North Atlantic stock is not considered strategic under the Marine Mammal Protection Act. The 2015–2019 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 Optimum Sustainable Population (OSP), in the U.S. Atlantic EEZ 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, satellite telemetry data, passive acoustic monitoring, strandings and takes reported by NMFS observers in the Sea Sampling Programs. During summer (July to September), harbor porpoises are concentrated in the northern Gulf of Maine, southern Bay of Fundy and around the southern tip of Nova Scotia, generally in waters less than 150 m deep (Gaskin 1977; Kraus *et al*. 1983; Palka 1995), with lower densities in the upper Bay of Fundy and on Georges Bank (Palka 2000). During fall (October–December) and spring (April–June), harbor porpoises are widely dispersed from New Jersey to Maine, with lower densities farther north and south. During winter (January to March), intermediate densities of harbor porpoises can be found in waters off New Jersey to North Carolina, and lower densities are found in waters off New York to New Brunswick, Canada. In non-summer months they have been seen from the coastline to deep waters (>1800 m; Westgate *et al*. 1998), although the majority are found over the continental shelf. Passive acoustic monitoring detected harbor porpoises regularly during the period January–May offshore of Maryland (Wingfield *et al.* 2017). There does not appear to be a temporally coordinated migration or a specific migratory route to and from the Bay of Fundy region. However, during the fall, several satellite-tagged harbor porpoises did favor the waters around the 92-m isobath, which is consistent with observations of high rates of incidental catches in this depth range (Read and Westgate 1997). There were two stranding records from Florida during the 1980s (Smithsonian strandings database) and one in 2003 (NE Regional Office/NMFS strandings and entanglement database).

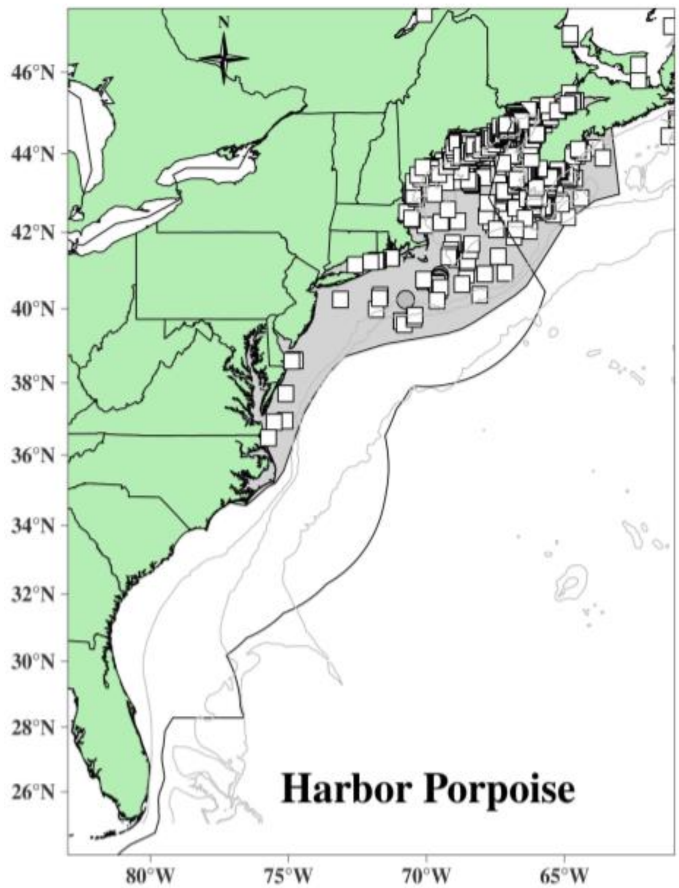

Figure 1. Distribution of harbor porpoises from NEFSC and SEFSC shipboard and aerial surveys during the summers of 1995, 1998, 1999, 2002, 2004, 2006, 2007, 2008, 2010, 2011 and 2016 and portions of DFO's 2007 TNASS and 2016 NAISS surveys. Isobaths are the 100m, 200m, 1000m, and 4000m depth contours.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

Gaskin (1984, 1992) proposed that there were four separate populations in the western North Atlantic: the Gulf of Maine/Bay of Fundy, Gulf of St. Lawrence, Newfoundland, and Greenland populations. Analyses involving mtDNA (Wang *et al*. 1996; Rosel *et al*. 1999a, 1999b), organochlorine contaminants (Westgate *et al*. 1997; Westgate and Tolley 1999), heavy metals (Johnston 1995), and life history parameters (Read and Hohn 1995) support Gaskin's proposal. Genetic studies using mitochondrial DNA (Rosel *et al*. 1999a) and contaminant studies using total PCBs

(Westgate and Tolley 1999) indicate that the Gulf of Maine/Bay of Fundy females were distinct from females from the other populations in the Northwest Atlantic. Gulf of Maine/Bay of Fundy males were distinct from Newfoundland and Greenland males, but not from Gulf of St. Lawrence males according to studies comparing mtDNA (Palka *et al*. 1996; Rosel *et al*. 1999a) and CHLORs, DDTs, PCBs and CHBs (Westgate and Tolley 1999). Nuclear microsatellite markers have also been applied to samples from these four populations, but this analysis failed to detect significant population sub-division in either sex (Rosel *et al*. 1999a). These patterns may be indicative of female philopatry coupled with dispersal of males. Both mitochondrial DNA and microsatellite analyses indicate that the Gulf of Maine/Bay of Fundy stock is not the sole contributor to the aggregation of porpoises found off the mid-Atlantic states during winter (Rosel *et al*. 1999a; Hiltunen 2006). Mixed-stock analyses using twelve microsatellite loci in both Bayesian and likelihood frameworks indicate that the Gulf of Maine/Bay of Fundy is the largest contributor (~60%), followed by Newfoundland (~25%) and then the Gulf of St. Lawrence (~12%), with Greenland making a small contribution (<3%). For Greenland, the lower confidence interval of the likelihood analysis includes zero. For the Bayesian analysis, the lower 2.5% posterior quantiles include zero for both Greenland and the Gulf of St. Lawrence. Intervals that reach zero provide the possibility that these populations contribute no animals to the mid-Atlantic aggregation.

This report follows Gaskin's hypothesis on harbor porpoise stock structure in the western North Atlantic, where the Gulf of Maine and Bay of Fundy harbor porpoises are recognized as a single management stock separate from harbor porpoise populations in the Gulf of St. Lawrence, Newfoundland, and Greenland. It is unlikely that the Gulf of Maine/Bay of Fundy harbor porpoise stock contains multiple demographically independent populations (Rosel *et al*. 1999a; Hiltunen 2006), but a comparison of samples from the Scotian shelf to the Gulf of Maine has not yet been made.

POPULATION SIZE

The best current abundance estimate of the Gulf of Maine/Bay of Fundy harbor porpoise stock is the sum of the 2016 NEFSC and Department of Fisheries and Oceans Canada (DFO) surveys: 95,543 (CV=0.31; Table 1). Because the survey areas did not overlap, the estimates from the two surveys were added together and the CVs pooled using a delta method to produce a species abundance estimate for the stock area. A key uncertainty in the population size estimate is the precision and accuracy of the availability bias correction factor that was applied. More information on the spatio-temporal variability of the animals' dive profile is needed.

Earlier Abundance Estimates

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

Recent Surveys and Abundance Estimates

An abundance estimate of 75,079 (CV=0.38) harbor porpoises was generated from a U.S. shipboard and aerial survey conducted during 27 June–28 September 2016 (Table 1; Palka 2020) in a region covering 425,192 km^2. The aerial portion included 11,782 km of tracklines that were over waters north of New Jersey from the coastline to the 100-m depth contour, throughout the U.S. waters. The shipboard portion included 4,351 km of tracklines that were in waters offshore of central Virginia to Massachusetts (waters that were deeper than the 100-m depth contour out to beyond the outer limit of the U.S. EEZ). Both sighting platforms used a two-team data collection procedure, which allows estimation of abundance to correct for perception bias of the detected species (Laake and Borchers 2004). The estimates were also corrected for availability bias.

An abundance estimate of 20,464 (CV=0.39) harbor porpoises from the Canadian Bay of Fundy/Scotian shelf region was generated from an aerial survey conducted by the Department of Fisheries and Oceans, Canada (DFO). The entire survey covered Atlantic Canadian shelf and shelf break waters extending from the northern tip of Labrador to the U.S border off southern Nova Scotia in August and September of 2016 (Lawson and Gosselin 2018). A total of 29,123 km were flown over the Gulf of St. Lawrence/Bay of Fundy/Scotian Shelf strata using two Cessna Skymaster 337s and 21,037 km were flown over the Newfound/Labrador strata using a DeHavilland Twin Otter. The harbor porpoise estimate was derived from the Skymaster data using single team multi-covariate distance sampling with left truncation (to accommodate the obscured area under the plane) where size-bias was also investigated. The Otter-based perception bias correction, which used double platform mark-recapture methods, was applied. An availability bias correction factor, which was based on published records of the cetaceans' surface intervals, was also applied.

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

Month/Year	Area	N_{est}	CV
Jun–Sep 2016	Central Virginia to Maine	75,079	0.38
Aug–Sep 2016	Gulf of St. Lawrence/Bay of Fundy/Scotian Shelf	20,464	0.39
Jun–Sep 2016	**Central Virginia to Gulf of St. Lawrence/Bay of Fundy/Scotian Shelf - COMBINED**	**95,543**	**0.31**

Minimum Population Estimate

The minimum population estimate is the lower limit of the two-tailed 60% confidence interval of the log-normal distributed best abundance estimate. This is equivalent to the 20th percentile of the log-normal distribution as specified by Wade and Angliss (1997). The best estimate of abundance for harbor porpoises is 95,543 (CV=0.31). The minimum population estimate for the Gulf of Maine/Bay of Fundy harbor porpoise is 74,034 (Table 2).

Current Population Trend

A trend analysis has not been conducted for this stock. The statistical power to detect a trend in abundance for this stock is poor due to the relatively imprecise abundance estimates and 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 74,034. The maximum productivity rate for this stock is 0.046. The recovery factor is 0.5 because stock's status relative to Optimum Sustainable Population (OSP) is unknown and the CV of the average mortality estimate is less than 0.3 (Wade and Angliss 1997). PBR for the Gulf of Maine/Bay of Fundy harbor porpoise is 851 (Table 2).

Table 2. Best and minimum abundance estimates for the Gulf of Maine/Bay of Fundy harbor porpoise (Phocoena phocoena phocoena) with Maximum Productivity Rate (R_{max}), Recovery Factor (F_r) and PBR.

N_{est}	CV	N_{min}	F_r	R_{max}	PBR
95,543	0.31	74,034	0.5	0.046	851

ANNUAL HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

The total annual estimated average human-caused mortality and serious injury is 163 harbor porpoises per year (CV=0.13) from U.S. fisheries using observer data and an annual average of 1.6 animals from non-fishery stranding records (Table 3). Canadian bycatch information is not available.

Table 3. Total annual estimated average human-caused mortality and serious injury for the Gulf of Maine/Bay of Fundy harbor porpoise (Phocoena phocoena phocoena).

Years	Source	Annual Avg.	CV
2015–2019	U.S. fisheries using observer data	163	0.13
2015–2019	Non-fishery human caused stranding mortalities	0.6	-
TOTAL		164	-

A key uncertainty is the potential that the observer coverage in the Mid-Atlantic gillnet fishery may not be representative of the fishery during all times and places, since the observer coverage was relatively low for some times and areas (0.02–0.10). The effect of this is unknown. Another key uncertainty is that mortalities and serious injuries in Canadian waters are largely unquantified. There are no major known sources of unquantifiable human-caused mortality or serious injury for the U.S. waters within the Gulf of Maine/Bay of Fundy harbor porpoise stock's habitat.

United States

Northeast Sink Gillnet

Harbor porpoise bycatch in the northern Gulf of Maine occurs primarily from June to September, while in the southern Gulf of Maine and south of New England, bycatch occurs from January to May and September to December. Annual bycatch is estimated using ratio estimator techniques that account for the use of pingers (Orphanides and Hatch 2017; Orphanides 2019, 2020, 2021; Precoda and Orphanides 2022). See Table 4 for bycatch estimates and observed mortality and serious injury for the current 5-year period, and Appendix V for historical bycatch information.

Mid-Atlantic Gillnet

Harbor porpoise bycatch in Mid-Atlantic waters occurs primarily from December to May in waters off New Jersey and less frequently in other waters ranging farther south, from New Jersey to North Carolina. Annual bycatch is estimated using ratio estimator techniques (Orphanides and Hatch 2017; Orphanides 2019, 2020, 2021; Precoda and Orphanides 2022). See Table 4 for bycatch estimates and observed mortality and serious injury for the current 5-year period, and Appendix V for historical bycatch information.

Northeast Bottom Trawl

Since 1989, harbor porpoise mortalities have been observed in the northeast bottom trawl fishery, but many of these were not attributable to this fishery because decomposed animals are presumed to have been dead prior to being taken by the trawl. Those infrequently caught freshly dead harbor porpoises have been caught during January to April on Georges Bank or in the southern Gulf of Maine. Fishery-related bycatch rates were estimated using an annual stratified ratio-estimator (Lyssikatos and Chavez-Rosales 2022). See Table 4 for bycatch estimates and observed mortality and serious injury for the current 5-year period, and Appendix V for historical bycatch information.

Canada

No current estimates exist, but harbor porpoise interactions have been documented in the Bay of Fundy sink gillnet fishery and in herring weirs between the years 1998–2001 in the lower Bay of Fundy demersal gillnet fishery (Trippel and Shepherd 2004). That fishery has declined since 2001 and it is assumed bycatch is very small, if any (H. Stone, Department of Fisheries and Oceans Canada, pers. comm.).

Table 4. From observer program data, summary of the incidental mortality of Gulf of Maine/Bay of Fundy harbor porpoise (Phocoena phocoena phocoena) by commercial fishery including the years sampled, the type of data used, the annual observer coverage, the mortalities and serious injuries recorded by on-board observers, the estimated

annual serious injury and mortality, the estimated CV of the annual mortality, and the mean annual combined mortality with its CV.

Fishery	Years	Data Type [a]	Observer Coverage [b]	Obs. Serious Injury[c]	Obs. Mortality	Est. Serious Injury[c]	Est. Mortality	Est. Combined Mortality	Est. CVs	Mean Combined Annual Mortality
Northeast Sink Gillnet	2015	Obs. Data, Trip Logbook, Allocated Dealer Data	0.14	0	23	0	177	177	0.28	145 (0.14)
	2016		0.10	0	11	0	125	125	0.34	
	2017		0.12	1	18	7	129	136	0.28	
	2018		0.11	0	9	0	92	92	0.52	
	2019		0.12	0	33	0	195	195	0.23	
Mid-Atlantic Gillnet	2015	Obs. Data, Weighout	0.06	0	2	0	33	33	1.16	16 (0.68)
	2016		0.08	0	2	0	23	23	0.64	
	2017		0.09	0	1	0	9.1	9.1	0.95	
	2018		0.09	0	0	0	0	0	0	
	2019		0.13	0	2	0	13	13	0.5	
Northeast Bottom Trawl	2015	Obs. Data, Weighout	0.19	0	0	0	0	0	0	2.2 (.63)
	2016		0.12	0	0	0	0	0	0	
	2017		0.12	0	0	0	0	0	0	
	2018		0.12	0	0	0	0	0	0	
	2019		0.16	0	2	0	11	11	0.63	
TOTAL										163 (0.13)

a. Observer data (Obs. Data) are used to measure bycatch rates and the data are collected within the Northeast Fisheries Observer Program. NEFSC collects Weighout (Weighout) landings data that are used as a measure of total effort for the U.S. gillnet fisheries. Mandatory vessel trip report (VTR; Trip Logbook) data are used to determine the spatial distribution of fishing effort in the Northeast sink gillnet fishery.
b. Observer coverage for the U.S. Northeast and mid-Atlantic coastal gillnet fisheries is based on tons of fish landed. Northeast bottom trawl fishery coverages are ratios based on trips.
c. Serious injuries were evaluated for the 2015–2019 period and include both at-sea monitor and traditional observer data (Josephson *et al.* 2022).

Other Mortality

United States

Recent harbor porpoise strandings on the U.S. Atlantic coast are documented in Table 5 (NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 17 November 2020). Of the 417 stranding mortalities reported during this time period, 17 were coded as having signs of human interaction. Of these, 3 were deemed fishery interactions (assumend to be subsumed in the extrapolated fishery bycatch estimates) and 1 was a vessel strike. Most of the remaining Human Interaction (HI) cases were harassment, unlikely to have contributed to the stranding or post-mortem interactions. In only 3 cases were the non-fishery human interactions likely to have been contributing factors in the animal's mortality.

Stranding data underestimate the extent of mortality and serious injury because all of the marine mammals that die or are seriously injured may not wash ashore, nor will all of those that do wash ashore necessarily show signs of entanglement or other fishery-interaction. Finally, the level of technical expertise among stranding network personnel varies widely as does the ability to recognize signs of fishery interaction.

Table 5. Harbor porpoise (Phocoena phocoena phocoena) reported strandings along the U.S. and Canadian Atlantic coast, 2015–2019.

Area	2015	2016	2017	2018	2019	Total
Maine[a, b, e]	2	5	8	8	7	30
New Hampshire	0	1	2	2	6	11
Massachusetts[a, b, d, e, h]	18	8	29	13	68	136
Rhode Island [b, d]	2	2	0	0	2	6
Connecticut	0	0	0	0	1	1
New York [a, d]	3	1	12	2	13	31
New Jersey [a, c, d]	2	5	14	5	6	32
Delaware	0	0	6	0	3	9

Maryland	0	0	2	0	6	8
Virginia [c]	3	2	5	1	6	17
North Carolina [b]	14	1	1	3	13	32
TOTAL U.S.	44	25	79	34	131	313
Nova Scotia/Prince Edward Island[f]	13	16	22	20	30	101
Newfoundland and New Brunswick[g]	2	0	0	0	1	3
GRAND TOTAL	59	41	101	54	162	417

a. In 2016, one animal in Maine and one animal in New Jersey were responded to and released alive. Ten animals were released alive in 2017, 6 of them in Massachusetts, 2 in Maine and 2 in New York.
b. Two HI cases in 2015: 1 in Rhode Island and 1 in North Carolina
c. Two HI cases in 2016: 1 in New Jersey and 1 in Virginia. The Virginia case was coded as a fishery interaction, and the New Jersey case was alive animal relocation.
d. Seven HI cases in 2017: 2 in Maine were released alive and another was a neonate with an infected laceration that required euthanization. One dead HI animal in Massachusetts was coded as a fishery interaction and another HI animal was released alive. One HI animal in New York was released alive and one dead animal in New Jersey had evidence of vessel interaction.
e. Two HI cases in 2018; both in Massachusetts. One was coded as a fishery interaction.
f. Data supplied by Nova Scotia Marine Animal Response Society (pers. comm.). One of the 2015 animals a suspected fishery interaction.
g. See Ledwell and Huntington (2015, 2017, 2018, 2019, 2020).
h. Three Massachusetts stranding mortalities in 2019 were classified as non-fishery human interaction.

Canada

Whales and dolphins stranded on the coast of Nova Scotia, New Brunswick and Prince Edward Island are recorded by the Marine Animal Response Society and the Nova Scotia Stranding Network. See Table 3 for details.

Harbor porpoises stranded on the coasts of Newfoundland and Labrador are reported by the Newfoundland and Labrador Whale Release and Strandings Program (Ledwell and Huntington 2015, 2017, 2018, 2019, 2020; Table 5).

HABITAT ISSUES

In U.S. waters, harbor porpoise are mostly found in nearshore areas and inland waters, including bays, tidal areas, and river mouths. As a result, in addition to fishery bycatch, harbor porpoise are vulnerable to contaminants, such as PCBs (Hall *et al.* 2006), ship traffic (Oakley *et al.* 2017; Terhune 2015) and physical modifications resulting from urban and industrial development activities such as construction of docks and other over-water structures, dredging (Todd *et al*. 2015), installation of offshore windfarms (Carstensen *et al*. 2006; Brandt *et al*. 2011; Teilmann and Carstensen 2012; Dähne *et al.* 2013; Benjamins *et al.* 2017), seismic surveys and other sources of anthropogenic noise (Lucke *et al.* 2009).

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

STATUS OF STOCK

Harbor porpoise in the Gulf of Maine/Bay of Fundy stock are not listed as threatened or endangered under the Endangered Species Act, and this stock is not considered strategic under the MMPA. The total U.S. fishery-related mortality and serious injury for this stock is not less than 10% of the calculated PBR and, therefore, cannot be considered to be insignificant and approaching zero mortality and serious injury rate. The status of harbor porpoises, relative to OSP, in the U.S. Atlantic EEZ is unknown. Population trends for this species have not been investigated.

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

STOCK DEFINITION AND GEOGRAPHIC RANGE

The harbor seal (*Phoca vitulina*) is widespread in all nearshore waters of the North Atlantic and North Pacific Oceans and adjoining seas above about 30ºN (Burns 2009; Desportes *et al*. 2010).

Harbor seals are year-round inhabitants of the coastal waters of eastern Canada and Maine (Katona *et al*. 1993), and occur seasonally along the coasts from southern New England to Virginia from September through late May (Schneider and Payne 1983; Schroeder 2000; Rees *et al*. 2016; Toth *et al*. 2018). Scattered sightings and strandings have been recorded as far south as Florida (NOAA National Marine Mammal Health and Stranding Response Database, accessed 23 October 2018). A general southward movement from the Bay of Fundy to southern New England and mid-Atlantic waters occurs in autumn and early winter (Rosenfeld *et al.* 1988; Whitman and Payne 1990; Jacobs and Terhune 2000). A northward movement to Maine and eastern Canada occurs prior to the pupping season, which takes place from early May through early June primarily along the Maine coast (Gilbert *et al.* 2005; Skinner 2006). The amount of pupping that occurs in Canadian waters is currently unknown.

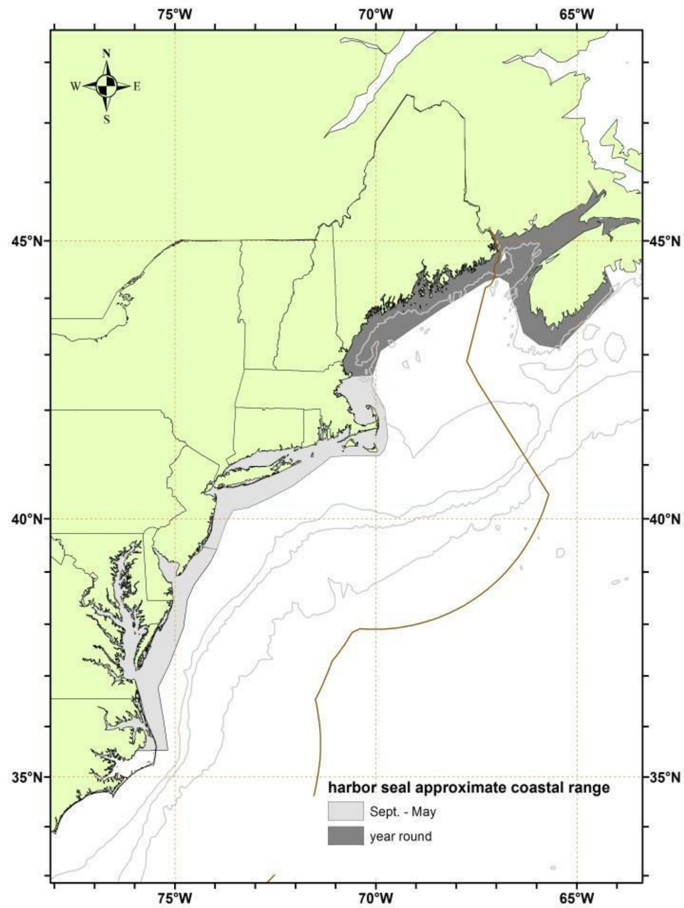

Figure 1. Approximate coastal range of harbor seals. Isobaths are the 100-m, 1000-m, and 4000-m depth contours.

Tagging studies of adult harbor seals demonstrate that adults can make long-distance migrations through the mid-Atlantic and Gulf of Maine (Waring *et al.* 2006; Ampela *et al.* 2018). Prior to these studies, it was believed that the majority of seals moving into southern New England and mid-Atlantic waters were subadults and juveniles (Whitman and Payne 1990; Katona *et al.* 1993). The more recent studies demonstrate that various age classes utilize habitat along the eastern seaboard throughout the year. 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 estimate of harbor seal abundance in U.S. waters is 61,336 (CV=0.08) for 2018, the last year surveyed, based on a Bayesian hierarchical analysis of abundance trends from 1993 to 2018 (Sigourney *et al.* 2021). Estimates of abundance are based on surveys conducted during the pupping season, when most of the population is assumed to be congregated along the Maine coast. Abundance estimates do not reflect the portion of the stock that might pup in Canadian waters. Survey specific correction factors, a means to adjust the survey counts to account for the number of seals in the water at the time of the survey, were not available for most years in the analysis including 2018. Therefore, multiple sources of information on harbor seal haul-out behavior were used to adjust observed counts to estimate total abundance. The 2018 estimate is an average of 2 abundance estimates [70,663 (CV=0.11) and 51,878 (CV=0.10)] derived using different correction factors applied to the estimated number of seals hauled out under ideal conditions.

The 2018 harbor seal pupping survey was designed to survey ledges of known historic occupancy in U.S. waters. If new areas are being populated, they need to be incorporated into future surveys for abundance. Reconnaissance flights for pupping south of Maine would help confirm the extent of the current pupping range and help ensure that some portion of the population is not missed during the survey.

Table 1. Summary of recent abundance estimates for the western North Atlantic harbor seal (Phoca vitulina vitulina) by month, year, and area covered during each abundance survey, and resulting abundance estimate (N_{est}) and coefficient of variation (CV).

Month/Year	Area	N_{est}	CV
May/June 2018	Maine coast	61,336	0.08
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% credible interval about the median of the posterior abundance estimates using the methods of Sigourney *et al*. 2021. This is roughly equivalent to the 20th percentile of the log-normal distribution as specified by Wade and Angliss (1997). The minimum population estimate is 57,637 based on corrected available counts along the Maine coast in 2018.

Current Population Trend

Aerial surveys of harbor seals during the pupping season in Maine have been conducted periodically since 1981 (Gilbert *et al.* 2005; Waring *et al*. 2015; Sigourney *et al*. 2021) and some of these surveys have been used to estimate trends in abundance. Trend in the population from 1993–2018 was estimated for non-pups and pups using a Bayesian hierarchical model to account for missing data both within and between survey years (Sigourney *et al*. 2021). The estimated mean change in non-pup harbor seal abundance per year was positive from 2001 to 2004, but close to zero or negative between 2005 and 2018 (Figure 1a). However, these mean percent changes each year were not statistically significant as evidenced by 95% credible intervals. The estimated mean change in pup abundance was significantly positive from 2001 to 2005. After 2005, mean change in pup abundance was steady or declining until 2018 but these changes were not significant (Figure 1b).

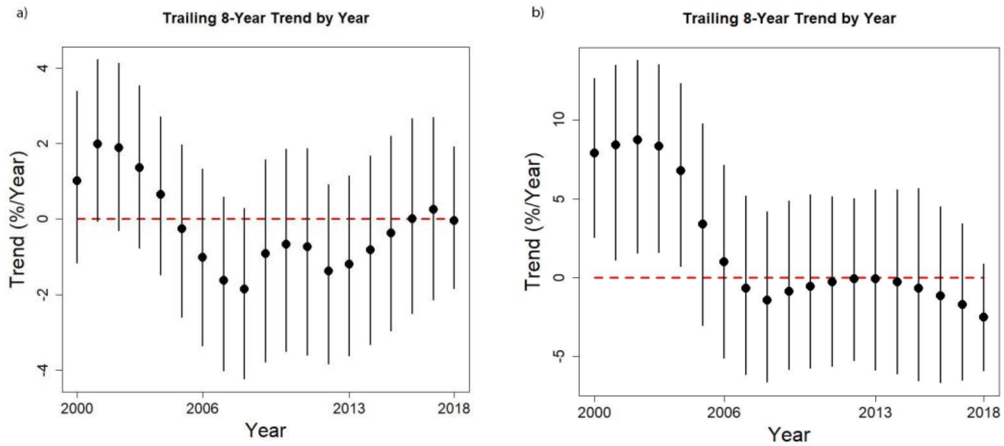

Figure 1. Estimates of average percent change in non-pup (a) and pup (b) harbor seal abundance with 95% Bayesian credible intervals (vertical lines) around the posterior mean over a trailing 8-year moving window starting from 1993.

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 57,637 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 portion of the western North Atlantic stock of harbor seals in U.S. waters is 1,729.

Table 2. Best and minimum abundance estimates for the Western North Atlantic harbor seal (Phoca vitulina vitulina) with Maximum Productivity Rate (R_{max}), Recovery Factor (F_r) and PBR.

N_{est}	CV	N_{min}	F_r	R_{max}	PBR
61,336	0.08	57,637	0.5	0.12	1,729

ANNUAL HUMAN-CAUSED SERIOUS INJURY AND MORTALITY

For the period 2015–2019, the annual average annual estimated human-caused mortality and serious injury to harbor seals in the U.S. is 339 (Table 3). Mortality in U.S. fisheries is explained in further detail below.

Table 3. The total annual estimated average human-caused mortality and serious injury for the Western North Atlantic harbor seal (Phoca vitulina vitulina).

Years	Source	Annual Avg.	CV
2015–2019	U.S. fisheries using observer data	334	0.09
2015–2019	Non-fishery human interaction stranding mortalities	4.6	-

2015–2019	Research mortalities	0	-
TOTAL		339	-

Fishery Information

Detailed fishery information is given in Appendix III.

United States

Northeast Sink Gillnet

The Northeast sink gillnet fishery is a Category I fishery. The average annual observed mortality from 2015–2019 was 53 animals, and the average annual total mortality was 304 (CV=0.10; Orphanides and Hatch 2017; Orphanides 2019, 2020, 2021; Precoda and Orphanides 2022; Josephson *et al.* 2022). See Table 4 for bycatch estimates and observed mortality and serious injury for the current 5-year period, and Appendix V for historical bycatch information.

Mid-Atlantic Gillnet

The Mid-Atlantic sink gillnet fishery is a Category I fishery. The average annual observed mortality from 2015–2019 was 3 animals, and the average annual total mortality was 22 (CV=0.30; Orphanides and Hatch 2017; Orphanides 2019, 2020, 2021; Precoda and Orhanides 2022; Josephson *et al.* 2022). See Table 4 for bycatch estimates and observed mortality and serious injury for the current 5-year period, and Appendix V for historical bycatch information.

Northeast Bottom Trawl

The Northeast bottom trawl fishery is a Category II fishery. The average annual observed mortality from 2015–2019 was <1 animal, and the average annual total mortality was 3 (CV=0.68; Lyssikatos and Chavez-Rosales 2022). See Table 4 for bycatch estimates and observed mortality and serious injury for the current 5-year period, and Appendix V for historical bycatch information.

Mid-Atlantic Bottom Trawl

The Mid-Atlantic bottom trawl fishery is a Category II fishery. The average annual observed mortality from 2015–2019 was <1 animal, and the average annual total mortality was 4 (CV=0.56; Lyssikatos and Chavez-Rosales 2022). See Table 4 for bycatch estimates and, observed mortality and serious injury for the current 5-year period, and Appendix V for historical bycatch information.

Northeast Mid-water Trawl Fishery (Including Pair Trawl)

The Northeast mid-water and pair trawl fisheries are Category II fisheries. The average annual observed mortality from 2015–2019 was <1 animal. An expanded bycatch estimate has not been calculated for the current 5-year period. See Table 4 for observed mortality and serious injury during the current 5-year period, and Appendix V for historical bycatch information.

Canada

Currently, scant data are available on bycatch in Atlantic Canada fisheries due to limited observer programs (Baird 2001). An unknown number of harbor seals have been taken in Newfoundland, Labrador, Gulf of St. Lawrence and Bay of Fundy groundfish gillnets; Atlantic Canada and Greenland salmon gillnets; Atlantic Canada cod traps; and in Bay of Fundy herring weirs (Read 1994; Cairns *et al.* 2000). Furthermore, some of these mortalities (e.g., seals trapped in herring weirs) are the result of direct shooting under nuisance permits.

Table 4. Summary of the incidental mortality of harbor seals (Phoca vitulina vitulina) by commercial fishery including the years sampled (Years), the type of data used (Data Type), the annual observer coverage (Observer Coverage), the mortalities recorded by on-board observers (Observed Mortality), the estimated annual mortality (Estimated Mortality), the estimated CV of the annual mortality (Estimated CVs) and the mean annual mortality (CV in parentheses).

Fishery	Years	Data Type [a]	Observer Coverage [b]	Observed Serious Injury [c]	Observed Mortality	Estimated Serious Injury	Estimated Mortality	Estimated Combined Mortality	Estimated CVs	Mean Annual Mortality
Northeast Sink Gillnet	2015 2016 2017 2018 2019	Obs. Data, Weighout, Logbooks	0.14 0.10 0.12 0.11 0.13	0 0 0 0 0	87 36 63 22 59	0 0 0 0 0	474 245 298 188 316	474 245 298 188 316	0.17 0.29 0.18 0.36 0.15	304 (0.1)
Mid-Atlantic Gillnet	2015 2016 2017 2018 2019	Obs. Data, Weighout	0.06 0.08 0.09 0.09 0.12	0 0 0 0 0	5 2 1 3 3	0 0 0 0 0	48 18 3 26 17	48 18 3 26 17	0.52 0.95 0.62 0.52 0.35	22 (0.3)
Northeast Bottom Trawl	2015 2016 2017 2018 2019	Obs. Data, Weighout	0.19 0.12 0.16 0.12 0.16	0 0 0 0 0	0 0 0 0 1	0 0 0 0 0	0 0 8 0 5	0 0 8 0 5	0 0 0.96 0 0.88	2.7 (0.68)
Mid-Atlantic Bottom Trawl	2015 2016 2017 2018 2019	Obs. Data, Dealer	0.09 0.10 0.14 0.12 0.12	0 0 0 0 0	1 0 0 1 1	0 0 0 0 0	7 0 0 6 7	7 0 0 6 7	1 0 0 0.94 0.93	4.0 (0.56)
Northeast Mid-water Trawl - Including Pair Trawl	2015 2016 2017 2018 2019	Obs. Data, Weighout, Trip Logbook	0.08 0.27 0.16 0.14 0.28	0 0 0 0 0	2 1 0 0 0	0 0 0 0 0	na na 0 0 0	na na 0 0 0	na na 0 0 0	0.6 (na)
TOTAL										334 (0.09)

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 2014–2018 includes samples collected from traditional fisheries observers in addition to fishery monitors through the Northeast Fisheries Observer Program (NEFOP).
c. Serious injuries were evaluated for the 2015–2019 period and include both at-sea monitor and traditional observer data (Josephson *et al.* 2022)

Other Mortality

United States

Historically, harbor seals were bounty-hunted in New England waters, which may have caused a severe decline of this stock in U.S. waters (Katona *et al*. 1993; Lelli *et al*. 2009). Bounty-hunting ended in the mid-1960s.

Harbor seals strand each year throughout their migratory range. Stranding data provide insight into some of these sources of mortality. Tables 5 and 6 present summaries of harbor seal stranding mortalities as reported to the NOAA National Marine Mammal Health and Stranding Response Database (accessed 17 November 2020). 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.

Table 5. Harbor seal (Phoca vitulina vitulina) stranding mortalities along the U.S. Atlantic coast (2015–2019) with subtotals of animals recorded as pups in parentheses.

State	2015	2016	2017	2018	2019	Total
Maine	73 (47)	76 (58)	120 (84)	819 (75)	188 (59)	1,276 (323)
New Hampshire	56 (43)	45 (27)	26 (20)	113 (60)	26 (2)	266 (152)

Massachusetts	81 (24)	55 (19)	78 (29)	204 (58)	72 (12)	490 (142)
Rhode Island	8 (0)	5 (1)	9 (3)	9 (0)	10 (3)	41 (7)
Connecticut	2 (1)	1 (0)	2 (0)	2 (1)	2 (0)	9 (2)
New York	21 (0)	1 (0)	11 (0)	12 (1)	13 (0)	58 (1)
New Jersey	9 (4)	4 (0)	9 (3)	14 (2)	4 (0)	40 (9)
Delaware	1 (0)	1 (1)	1 (0)	2 (1)	3 (0)	8 (2)
Maryland	0	0	1 (0)	4 (0)	2 (0)	7 (0)
Virginia	1 (0)	1 (0)	2 (0)	1 (0)	4 (0)	9 (0)
North Carolina	5 (2)	4 (2)	4 (4)	7 (2)	2 (1)	22 (11)
Total	257 (121)	193 (108)	263 (143)	1,187 (200)	326 (77)	2,226 (649)
Unspecified seals (all states)	31	13	86	92	80	302

Table 6. Harbor seal (Phoca vitulina vitulina) human-interaction stranding mortalities along the U.S. Atlantic coast (2015–2019) by type of interaction.

Cause	2015	2016	2017	2018	2019	Total
Fishery Interaction	2	3	1	5	3	14
Boat Strike	1	5	3	2	0	11
Shot	0	0	0	0	0	0
HI - Other - possible contribution to death	4	1	1	5	1	12
HI - Other - not contributing to death, or unk	11	7	5	17	8	48
TOTAL	18	16	10	29	12	85

A number of Unusual Mortality Events (UMEs) have affected harbor seals over the past decade. The most recent was declared by the NMFS beginning in July 2018 due to increased numbers of harbor and gray seal strandings along the U.S. coasts of Maine, New Hampshire, and Massachusetts. Strandings remained elevated over the summer and the UME area was expanded to include nine states from Maine to Virginia with strandings continuing into 2019. From July 1, 2018 to March 13, 2020, 3,152 seals (including harbor and gray seals) stranded from Maine to Virginia. The preliminary cause of the UME was attributed to a phocine distemper outbreak (https://www.fisheries.noaa.gov/new-england-mid-atlantic/marine-life-distress/2018-2020-pinniped-unusual-mortality-event-along).

Stobo and Lucas (2000) have documented shark predation as an important source of natural mortality at Sable Island, Nova Scotia. They suggest that shark-inflicted mortality in pups, as a proportion of total production, was less than 10% in 1980–1993, approximately 25% in 1994–1995, and increased to 45% in 1996. Also, shark predation on adults was selective towards mature females. The decline in the Sable Island population appears to result from a combination of shark-inflicted mortality on both pups and adult females and inter-specific competition with the much more abundant gray seal for food resources (Stobo and Lucas 2000; Bowen *et al.* 2003).

Canada

Aquaculture operations in eastern Canada can be licensed to shoot nuisance seals, but the number of seals killed is unknown (Jacobs and Terhune 2000; Baird 2001). Small numbers of harbor seals are taken in subsistence hunting in northern Canada (DFO 2011). Four animals were taken in 2019 for scientific research (Samuel Mongrain, pers comm.).

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 2015–2019 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*) is found on both sides of the North Atlantic, with three major populations: Northeast Atlantic, Northwest Atlantic and the Baltic Sea (Haug *et al.* 2007). The Northeast Atlantic and the Northwest Atlantic populations are classified as the subspecies *H. g. atlantica* (Olsen *et al.* 2016). The Northwest Atlantic population which defines the western North Atlantic stock ranges from New Jersey to Labrador (Davies 1957; Mansfield 1966; Katona *et al.* 1993; Lesage and Hammill 2001). This stock is separated from the northeastern Atlantic stocks by geography, differences in the breeding season, and mitochondrial and nuclear DNA variation (Bonner 1981; Boskovic *et al.* 1996; Lesage and Hammill 2001; Klimova *et al.* 2014). There are three breeding aggregations in eastern Canada: Sable Island, Gulf of St. Lawrence, and at sites along the coast of Nova Scotia (Laviguer and Hammill 1993). Animals from these aggregations mix outside the breeding season (Lavigueur and Hammill 1993; Harvey *et al.* 2008; Breed *et al.* 2006, 2009) and they are considered a single population based on genetic similarity (Boskovic *et al.* 1996; Wood *et al.* 2011).

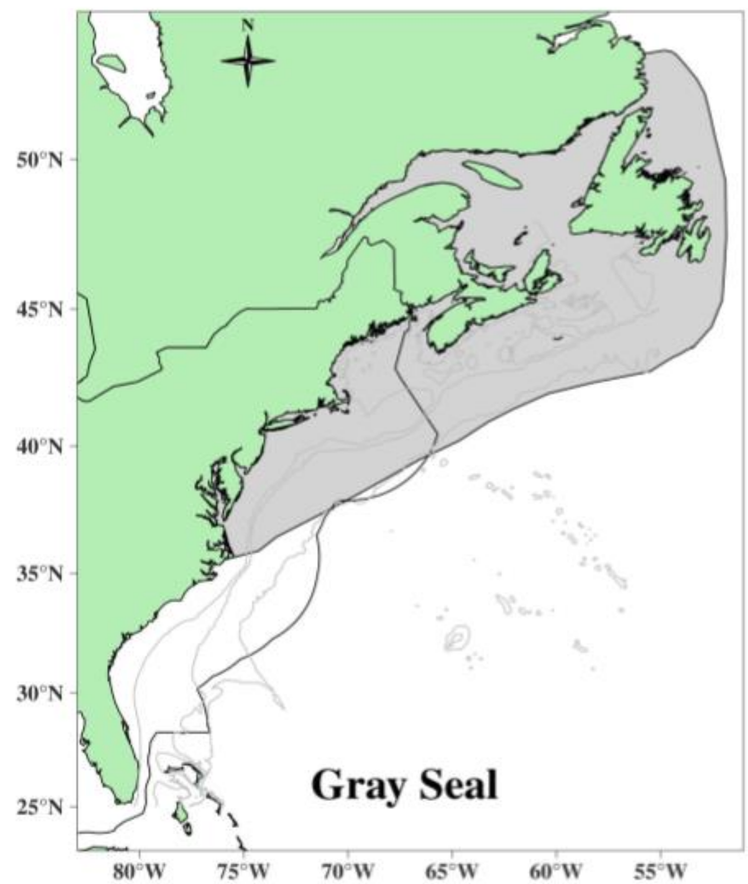

Figure 1: Approximate range of the Western North Atlantic stock of gray seals (Halichoerus grypus atlantica).

After near extirpation due to bounties, which ended in the 1960s, small numbers of animals and pups were observed on several isolated islands along the Maine coast and in Nantucket Sound, Massachusetts (Katona *et al.* 1993; Rough 1995; Gilbert *et al*. 2005). In December 2001, NMFS initiated aerial surveys to monitor gray seal pup production on Muskeget Island and adjacent sites in Nantucket Sound, and Green and Seal Islands off the coast of Maine (Wood *et al.* 2007). Tissue samples collected from Canadian and U.S. populations were examined for genetic variation using mitochondrial and nuclear DNA (Wood *et al.* 2011). All individuals were identified as belonging to one population, confirming the new U.S. population was recolonized by Canadian gray seals. The genetic evidence (Boskovic *et al.* 1996; Wood *et al.* 2011) provides a high degree of certainty that the western North Atlantic stock of gray seals comprise a single stock. Further supporting evidence comes from sightings of seals in the U.S. that had been branded on Sable Island, resights of tagged animals, and satellite tracks of tagged animals (Puryear *et al.* 2016). The amount of mixing and percentage of time that individuals use U.S. and Canadian waters is unknown.

POPULATION SIZE

Currently there is a lack of information on the rate of exchange between animals in the U.S. and Canada, which influences seasonal changes in abundance throughout the range of this transboundary stock as well as life history parameters in population models. As a result, the size of the Northwest Atlantic gray seal population is estimated separately for the portion of the population in Canada versus the U.S., and mainly reflects the size of the breeding population in each respective country (Table 1). Total pup production in 2016 at breeding colonies in Canada was estimated to be 102,100 pups (CV=0.15; den Heyer *et al*. 2020). Production at Sable Island, Gulf of St. Lawrence, and Coastal Nova Scotia colonies accounted for 85%, 10% and 5%, respectively, of the estimated total number of pups born. Population models, incorporating estimates of age-specific reproductive rates and removals, are fit to these pup production estimates to estimate total population levels in Canada. The total Canadian gray seal population in 2016 was estimated to be 424,300 (95% CI=263,600 to 578,300; DFO 2017). Uncertainties in the population estimate derive from uncertainties in life history parameters such as mortality rates and sex ratios (DFO 2017).

The Northwest Atlantic gray seal population has been described as a metapopulation with a mainland-island structure, due to the size of the breeding colony on Sable Island in relation to other colonies and the movement of animals between them (den Heyer *et al*. 2020). In U.S. waters, the number of pupping sites has increased from 1 in 1988 to 9 in 2019, and are located in Maine and Massachusetts (Wood *et al*. 2020). Although white-coated pups have stranded on eastern Long Island beaches in New York, no pupping colonies have been detected in that region.

An estimated 6,500 pups were born in 2016 at U.S. breeding colonies (den Heyer *et al*. 2020), approximately 6% of the total pup production over the entire range of the population (den Heyer *et al.* 2020). Muskeget Island is the largest pupping colony in the U.S. and the third largest of all colonies across the U.S. and Canada (den Heyer *et al*. 2020). Mean rates of increase in the minimum number of pups born at various times since 1988 at 4 of the more frequently surveyed pupping sites (Muskeget, Monomoy, Seal, and Green Islands) ranged from -0.2% (95%CI: -2.3–1.9%) to 26.3% (95%CI: 21.6–31.4%; Wood *et al.* 2020). These high rates of increase provide further support that seals are recruiting to U.S. colonies from larger established breeding colonies in Canada.

The number of pups born at U.S. breeding colonies can be used to approximate the total size (pups and adults) of the gray seal population in U.S. waters, based on the ratio of total population size to pups in Canadian waters (4.2:1) (den Heyer *et al.* 2020; DFO 2017). Although not yet measured for U.S. waters, this ratio falls within the range of other adult to pup ratios suggested for pinniped populations (Harwood and Prime 1978; Thomas *et al.* 2019). Using this approach, the population estimate during the pupping season in U.S. waters is 27,300 (CV=0.22, 95% CI: 17,828–41,804) animals. The CV and CI around this estimate is based on CVs and CIs from Canadian population estimates, rather than using a default CV when the variance is unknown (Wade and Angliss 1997). There is further uncertainty in this abundance level in the U.S. because life history parameters that influence the ratio of pups to total individuals in this portion of the population are unknown. It also does not reflect seasonal changes in stock abundance in the Northeast region for a transboundary stock. For example, roughly 24,000 seals were observed in southeastern Massachusetts alone in 2015 (Pace *et al.* 2019), yet 28,000–40,000 gray seals were estimated to be in this region in 2015 using correction factors applied to seal counts obtained from Google Earth imagery (Moxley *et al.* 2017).

Table 1. Summary of recent abundance estimates for the western North Atlantic gray seal (Halichoerus grypus atlantica) by year, and area covered, resulting total abundance estimate and 95% confidence interval.

Year	Area	N_{est} [a]	CI
2014[b]	Gulf of St Lawrence + Nova Scotia Eastern Shore + Sable Island	505,000	329,000–682,000
2016[c]	Gulf of St Lawrence + Nova Scotia Eastern Shore + Sable Island	424,300	263,600–578,300
2016	U.S.	27,300[d]	17,828–41,804

a. These are model-based estimates derived from pup surveys.
b. DFO 2014
c. DFO 2017
d. This is derived from total population size to pup ratios in Canada, applied to U.S. pup counts.

Minimum Population Estimate

The minimum population estimate is the lower limit of the two-tailed 60% confidence interval of the log-normally distributed best abundance estimate. This is equivalent to the 20th percentile of the log-normal distribution as specified by Wade and Angliss (1997). Based on an estimated U.S. population in 2016 of 27,300 (CV=0.22), the minimum population estimate in U.S. waters is 22,785 (Table 2). Similar to the best abundance estimate, there is uncertainty in this minimum abundance level in the U.S. because life history parameters that influence the ratio of pups to total individuals in this population are unknown. Furthermore, this minimum population estimate reflects a portion of the stock's range and may vary seasonally as some portion of the larger stock moves in and out of U.S. waters.

Current Population Trend

In the U.S., the mean rate of increase in the number of pups born differs across the pupping colonies. From 1988–2019, the estimated mean rate of increase in the minimum number of pups born was 12.8% on Muskeget Island, 26.3% on Monomoy Island, 11.5% on Seal Island, and -0.2% on Green Island (Wood *et al*. 2020). These rates only reflect new recruits to the population and do not reflect changes in total population growth resulting from Canadian seals migrating to the region.

The total population of gray seals in Canada was estimated to be increasing by 4.4% per year from 1960–2016 (Hammill *et al*. 2017), primarily due to increases at Sable Island. Pup production on Sable Island increased exponentially at a rate of 12.8% per year between the 1970s and 1997 (Bowen *et al.* 2003). Pupping also occurs on Hay Island off Nova Scotia, in colonies off southwestern Nova Scotia, and in the Gulf of St. Lawrence. Since 1997, the rate of increase has slowed (Bowen *et al.* 2011; den Heyer *et al.* 2017), supporting the hypothesis that density-dependent changes in vital rates may be limiting population growth. While slowing, pup production is still increasing on Sable Island at a rate of 5–7% per year (den Heyer *et al.* 2020). Pup production is also increasing in southwest Nova Scotia, and appears to be stabilizing in the Gulf of St. Lawrence (DFO 2017; den Heyer *et al.* 2020). In the Gulf of St. Lawrence, the proportion of pups born on the ice has declined from 100% in 2004 to 1% in 2016 due to a decline in winter ice cover in the area, and seals have responded by pupping on nearby islands (DFO 2017).

CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

For purposes of this assessment, the maximum net productivity rate was assumed to be 0.128, based on historic rates of increase observed on Sable Island (Bowen *et al*. 2003).

POTENTIAL BIOLOGICAL REMOVAL

Potential Biological Removal (PBR) is the product of minimum population size, one-half the maximum productivity rate, and a recovery factor (MMPA Sec. 3. 16 U.S.C. 1362; Wade and Angliss 1997). The minimum population size for the portion of the stock residing in U.S. waters is 23,158. The maximum productivity rate is 0.128. 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 portion of the western North Atlantic stock of gray seals residing in U.S. waters is 1,458 animals (Table 2). Uncertainty in the PBR level arises from uncertainty in seasonal changes in gray seal abundance in U.S. waters, and rates of exchange between animals in Canada and the U.S.

Table 2. Best and minimum abundance estimates for the western North Atlantic gray seal (Halichoerus grypus atlantica) with Maximum Productivity Rate (R_{max}), Recovery Factor (F_r) and PBR.

N_{est}	CV	N_{min}	F_r	R_{max}	PBR
27,300	0.22	22,785	1	0.128	1,458

ANNUAL HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

For the period 2015–2019, the average annual estimated human-caused mortality and serious injury to gray seals in the U.S. and Canada was 4,452 (1,178 for the U.S. and 3,274 for Canada) per year. Mortality in U.S. fisheries is explained in further detail below.

Table 3. The total annual estimated average human-caused mortality and serious injury for the western North Atlantic gray seal (Halichoerus grypus atlantica).

Years	Source	Annual Avg.
2015–2019	U.S. commercial fisheries using observer data	1,169
2015–2019	U.S. non-fishery human-caused stranding mortalities	7.8

2015–2019	U.S. research mortalities	1.2
2015–2019	Canadian commercial harvest	867
2015–2019	DFO Canada scientific collections	57
2015–2019	Canadian removals of nuisance animals	2,350
TOTAL		4,452

Some human-caused mortality or serious injury may not be able to be quantified. Observed serious injury rates are lower than would be expected from the anecdotally observed numbers of gray seals living with ongoing entanglements. Estimated rates of entanglement in gillnet gear, for example, may be biased low because 100% of observed animals are dead when they come aboard the vessel (Josephson *et al.* 2022); therefore, rates do not reflect the number of live animals that may have broken free of the gear, but remain entangled. For example, mean prevalence of live entangled gray seals ranged from roughly 1 to 4% at haul-out sites in Massachusetts and Isles of Shoals (Iruzun Martins *et al.* 2019). Reports of seal shootings and other non-fishery-related human M/SI are minimum counts. Incomplete information on the true number of seals living with serious injuries from entanglements increases the amount of uncertainty in the estimated fisheries-related mortality.

Fishery Information

Detailed fishery information is given in Appendix III.

United States

Northeast Sink Gillnet

Northeast sink gillnet fishery is a Category I fishery. The average annual observed mortality from 2015–2019 was 137 animals, and the average annual estimated total mortality was 1,115 (CV=0.17; Orphanides and Hatch 2017; Orphanides 2019, 2020, 2021; Precoda and Orphanides 2022). See Table 4 for bycatch estimates and observed mortality and serious injury for the current 5-year period, and Appendix V for historical bycatch information.

Mid-Atlantic Gillnet

The Mid-Atlantic sink gillnet fishery is a Category I fishery. The average annual observed mortality from 2015–2019 was 1 animal, and the average annual total mortality was 8 (CV=0.46; Hatch and Orphanides 2016; Orphanides and Hatch 2017; Orphanides 2019, 2020, 2021; Precoda and Orphanides 2022). See Table 4 for bycatch estimates and observed mortality and serious injury for the current 5-year period, and Appendix V for historical bycatch information.

Gulf of Maine Atlantic Herring Purse Seine Fishery

The Gulf of Maine Atlantic Herring Purse Seine Fishery is a Category III fishery. No mortalities have been observed in this fishery, during the current 5-year period, however, 5 gray seals were captured and released alive in 2016 and 1 in 2018. In addition, 2 seals of unknown species were captured and released alive in 2015 and 1 in 2016 (Josephson *et al.* 2022).

Northeast Bottom Trawl

The Northeast bottom trawl fishery is a Category II fishery. The average annual observed mortality from 2015–2019 was 3 animals, and the average annual total mortality was 20 (CV=0.23; Lyssikatos and Chavez-Rosales 2022). See Table 4 for bycatch estimates and observed mortality and serious injury for the current 5-year period, and Appendix V for historical bycatch information.

Mid-Atlantic Bottom Trawl

The Mid-Atlantic bottom trawl fishery is a Category II fishery. The average annual observed mortality from 2015–2019 was 4 animals, and the average annual total mortality was 26 (CV=0.30; Lyssikatos and Chavez-Rosales 2022). See Table 4 for bycatch estimates and observed mortality and serious injury for the current 5-year period, and Appendix V for historical bycatch information.

Northeast Mid-Water and Pair Trawl

The Northeast mid-water and pair trawl fisheries are Category II fisheries. Only 1 gray seal was observed in these fisheries from 2015–2019 and an expanded bycatch estimate has not been generated. See Table 4 for observed mortality and serious injury for during the current 5-year period, and Appendix V for historical bycatch information.

Table 4. Summary of the incidental serious injury and mortality of gray seal (Halichoerus grypus atlantica) by commercial fishery including the years sampled, the type of data used (Data Type), the annual observer coverage (Observer Coverage), the mortalities recorded by on-board observers (Observed Mortality), the estimated annual mortality (Estimated Mortality), the estimated CV of the annual mortality (Estimated CVs) and the mean annual mortality (CV in parentheses).

Fishery	Years	Data Type [a]	Observer Coverage [b]	Observed Serious Injury [c]	Observed Mortality	Est. Serious Injury	Est. Mortality	Est. Comb. Mortality	Est. CVs	Mean Annual Combined Mortality
Northeast Sink Gillnet	2015	Obs. Data, Weighout, Trip Logbook	0.14	0	131	0	1021	1021	0.25	1,115 (0.11)
	2016		0.10	0	43	0	498	498	0.33	
	2017		0.12	0	158	0	930	930	0.16	
	2018		0.11	0	103	0	1113	1113	0.32	
	2019		0.13	0	251	0	2014	2014	0.17	
Mid-Atlantic Gillnet	2015	Obs. Data, Trip Logbook, Allocated Dealer Data	0.06	0	1	0	15	15	1.04	8.0 (0.46)
	2016		0.08	0	1	0	7	7	0.93	
	2017		0.09	0	0	0	0	0	0	
	2018		0.09	0	0	0	0	0	0	
	2019		0.12	0	3	0	18	18	0.4	
Northeast Bottom Trawl	2015	Obs. Data, Trip Logbook	0.19	0	4	0	23	23	0.46	20 (0.23)
	2016		0.12	0	0	0	0	0	0	
	2017		0.12	0	2	0	16	16	0.24	
	2018		0.12	0	5	0	32	32	0.42	
	2019		0.16	0	6	0	30	30	0.37	
Mid-Atlantic Bottom Trawl	2015	Obs. Data, Trip Logbook	0.09	0	0	0	0	0	0	26 (0.30)
	2016		0.10	0	3	0	26	26	0.57	
	2017		0.14	0	5	0	26	26	0.40	
	2018		0.12	0	7	0	56	56	0.58	
	2019		0.12	0	3	0	22	22	0.53	
Northeast Mid-water Trawl – Incl.Pair Trawl	2015	Obs. Data, Trip Logbook	0.08	0	0	0	0	0	0	0.2 (na) [d]
	2016		0.27	0	0	0	0	0	0	
	2017		0.16	0	0	0	0	0	0	
	2018		0.14	0	1	0	na	na	na	
	2019		0.28	0	0	0	0	0	0	
TOTAL										1169 (0.10)

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 2015–2019 period (Josephson *et al*. 2022)
d. No estimate made. Raw counts provided.

Research Takes

From 2015–2019 there were a total of 6 gray seal mortalities which occurred incidentally during research activities: 0 in 2015, 3 in 2016, 1 in 2017, 2 in 2018, and 0 in 2019.

Canada

There is limited information on Canadian fishery bycatch (DFO 2017). Historically, an unknown number of gray seals have been taken in Newfoundland and Labrador, Gulf of St. Lawrence, and Bay of Fundy groundfish gillnets; Atlantic Canada and Greenland salmon gillnets; Atlantic Canada cod traps, and Bay of Fundy herring weirs (Read 1994). The lack of information on bycatch in Canada increases the uncertainty in the total level of fishery mortality impacting this transboundary stock.

Other Mortality

United States

Gray seals, like harbor seals, were hunted for bounty in New England waters until the late 1960s (Katona *et al.* 1993; Lelli *et al*. 2009). This hunt may have severely depleted this stock in U.S. waters (Rough 1995; Lelli *et al*. 2009). Other sources of mortality include human interactions, storms, abandonment by the mother, disease, and shark predation. Mortalities caused by human interactions include research mortalities, boat strikes, fishing gear interactions, power plant entrainment, oil spill/exposure, harassment, and shooting. Seals entangled in netting are common at haul-out sites in the Gulf of Maine and Southeastern Massachusetts.

Tables 5 and 6 present summaries of gray seal stranding as reported to the NOAA National Marine Mammal Health and Stranding Response Database (accessed 17 November 2020). Most stranding mortalities were in Massachusetts, which is the center of gray seal abundance in U.S. waters. In an analysis of mortality causes of stranded marine mammals on Cape Cod and southeastern Massachusetts between 2000 and 2006, Bogomolni *et al.* (2010) reported that 45% of gray seal stranding mortalities were attributed to human interaction.

An Unusal Mortality Event (UME) was declared in November of 2011 that involved at least 137 gray seal stranding mortalities between June 2011 and October 2012 in Maine, New Hampshire, and Massachusetts. The UME was declared closed in February 2013 (https://www.fisheries.noaa.gov/national/marine-life-distress/active-and-closed-unusual-mortality-events) and was believed to have been caused by an Influenza A virus (Anthony *et al*. 2012). More reently, a UME was declared in July 2018 due to increased numbers of harbor and gray seal strandings along the U.S. coasts of Maine, New Hampshire, and Massachusetts. From July 1, 2018 to March 13, 2020, 3,152 seals (including harbor and gray seals) stranded from Maine to Virginia. The preliminary cause of the UME was attributed to a phocine distemper outbreak (https://www.fisheries.noaa.gov/new-england-mid-atlantic/marine-life-distress/2018-2020-pinniped-unusual-mortality-event-along).

Table 5. Gray seal (Halichoerus grypus atlantica) stranding mortalities along the U.S. Atlantic coast (2015–2019) with subtotals of animals recorded as pups in parentheses.

State	2015	2016	2017	2018	2019	Total
Maine	5	6 (0)	14 (1)	25 (0)	15 (0)	65 (1)
New Hampshire	2	0	3 (0)	9 (3)	5 (0)	19 (3)
Massachusetts	77 (3)	54 (0)	135 (21)	261 (29)	260 (80)	787 (133)
Rhode Island	7 (1)	4 (0)	16 (5)	20 (3)	28 (8)	75 (17)
Connecticut	0	0	3 (0)	1 (0)	0	4 (0)
New York	10	1 (1)	16 (0)	25 (1)	43 (4)	95 (6)
New Jersey	7 (6)	3 (1)	4 (3)	14 (10)	9 (8)	37 (28)
Delaware	3 (3)	0	1 (0)	4 (2)	2 (1)	10 (6)
Maryland	0	0	0	1 (1)	0	1 (1)
Virginia	3	0	0	1 (1)	0	4 (1)
North Carolina	0	0	0	5 (2)	0	5 (2)
Total	114 (13)	68 (2)	233 (30)	366 (52)	362 (101)	1143 (198)
Unspecified seals (all states)	31	13	86	92	80	302

Table 6. Documented gray seal (Halichoerus grypus atlantica) human-interaction related stranding mortalities along the U.S. Atlantic coast (2015–2019) by type of interaction. "Fishery interactions" are subsumed in the total estimated mortality calculated from observer data.

TypeCause	2015	2016	2017	2018	2019	Total
Fishery Interaction	14	0	10	10	8	42
Boat Strike	3	0	4	2	1	10
Shot	1	1	0	0	0	2
Human Interaction - Other	2	0	3	9	13	27
TOTAL	20	1	17	21	22	81

Canada

Between 2015–2019, the average annual human-caused mortality and serious injury to gray seals in Canadian waters from commercial harvest is 867, though up to 60,000 seals/year are permitted (http://www.dfo-mpo.gc.ca/decisions/fm-2015-gp/atl-001-eng.htm). This included: 1,381 in 2015, 1,588 in 2016, 64 in 2017, 66 in 2018, and 1,235 in 2019 (DFO 2017; Fitzgibbon pers. comm.). In addition, between 2015 and 2019, an average of 2350 nuisance animals per year were killed. This included 3,732 annually in 2014–2017 (DFO 2017), 461 in 2018 based on the total number of licenses that were issued (Courtney D'Aoust, pers. comm.), and 95 in 2019 (Sylvia Fitzgibbon pers. comm.). Lastly, DFO took 42 animals in 2015, 30 animals in 2016, 60 animals in 2017, 96 animals in 2018, and 58 animals in 2019 for scientific collections, for an annual average of 57 animals (DFO 2017; Samuel Mongrain pers. comm.).

STATUS OF STOCK

Gray seals are not listed as threatened or endangered under the Endangered Species Act, and the western North Atlantic stock is not considered strategic under the Marine Mammal Protection Act. The average annual human-caused mortality and serious injury during 2015–2019 in U.S. waters does not exceed the PBR of the U.S. portion of the stocks. The status of the gray seal population relative to Optimum Sustainable Population (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 in the rates of exchange and levels of mixing between animals using U.S. and Canadian waters, as well as fishery related mortality in both the U.S. and Canada, 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 (Figure 1). The Front herd breeds off the coast of Newfoundland and Labrador, and the Gulf herd breeds near the Magdalen Islands in the middle of the Gulf of St. Lawrence (Sergeant 1965; Lavigne and Kovacs 1988). The second stock breeds on the West Ice off eastern Greenland (Lavigne and Kovacs 1988), and the third stock breeds on the ice in the White Sea off the coast of Russia. The Front/Gulf stock is equivalent to the western North Atlantic stock. Perry *et al.* (2000) found no significant genetic differentiation between the two Northwest Atlantic whelping areas, though the authors pointed out some uncertainty surrounding that finding due to small sample sizes.

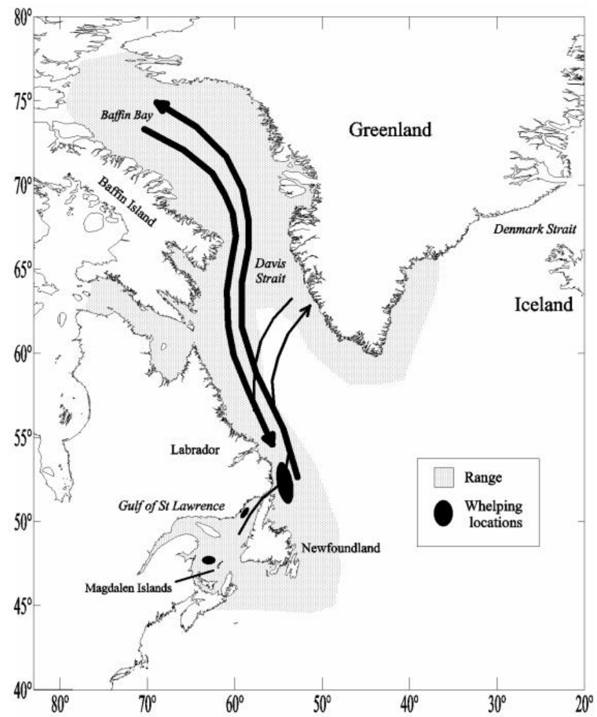

Figure 1. Current Status of Northwest Atlantic Harp Seals, Pagophilus groenlandicus

Harp seals are highly migratory (Sergeant 1965; Stenson and Sjare 1997). Breeding occurs at different times for each stock between late-February and April. Adults then assemble on suitable pack ice to undergo the annual molt. The migration then continues north to Arctic summer feeding grounds. In late September, after a summer of feeding, nearly all adults and some of the immature animals of the western North Atlantic stock migrate southward along the Labrador coast, usually reaching the entrance to the Gulf of St. Lawrence by early winter. There they split into two groups, one moving into the Gulf and the other remaining off the coast of Newfoundland. The southern limit of the harp seal's habitat extends into the U.S. Atlantic Exclusive Economic Zone (EEZ) during winter and spring.

Since the early 1990s, numbers of sightings and strandings have been increasing off the east coast of the United States from Maine to New Jersey (Katona *et al.* 1993; Rubinstein 1994; Stevick and Fernald 1998; McAlpine 1999; Lacoste and Stenson 2000; Soulen *et al*. 2013). These appearances usually occur in January–May (Harris *et al.* 2002), when the western North Atlantic stock of harp seals is at its most southern point of migration. Concomitantly, a southward shift in winter distribution off Newfoundland was observed during the mid-1990s, which was attributed to abnormal environmental conditions (Lacoste and Stenson 2000).

POPULATION SIZE

The size of the western North Atlantic stock of harp seals is estimated by fitting age-structured population models to estimates of total pup production in Canada. Since 1990, aerial surveys of the whelping patches have been flown to determine pup production (Stenson *et al.* 2020a). These estimates are then fit to population models taking into account reproductive rates, ice-related mortality, and anthropogenic removals. Total estimated pup production from the last pupping survey which occurred in March 2017 was 746,500 (95% CI: 570,300–922,700; DFO 2020). There was some uncertainty in results of the survey due to poor ice conditions in the southern Gulf of St Lawrence, and changes in the timing of pupping due to the movement of animals among whelping patches (Stenson *et al.* 2020a). After the 2017 survey the population model was updated to account for the effects of continued poor ice conditions and other environmental changes acting on juvenile mortality and reproductive rates. In 2019, estimated pup production from the model was 1.4 million (95% CI: 1.2–1.5 million), and the total population size was estimated to be 7.6 million (95% CI: 6.6–8.8 million; DFO 2020). The estimated population size in 2019 was slightly higher than in 2012, when the last pupping survey was conducted (Table 1). Sources of uncertainty in the population models include annual reproductive rate data, the level and age structure of various sources of removals, changes in mortality due to varying ice conditions and predicted ice changes in the future and its impact on prey availability (DFO 2020).

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_{est}) and confidence interval (CI).

Year	Area	N_{est}	CI
2014 [a]	Front and Gulf	7.4 million	(95% CI: 6.1–8.7 million)
2019 [b]	Front and Gulf	7.6 million	(95% CI: 6.5 – 8.8 million)

a. The 2014 abundance estimate is based on model projections from the 2012 survey
b. The 2019 abundance estimate is based on model projections from the 2017 survey

Minimum Population Estimate

The minimum population estimate is the lower limit of the two-tailed 60% confidence interval of the log-normally distributed best abundance estimate. This is equivalent to the 20th percentile of the log-normal distribution as specified by Wade and Angliss (1997). The best estimate of abundance for western North Atlantic harp seals, based on the last 2017 survey, is 7.6 million (95% CI: 6.5–8.8 million; DFO 2020). The minimum population is 7.1 million. Data are insufficient to calculate the minimum population estimate for U.S. waters due to low sighting rates.

Current Population Trend

Between 1990 and 2017 harp seal pup production has been variable, reaching a high of 1.6 million (SE=117,900) in 2008 (DFO 2020). Estimated pup production in 2017 was 746,500 (95% CI: 570,300–922,700), almost half the number of pups born in 2008 (DFO 2020). The population model used to estimate total abundance from pup production indicates that the population has been relatively stable since 1995 (Hammill *et al.* 2015), declined in 2010 and 2011, but has increased since then, likely due to reductions in removals and high reproductive rates (DFO 2020). There is large inter-annual variability in reproductive rates due to varying rates of late term abortions which appear to be related to changes in capelin abundance, and mid-winter ice coverage (Buren *et al.* 2014; Lewis *et al*. 2019; Stenson *et al.* 2020b; DFO 2020). In the long term, there is uncertainty as to how the changes in ice formation and capelin biomass will affect the reproductive rates of harp seals.

CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

Current and maximum net productivity rates are unknown for this stock due to limited understanding of stock specific life history parameters in U.S. waters. Therefore, for purposes of this assessment, the maximum net productivity rate was assumed to be 0.12. This value is based on theoretical modeling showing that pinniped populations may not grow at rates much greater than 12% given the constraints of their reproductive life history (Barlow *et al.* 1995).

POTENTIAL BIOLOGICAL REMOVAL

Potential Biological Removal (PBR) is the product of minimum population size, one-half the maximum productivity rate, and a recovery factor (MMPA Sec. 3. 16 U.S.C. 1362; Wade and Angliss 1997). The minimum population size in U.S. waters is unknown. As there is no resident population of harp seals in U.S. waters, PBR for this stock is based on the minimum estimate of abundance in Canadian waters. 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 for increasing populations. PBR for the western North Atlantic harp seal is 426,000.

Table 2. Best and minimum abundance estimates for western North Atlantic harp seals (Pagophilus groenlandicus), with Maximum Productivity Rate (Rmax), Recovery Factor (Fr) and PBR.

N_{est}	CV	N_{min}	F_r	R_{max}	PBR
7.6 million	0.07	7.1 million	1.0	0.12	426,000

ANNUAL HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

For the period 2015–2019 the total estimated annual human caused mortality and serious injury to harp seals was 178,573. This is derived from three components: 1) 86 harp seals (CV=0.16) from the observed U.S. fisheries (Table 3); 2) an average of 1 stranded seal from 2015–2019 that showed signs of non-fishing human interaction as a possible contributor to the mortality; and 3) an average catch of 178,486 seals from 2015–2019 by Canada and Greenland, including bycatch in the lumpfish fishery (Table 4). 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

United States

Detailed fishery information is reported in the Appendix III.

Northeast Sink Gillnet

During 2015–2019, 59 mortalities were observed in the northeast sink gillnet fishery (Hatch and Orphanides 2014, 2015, 2016; Orphanides 2019, 2020). There were no observed injuries of harp seals in the Northeast region during 2015–2019 to assess using new serious injury criteria.

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 Bottom Trawl

Harp seals are rarely observed as bycatch in the Gulf of Maine. A single observed take in 2019 occurred in March in Massachusetts Bay. Fishery-related bycatch rates were estimated using an annual stratified ratio-estimator (Lyssikatos and Chavez-Rosales 2022). See Table 3 for bycatch estimates and observed mortality and serious injury for the current 5-year period, and Appendix V for long-term bycatch information.

Table 3. 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	Est. CVs	Mean Annual Mortality
Northeast Sink Gillnet	2015 2016 2017 2018 2019	Obs. Data, Weighout, Logbooks	0.14 0.10 0.12 0.11 0.13	0 0 0 0 0	12 5 6 2 34	0 0 0 0 0	119 85 44 14 162	119 85 44 14 162	0.34 0.50 0.37 0.8, 0.19	85 (0.16)
Northeast Bottom Trawl	2015 2016 2017 2018 2019	Obs. Data, Weighout, Logbooks	0.19 0.12 0.12 0.12 0.16	0 0 0 0 0	0 0 0 0 1	0 0 0 0 0	0 0 0 0 5.39	0 0 0 0 5.39	na na na na 0.89	1.08 (0.89)
TOTAL										86 (0.16)

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 2015–2019 period and include both at-sea monitor and traditional observer data (Josephson *et al.* 2022).

Other Mortality

United States

From 2015–2019, 363 harp seal stranding mortalities were reported (Table 5; NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 17 November 2020). Nine (2.5%) of the mortalities during this five-year period showed signs of human interaction (2 in 2015, 1 in 2016, 2 in 2017, 0 in 2018 and 4 in 2019), 1 of which with some sign of fishery interaction (2019). One harp seal was reported shot, and in 4 other cases the human interaction could have contributed to the death. 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). After 2005, TACs were set annually to ensure that the population did not decline below a precautionary reference level within a 15 year period (Hammill and Stenson 2007). In 2011, the TAC was raised to 400,000, but no TAC has been announced since 2017. Commercial catches in Canada have remained below 80,000 since 2009 (Table 2b).

Table 4. Summary of the Canadian directed catch and bycatch mortality of Northwest Atlantic harp seal (Pagophilus groenlandicus) by year.

Fishery	2015	2016	2017	2018	2019	Average
Commercial catches[a]	35,382	66,360	81,742	61,022	32,038	55,309
Struck and lost[b]	64,705	67,075	63,686	67,455	63,313	64,733
Greenland subsistence catch[c]	61,767	56,730	48,493	58,614	58,614	56,864
Canadian Arctic[d]	1,000	1,000	1,000	1,000	1,000	1,000
Newfoundland lumpfish[e]	920	518	169	555	541	541
Total	163,774	189,313	195,190	188,646	155,506	178,486

a. ICES 2019
b. Animals that are killed but not recovered and reported. Stenson and Upward 2020.
c. Stenson and Upward 2020
d. Stenson and Upward 2020
e. ICES 2019. Estimates of bycatch in 2019 were not available so the average from 2015–2018 is reported for 2019.

Table 5. Harp seal (Pagophilus groenlandicus) stranding mortalities[a] along the U.S. Atlantic coast (2015–2019) with subtotals of animals recorded as pups in parentheses.

State	2015	2016	2017	2018	2019	Total
Maine	1	4	3	3	3	14
New Hampshire	0	2	0	1	1	4
Massachusetts	17	19 (1)	13 (1)	13	114	176 (2)
Rhode Island	4	3	4	3	20	34
Connecticut	0	1	1	0	12	14
New York	12	1	7	7	59	86
New Jersey	3	1	0	3	8	15
Delaware	0	0	0	2	3	5
Maryland	1	0	0	0	0	1
Virginia	4	1	1	0	0	6

North Carolina	2	2 (1)	2 (1)	0	1	7 (2)
Total	44	34 (2)	31 (2)	32	221	362 (4)
Unspecified seals (all states)	31	13	86	92	80	302

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 below PBR. 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 size of the population relative to fishery removals, it is expected that the uncertainties described above will have little effect on the status of this stock.

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COMMON BOTTLENOSE DOLPHIN (*Tursiops truncatus truncatus*): Northern Gulf of Mexico Continental Shelf Stock

STOCK DEFINITION AND GEOGRAPHIC RANGE

The northern Gulf of Mexico (i.e., U.S. Gulf of Mexico) Continental Shelf Stock of common bottlenose dolphins inhabits waters from 20 to 200 m deep in the northern Gulf from the U.S.-Mexican border to the Florida Keys (Figure 1). Genetically distinct "coastal" and "offshore" ecotypes of bottlenose dolphins (Hoelzel *et al.* 1998; Vollmer 2011) occur in the Gulf of Mexico, and the Continental Shelf Stock, while predominantly of the coastal ecotype, may also include dolphins of the offshore ecotype (Vollmer 2011). The Continental Shelf Stock range may extend into Mexican and Cuban territorial waters; for example, a stranded dolphin from the Florida Panhandle was rehabilitated and released over the shelf off western Florida and traveled into the Atlantic Ocean (Wells *et al.* 1999). However, there are no available estimates of either abundance or mortality from Mexico or Cuba to incorporate in this assessment. Recently, genetic analyses of population structure in coastal, shelf, and oceanic waters of the Gulf of Mexico revealed seven demographically independent populations in the northern Gulf of Mexico, suggesting the current stock designations and boundaries in these waters do not accurately reflect the population structure (Vollmer and Rosel 2017). In continental shelf waters, at least two demographically independent populations were identified, split in the north central Gulf of Mexico (Vollmer and Rosel 2017).

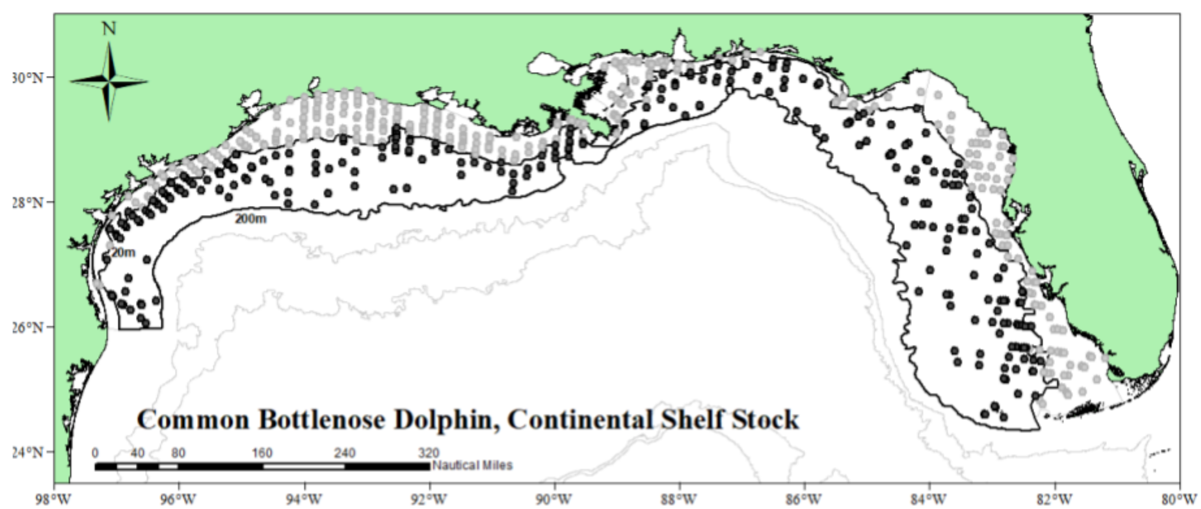

Figure 1. Distribution of common bottlenose dolphin on-effort sightings in coastal (gray circles) and continental shelf (black circles) waters during SEFSC aerial surveys in summer 2017, winter 2018, and fall 2018. Isobaths are the 20-m, 200-m, 1,000-m, and 2,000-m depth contours.

This stock's boundaries abut other bottlenose dolphin stocks, namely the Oceanic Stock and the three coastal stocks. While individuals from different stocks may occasionally overlap, the degree of overlap is unknown and it is not thought that significant mixing or interbreeding occurs between them. Genetic studies have shown significant differentiation between inshore stocks and the adjacent coastal stock (Sellas *et al.* 2005) and among dolphins living in coastal and shelf waters (Vollmer 2011; Vollmer and Rosel 2017). These results suggest that if there is spatial overlap there may be mechanisms reducing interbreeding between the stocks.

POPULATION SIZE

The best abundance estimate available for the northern Gulf of Mexico Continental Shelf Stock of common bottlenose dolphins is 63,280 (CV=0.11; Table 1; Garrison *et al.* 2021). This estimate is from an inverse-variance weighted average of seasonal abundance estimates from aerial surveys conducted during summer 2017 and fall 2018.

Earlier Abundance Estimates

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

Recent Survey and Abundance Estimate

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 summer (June–August) 2017 and fall (October–November) 2018 (Garrison *et al.* 2021). The stock was only partially surveyed during a winter 2018 aerial survey, and therefore this survey was not included in the current abundance estimates (Garrison *et al.* 2021). The surveys were conducted along tracklines oriented perpendicular to the shoreline and spaced 20 km apart. The total survey effort varied during each survey due to weather conditions, and was 10,781 km (fall) and 14,590 km (summer). 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). Abundance was calculated using mark-recapture distance sampling implemented in package mrds (version 2.21; Laake *et al.* 2020) in the R statistical programming language. This approach estimates both the probability of detection on the trackline and within the surveyed strip accounting for the effects of sighting conditions (e.g., sea state, glare, turbidity, and cloud cover). A different detection probability model was used for each seasonal survey (Garrison *et al.* 2021). The survey data were post-stratified into spatial boundaries corresponding to the defined boundaries of common bottlenose dolphin stocks within the surveyed area. The abundance estimates for the Continental Shelf Stock of common bottlenose dolphins were based upon tracklines and sightings in waters from the 20-m to the 200-m isobaths and between the Texas-Mexico border and the Florida Keys. The seasonal abundance estimates for this stock were: summer – 74,959 (CV=0.15) and fall – 52,090 (CV=0.14). Due to the uncertainty in stock movements and apparent seasonal variability in the abundance of the stock, a weighted average of these seasonal estimates was taken where the weighting was the inverse of the CV. This approach weights estimates with higher precision more heavily in the final weighted mean. The resulting weighted mean and best estimate of abundance for the Continental Shelf Stock of common bottlenose dolphins was 63,280 (CV=0.11; Table 1; Garrison *et al.* 2021).

Table 1. Most recent abundance estimate (N_{est}) and coefficient of variation (CV) of the northern Gulf of Mexico Continental Shelf Stock of common bottlenose dolphins (20 – 200-m isobaths) based on season/year aerial surveys.

Years	Area	N_{est}	CV
2017, 2018	Gulf of Mexico	63,280	0.11

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 bottlenose dolphins is 63,280 (CV=0.11). The minimum population estimate for the northern Gulf of Mexico is 57,917 (Table 2).

Current Population Trend

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). Two point estimates of common bottlenose dolphin abundance for the Continental Shelf Stock have been made based on aerial data from surveys during 2011−2012 and 2017−2018 (Garrison *et al.* 2021). Each of these surveys had a similar design and was conducted using the same aircraft and observer configuration. The resulting inverse variance weighted best abundance estimates for seasonal surveys were: 2011–2012 – 48,060 (CV=0.11) and 2017−2018 – 63,280 (CV=0.11). A trends analysis is not possible because there are only two abundance estimates available. For further information on comparisons of old and current abundance estimates for this stock see Garrison *et al.* (2021).

CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

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

POTENTIAL BIOLOGICAL REMOVAL

Potential Biological Removal (PBR) is the product of the minimum population size, one half the maximum net productivity rate and a recovery factor (MMPA Sec. 3.16 U.S.C. 1362; Wade and Angliss 1997). The minimum population size is 57,917. The maximum productivity rate is 0.04, the default value for cetaceans. The recovery factor is 0.48 because the CV of the shrimp trawl mortality estimates is greater than 0.3 (Wade and Angliss 1997). PBR for the Gulf of Mexico Continental Shelf Stock of common bottlenose dolphins is 556 (Table 2).

Table 2. Best and minimum abundance estimates of the northern Gulf of Mexico Continental Shelf Stock of common bottlenose dolphins with Maximum Productivity Rate (R_{max}), Recovery Factor (F_r) and PBR.

N_{est}	N_{est} CV	N_{min}	F_r	R_{max}	PBR
63,280	0.11	57,917	0.48	0.04	556

ANNUAL HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

Total annual fishery-related mortality and serious injury for the Continental Shelf Stock of common bottlenose dolphins during 2015–2019 is unknown because this stock is known to interact with unobserved fisheries (see below). The minimum mean annual fishery-related mortality and serious injury during 2015–2019 was 64 (CV=0.34) based on observer data for the commercial shrimp trawl fishery (Table 3; see Fisheries Information section below), and 0.6 for the commercial reef fish fishery. Mean annual mortality and serious injury during 2015–2019 due to the *Deepwater Horizon* (DWH) oil spill was predicted to be 231 continental shelf dolphins, which includes both Atlantic spotted dolphins and the Continental Shelf Stock of common bottlenose dolphins (see Appendix VI). Therefore, the mean annual mortality and serious injury for the Continental Shelf Stock of common bottlenose dolphins during 2015–2019 due to the DWH oil spill is unknown. Mean annual mortality and serious injury during 2015–2019 due to other human-caused actions (research take in hook and line fishing gear) was 0.2. The minimum total mean annual human-caused mortality and serious injury for this stock during 2015–2019 was, therefore, 65 (Table 3). This is considered a minimum because 1) not all fisheries that could interact with this stock are observed and/or observer coverage is very low, and 2) the population model used to estimate population decline for the northern Gulf of Mexico stocks impacted by the DWH oil spill includes both Atlantic spotted dolphins and common bottlenose dolphins inhabiting the continental shelf and does not estimate mortality and serious injury to common bottlenose dolphins alone. Therefore no estimate for injury has been included for the Continental Shelf Stock of common bottlenose dolphins due to the DWH oil spill.

Fisheries Information

There are four commercial fisheries that interact, or that potentially could interact, with this stock. These include one Category II fishery (Southeastern U.S. Atlantic, Gulf of Mexico shrimp trawl commercial fishery) and three Category III fisheries (Southeastern U.S. Atlantic, Gulf of Mexico shark bottom longline/hook-and-line; Southeastern U.S. Atlantic, Gulf of Mexico, Caribbean snapper-grouper and other reef fish; and Atlantic Ocean, Gulf of Mexico, Caribbean commercial passenger fishing vessel (hook and line). Detailed fishery information is presented in Appendix III.

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

Shrimp Trawl

Between 1997 and 2019, 13 common bottlenose dolphins and nine unidentified dolphins, which could have been either common bottlenose dolphins or Atlantic spotted dolphins, became entangled in the lazy line, turtle excluder device or tickler chain gear in observed trips of the commercial shrimp trawl fishery in the Gulf of Mexico (Soldevilla *et al.* 2021). All dolphin bycatch interactions resulted in mortalities except for one unidentified dolphin that was released alive in 2009 (Maze-Foley and Garrison 2016). Soldevilla *et al.* (2015, 2016, 2021) provided mortality estimates calculated from analysis of shrimp fishery effort data and NMFS's Observer Program bycatch data. Annual mortality estimates were calculated for the years 2015–2019 from stratified annual fishery effort and bycatch rates, and the five-year unweighted mean mortality estimate was calculated for Gulf of Mexico dolphin stocks (Soldevilla *et al.* 2021). The four-area (TX, LA, MS/AL, FL) stratification method was chosen because it best approximates how fisheries operate (Soldevilla *et al.* 2015, 2016, 2021). The mean annual mortality estimate for the continental shelf bottlenose dolphin stock is 64 (CV=0.34). Limitations and biases of annual bycatch mortality estimates are described in detail in Soldevilla *et al.* (2015, 2016, 2021).

Shark Bottom Longline

No interactions between common bottlenose dolphins and this fishery were observed during 2015–2019 (Enzenauer *et al.* 2016; Mathers *et al.* 2017, 2018, 2020, *in press*). The shark bottom longline fishery has been observed since 1994, and three interactions with bottlenose dolphins have been recorded, two of which likely involved the Continental Shelf Stock: one mortality (2003) and one hooked animal that escaped at the vessel (2002; Burgess and Morgan 2003). For the shark bottom longline fishery in the Gulf of Mexico, Richards (2007) estimated common bottlenose dolphin mortalities of 58 (CV=0.99), 0 and 0 for 2003, 2004 and 2005, respectively.

Reef Fish

During 2015–2019, two mortalities and one serious injury were observed in the snapper-grouper and other reef fish fishery. During 2019 a mortality occurred when a dolphin was hooked in the mouth/jaw, and during 2016 a mortality occurred when a dolphin was entangled by its flukes in the mainline of bottom longline gear. During 2018, a serious injury occurred in which a common bottlenose dolphin that was entangled broke the mainline and swam away with the terminal tackle of the bandit (25 hooks and a weight; Maze-Foley and Garrison 2020). All three animals were likely from the Continental Shelf Stock, with the two mortalities occurring off Florida's west coast and the serious injury occurring off Louisiana. In July 2006, NMFS implemented a mandatory observer program for this commercial fishery operating within the U.S. Gulf of Mexico (Scott-Denton *et al.* 2011).

Hook and Line (Rod and Reel)

During 2015–2019, there were no documented interactions between common bottlenose dolphins and this fishery. It is not possible to estimate the total number of interactions with hook and line gear because there is no observer program.

Other Mortality

A population model was developed to estimate the injury and time to recovery for stocks affected by the DWH oil spill, taking into account long-term effects resulting from mortality, reproductive failure, reduced survival rates, and the proportion of the stock exposed to DWH oil (DWH MMIQT 2015). Overall, the model estimated that continental shelf dolphins, which included Atlantic spotted dolphins and the continental shelf stock of common bottlenose dolphins, experienced a 3% maximum reduction in population size due to the oil spill (DWH MMIQT 2015). The mortality projected for the years 2010–2014 due to the spill has not been reported previously. Based on the population model, it was projected that 3,384 continental shelf dolphins died during 2010–2014 (five-year annual average of 677) due to elevated mortality associated with oil exposure (see Appendix VI). For the 2015–2019 reporting period of this SAR, the population model estimated 1,153 continental shelf dolphins died due to elevated mortality associated with oil exposure. The population model used to predict shelf dolphin mortality due to the DWH event has a number of sources of uncertainty. Model parameters (e.g., survival rates, reproductive rates, and life-history parameters) were derived from literature sources for common bottlenose dolphins occupying waters outside of the Gulf of Mexico. In addition, proxy values for the effects of DWH oil exposure on both survival rates and reproductive success were applied based upon estimated values for common bottlenose dolphins in Barataria Bay. Finally, there was no estimation of uncertainty in model parameters or outputs.

During 2017, one animal ascribed to the Continental Shelf Stock was seriously injured due to entanglement in research hook and line fishing gear (Maze-Foley and Garrison 2020). This animal was included in the annual human-caused mortality and serious injury total for this stock (Table 3) and in the stranding database (NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 25 August 2020).

The use of explosives to remove oil rigs in portions of the continental shelf in the western Gulf of Mexico has the potential to cause serious injury or mortality to marine mammals. These activities have been closely monitored by NMFS observers since 1987 (Gitschlag and Herczeg 1994). There were no reports of either serious injury or mortality to common bottlenose dolphins during 2015–2019.

Table 3. Summary of the incidental mortality and serious injury of common bottlenose dolphins (Tursiops truncatus) of the Continental Shelf Stock. For fisheries that do not have an ongoing, federal observer program, 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 strandings, at-sea counts, and fisherman self-reported takes, the number reported is a minimum because not all strandings, at-sea cases, or gear interactions are detected. See the Annual Human-Caused Mortality and Serious Injury section for biases and limitations of mortality estimates, and the Strandings section for limitations of stranding data. 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, MMAP, or Observer Data
Shrimp Trawl	2015–2019	Observer Data	64 (CV=0.34)	NA
Shark Bottom Longline	2015–2019	Observer Data	NA	0
Reef Fish	2015–2019	Observer Data	NA	3
Hook and Line	2015–2019	Stranding Data and At-Sea Observations	NA	0
Mean Annual Mortality due to commercial fisheries (2015–2019)			**64.6**	
Mean Annual Mortality due to research takes, other takes, and DWH (2015–2019)			**0.2**	
Minimum Total Mean Annual Human-Caused Mortality and Serious Injury (2015–2019)			**65**	

Strandings

During 2015–2019, 2,007 common bottlenose dolphins were found stranded in the northern Gulf of Mexico (NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 25 August 2020). Of these, 207 showed evidence of human interactions (e.g., gear entanglement, mutilation, gunshot wounds). It should be noted that evidence of human interaction does not necessarily mean the interaction caused the animal's stranding or death. The vast majority of stranded bottlenose dolphins are assumed to come from stocks that live nearest to land, namely the bay, sound and estuary stocks and the three coastal stocks. Nevertheless, it is possible that some of the stranded bottlenose dolphins belonged to the Continental Shelf or Oceanic Stocks and that they were among those strandings with evidence of human interactions. (Strandings do occur for other cetacean species whose primary range in the Gulf of Mexico is outer continental shelf or oceanic waters.)

An Unusual Mortality Event (UME) was declared for cetaceans in the northern Gulf of Mexico beginning 1 March 2010 and ending 31 July 2014 (Litz *et al.* 2014; https://www.fisheries.noaa.gov/national/marine-life-distress/2010-2014-cetacean-unusual-mortality-event-northern-gulf-mexico). It included cetaceans that stranded prior to the DWH oil spill (see "Habitat Issues" below), during the spill, and after. Exposure to the DWH oil spill was determined to be the primary underlying cause of the elevated stranding numbers in the northern Gulf of Mexico after the spill (e.g., Schwacke *et al.* 2014; Venn-Watson *et al.* 2015; Colegrove *et al.* 2016; DWH NRDAT 2016; see Habitat Issues section). During 2010–2014, 973 common bottlenose dolphins were considered to be part of the UME. The vast majority of stranded common bottlenose dolphins are assumed to belong to one of the coastal stocks or to bay, sound and estuary stocks. Nevertheless, it is possible that some of the stranded common bottlenose dolphins considered part of the UME belonged to the Continental Shelf Stock.

HABITAT ISSUES

The *Deepwater Horizon* MC252 drilling platform, located approximately 80 km southeast of the Mississippi River Delta in waters about 1,500 m deep, exploded on 20 April 2010. The rig sank, and over 87 days 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 Natural Resource Damage Assessment (NRDA) process was initiated under the Oil Pollution Act of 1990. A variety of NRDA research studies were conducted to determine potential impacts of the spill on marine mammals. These studies estimated that 13% (95% CI: 9–19) of continental shelf dolphins, including Atlantic spotted dolphins and the continental shelf stock of common bottlenose dolphins, in the Gulf were exposed to oil, that 6% (95% CI: 3–8) of females suffered from reproductive failure, and 5% (95% CI: 2–7) of continental shelf dolphins suffered adverse health effects (DWH MMIQT 2015). A population model estimated that the stock experienced a 3% maximum reduction in population size (see Other Mortality section above).

STATUS OF STOCK

Common bottlenose dolphins are not listed as threatened or endangered under the Endangered Species Act, and the northern Gulf of Mexico Continental Shelf Stock is not considered strategic under the MMPA. Total U.S. fishery-related 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. The status of bottlenose dolphins, relative to optimum sustainable population, in the northern Gulf of Mexico continental shelf waters is unknown. There are insufficient data to determine population trends for this stock.

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COMMON BOTTLENOSE DOLPHIN (*Tursiops truncatus truncatus*): Gulf of Mexico Eastern Coastal Stock

STOCK DEFINITION AND GEOGRAPHIC RANGE

Common bottlenose dolphins inhabit coastal waters throughout the northern Gulf of Mexico (i.e., U.S. Gulf of Mexico; Mullin *et al.* 1990). As a working hypothesis, it is assumed that the dolphins occupying habitats with dissimilar climatic, coastal and oceanographic characteristics might be restricted in their movements between habitats, and thus constitute separate stocks. Therefore, northern Gulf of Mexico coastal waters have been divided for management purposes into three stock areas: eastern, northern and western, with coastal waters defined as waters between the shore, barrier islands or presumed outer bay boundaries out to the 20-m isobath (Figure 1). The 20-m depth seaward boundary corresponds to survey strata (Scott 1990; Blaylock and Hoggard 1994; Fulling *et al.* 2003), and thus represents a management boundary rather than an ecological boundary. The Eastern Coastal common bottlenose dolphin stock area extends from 84ºW longitude to Key West, Florida. The region is temperate to subtropical in climate, is bordered by a mixture of coastal marshes, sand beaches, marsh and mangrove islands, and has an intermediate level of freshwater input. It is bordered on the north by an extensive area of coastal marsh and marsh islands typical of Florida's Apalachee Bay. Dolphins belonging to this stock are all expected to be of the coastal ecotype (Vollmer 2011). Recently, genetic analyses of population structure in coastal, shelf, and oceanic waters of the Gulf of Mexico revealed seven demographically independent populations in the northern Gulf of Mexico, suggesting the current stock designations and boundaries in these waters do not accurately reflect the population structure (Vollmer and Rosel 2017). Sampling within the range of the Eastern Coastal Stock was very limited and further work is necessary to determine the boundaries of these demographically independent populations.

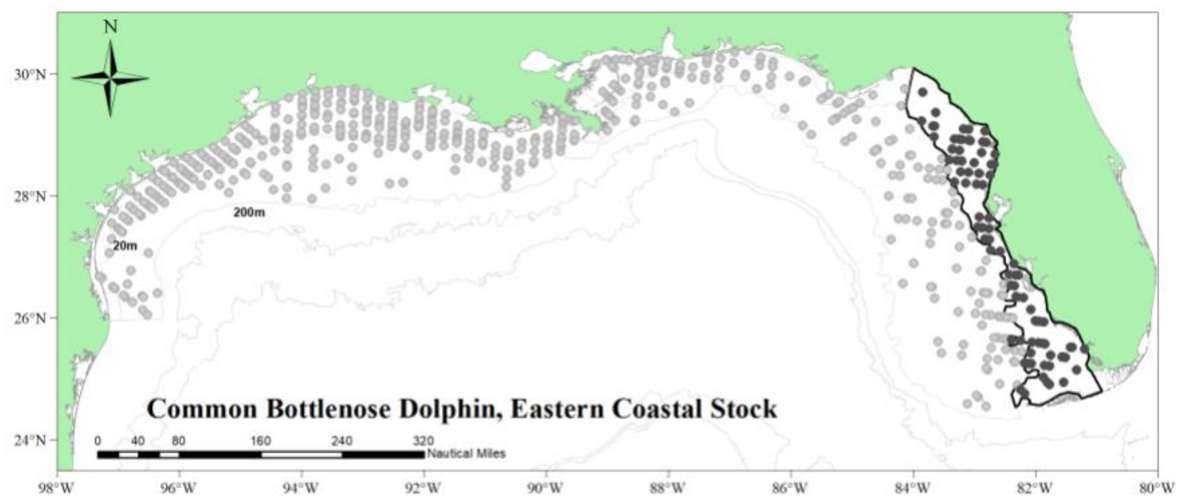

Figure 1. Distribution of common bottlenose dolphin on-effort sightings in coastal and continental shelf waters during SEFSC aerial surveys in summer 2017, winter 2018, and fall 2018. Sightings within the boundaries of the Eastern Coastal Stock are denoted by the black circles. Isobaths are the 20-m, 200-m, 1,000-m, and 2,000-m depth contours.

This stock's boundaries abut other common bottlenose dolphin stocks, namely the Continental Shelf Stock, the Northern Coastal Stock and several bay, sound and estuary stocks, and while individuals from different stocks may occasionally overlap, it is not thought that significant mixing or interbreeding occurs between them. Fazioli *et al.* (2006) conducted photo-identification surveys of coastal waters off Tampa Bay, Sarasota Bay and Lemon Bay, Florida, over 14 months. They found both 'inshore' and 'Gulf' dolphins inhabited coastal waters but the two types used coastal waters differently. Dolphins from the inshore communities were observed occasionally in Gulf near-shore waters adjacent to their inshore range, whereas 'Gulf' dolphins were found primarily in open Gulf of Mexico waters with some displaying seasonal variations in their use of the study area. The 'Gulf' dolphins did not show a preference for waters near passes as was seen for 'inshore' dolphins, but moved throughout the study area and made greater use of waters offshore of waters used by 'inshore' dolphins. During winter months abundance of 'Gulf' groups decreased while abundance for 'inshore' groups increased. These findings support an earlier report by Irvine *et al.* (1981) of increased use of pass and coastal waters by Sarasota Bay dolphins in winter. Seasonal movements of identified individuals and abundance indices suggested that part of the 'Gulf' dolphin community moved out of the study area during winter, but their destination is unknown (Fazioli *et al.* 2006). In a follow-up study, Sellas *et al.* (2005) examined genetic population subdivision in the study area of Fazioli *et al.* (2006), and found evidence of significant population structure among all areas. Rosel *et al.* (2017) also identified significant genetic differentiation between estuarine residents of Barataria Bay and the adjacent coastal stock, further supporting separation of coastal and estuarine stocks.

Finally, off Galveston, Texas, Beier (2001) reported an open population of individual dolphins in coastal waters, but several individual dolphins had been sighted previously by other researchers over a 10-year period. Some coastal animals may move relatively long distances alongshore. Two bottlenose dolphins previously seen in the South Padre Island area in Texas were seen in Matagorda Bay, 285 km north, in May 1992 and May 1993 (Lynn and Würsig 2002).

POPULATION SIZE

The best abundance estimate available for the northern Gulf of Mexico Eastern Coastal Stock of common bottlenose dolphins is 16,407 (CV=0.17; Table 1; Garrison *et al.* 2021). This estimate is from an inverse-variance weighted average of seasonal abundance estimates from aerial surveys conducted during summer 2017 and fall 2018.

Earlier Abundance Estimates

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

Recent Surveys and Abundance Estimates

The Southeast Fisheries Science Center conducted aerial surveys of continental shelf waters (shoreline to 200 m depth) along the U.S. Gulf of Mexico coast from the Florida Keys to the Texas/Mexico border during summer (June–August) 2017 and fall (October–November) 2018 (Garrison *et al.* 2021). The stock was only partially surveyed during a winter 2018 aerial survey, and therefore this survey was not included in the current abundance estimates (Garrison *et al.* 2021). The surveys were conducted along tracklines oriented perpendicular to the shoreline and spaced 20 km apart. The total survey effort varied during each survey due to weather conditions, and was 10,781 km (fall) and 14,590 km (summer). 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). Abundance was calculated using mark-recapture distance sampling implemented in package mrds (version 2.21; Laake *et al.* 2020) in the R statistical programming language. This approach estimates both the probability of detection on the trackline and within the surveyed strip accounting for the effects of sighting conditions (e.g., sea state, glare, turbidity, and cloud cover). A different detection probability model was used for each seasonal survey (Garrison *et al.* 2021). The survey data were post-stratified into spatial boundaries corresponding to the defined boundaries of common bottlenose dolphin stocks within the surveyed area. The abundance estimates for the Eastern Coastal Stock of common bottlenose dolphins were based upon tracklines and sightings in waters from the shoreline to the 20-m isobath and between 84°W longitude and the Florida Keys. The seasonal abundance estimates for this stock were: summer – 11,482 (CV=0.23) and fall – 21,386 (CV=0.24). Due to the uncertainty in stock movements and apparent seasonal variability in the abundance of the stock, a weighted average of these seasonal estimates was taken where the weighting was the inverse of the CV. This approach weights estimates with higher precision more heavily in the final weighted mean. The resulting weighted mean and best estimate of abundance for the Eastern Coastal Stock of common bottlenose dolphins was 16,407 (CV=0.17; Table 1; Garrison *et al.* 2021).

Table 1. Most recent abundance estimate (N_{est}) and coefficient of variation (CV) of the northern Gulf of Mexico Eastern Coastal Stock of common bottlenose dolphins (0–20-m isobaths) based on summer 2017, winter 2018, and fall 2018 aerial surveys.

Years	Area	N_{est}	CV
2017, 2018	Gulf of Mexico	16,407	0.17

Minimum Population Estimate

The minimum population estimate is the lower limit of the two-tailed 60% confidence interval of the log-normally distributed 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 Eastern Coastal Stock of common bottlenose dolphins is 16,407 (CV=0.17). The minimum population estimate for the northern Gulf of Mexico Eastern Coastal Stock is 14,199 common bottlenose dolphins (Table 2).

Current Population Trend

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). Two point estimates of common bottlenose dolphin abundance for the Eastern Coastal Stock have been made based on aerial data from surveys during 2011−2012 and 2017−2018 (Garrison *et al.* 2021). Each of these surveys had a similar design and was conducted using the same aircraft. The resulting inverse variance weighted best abundance estimates for seasonal surveys were: 2011–2012 – 12,181 (CV=0.14) and 2017−2018 – 16,407 (CV=0.17). A trends analysis is not possible because there are only two abundance estimates available. For further information on comparisons of old and current abundance estimates for this stock see Garrison *et al.* (2021).

CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

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

POTENTIAL BIOLOGICAL REMOVAL

Potential Biological Removal (PBR) is the product of minimum population size, one-half the maximum productivity rate and a recovery factor (Wade and Angliss 1997). The minimum population size is 14,199. 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 is greater than 0.8 (Wade and Angliss 1997). PBR for the northern Gulf of Mexico Eastern Coastal Stock of common bottlenose dolphins is 114 (Table 2).

Table 2. Best and minimum abundance estimates of the northern Gulf of Mexico Eastern Coastal Stock of common bottlenose dolphins with Maximum Productivity Rate (R_{max}), Recovery Factor (F_r) and PBR.

N_{est}	N_{est} CV	N_{min}	F_r	R_{max}	PBR
16,407	0.17	14,199	0.4	0.04	114

ANNUAL HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

The total annual human-caused mortality and serious injury for the Eastern Coastal Stock of common bottlenose dolphins during 2015–2019 is unknown because this stock is known to interact with unobserved fisheries (see below). The five-year unweighted mean annual mortality estimate for 2015–2019 for the commercial shrimp trawl fishery was 7.6 (CV=1.05; see Shrimp Trawl section below). The mean annual fishery-related mortality and serious injury during 2015–2019 for other observations identified as fishery-caused was 1.2. Additional mortality or serious injury documented from other human-caused actions was 0.4. The minimum total mean annual human-caused mortality and serious injury for this stock during 2015–2019 was 9.2 (Table 3). This is considered 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, and 4) the estimate of fishery-related interactions includes an actual count of verified fishery-caused deaths and serious injuries and should be considered a minimum (NMFS 2016).

Fisheries Information

There are eight commercial fisheries that interact, or that potentially could interact, with this stock. These include three Category II fisheries (Southeastern U.S. Atlantic, Gulf of Mexico shrimp trawl; Gulf of Mexico gillnet; and

Southeastern U.S. Atlantic, Gulf of Mexico stone crab trap/pot); and five Category III fisheries (Southeastern U.S. Atlantic, Gulf of Mexico shark bottom longline/hook-and-line; Florida spiny lobster trap/pot; Gulf of Mexico blue crab trap/pot; Florida West Coast sardine purse seine; and Atlantic Ocean, Gulf of Mexico, Caribbean commercial passenger fishing vessel (hook and line)). Detailed fishery information is presented in Appendix III.

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

Shrimp Trawl

Between 1997 and 2019, 13 common bottlenose dolphins and nine unidentified dolphins, which could have been either common bottlenose dolphins or Atlantic spotted dolphins, became entangled in the lazy line, turtle excluder device or tickler chain gear in observed trips of the commercial shrimp trawl fishery in the Gulf of Mexico (Soldevilla *et al.* 2021). All dolphin bycatch interactions resulted in mortalities except for one unidentified dolphin that was released alive in 2009 (Maze-Foley and Garrison 2016). Soldevilla *et al.* (2015, 2016, 2021) provided mortality estimates calculated from analysis of shrimp fishery effort data and NMFS's Observer Program bycatch data. Annual mortality estimates were calculated for the years 2015–2019 from stratified annual fishery effort and bycatch rates, and the five-year unweighted mean mortality estimate was calculated for Gulf of Mexico dolphin stocks (Soldevilla *et al.* 2021). The four-area (TX, LA, MS/AL, FL) stratification method was chosen because it best approximates how fisheries operate (Soldevilla *et al.* 2015, 2016, 2021). The mean annual mortality estimate for the Eastern Coastal Stock of common bottlenose dolphins is 7.6 (CV=1.05). Limitations and biases of annual bycatch mortality estimates are described in detail in Soldevilla *et al.* (2015, 2016, 2021).

Gillnet

During 2015–2019, there was one interaction observed between the Gulf of Mexico gillnet fishery and the Eastern Coastal Stock. During 2015, one animal was entangled and released alive without serious injury from a sink gillnet targeting Spanish mackerel (Mathers *et al.* 2016; Maze-Foley and Garrison 2020). Gillnet fishing is prohibited in Florida state waters, so there is no observer coverage of this fishery in state waters; however, there is limited observer coverage of this fishery in federal waters (e.g., Mathers *et al.* 2020). The documented interaction in this gear represents a minimum known count of interactions in the last five years.

Blue Crab, Stone Crab, and Spiny Lobster Trap/Pot

During 2015–2019, one entanglement associated with trap/pot fisheries was documented for the Eastern Coastal Stock. In 2018, one animal was disentangled from commercial stone crab trap/pot gear and released alive. It could not be determined if the animal was seriously injured following mitigation efforts (the initial determination was seriously injured; Maze-Foley and Garrison 2020). This live entanglement was included in the stranding database (NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 25 August 2020) and in the stranding totals presented in Table 4, but it was not included in the annual human-caused mortality and serious injury total for this stock (Table 3).

In addition to animals included in the stranding database, during 2015–2019, there was one at-sea observation in the Eastern Coastal Stock area (in 2017) of a live common bottlenose dolphin entangled in trap/pot gear, and this animal was considered seriously injured (Maze-Foley and Garrison 2020). This serious injury was included in the annual human-caused mortality and serious injury total for this stock (Table 3).

Since there is no observer program, it is not possible to estimate the total number of interactions or mortalities associated with these trap/pot fisheries. The documented interactions in this gear represent a minimum known count of interactions in the last five years.

Shark Bottom Longline

During 2015–2019, no interactions between common bottlenose dolphins and this fishery were observed (Enzenauer *et al.* 2016; Mathers *et al.* 2017, 2018, 2020, *in press*). The shark bottom longline fishery has been observed since 1994, and three interactions with bottlenose dolphins have been recorded, one of which likely involved the Eastern Coastal Stock: in 1999, a hooked dolphin escaped at the vessel (Burgess and Morgan 2003). For the shark bottom longline fishery in the Gulf of Mexico, Richards (2007) estimated common bottlenose dolphin mortalities of

58 (CV=0.99), 0 and 0 for 2003, 2004 and 2005, respectively.

Florida West Coast Sardine Purse Seine

There have been no documented interactions between common bottlenose dolphins of the Eastern Coastal Stock and the Florida West Coast sardine purse seine fishery; however, it should be noted there is no observer coverage of the sardine purse seine fishery. Without an observer program, it is not possible to estimate the total number of interactions or mortalities associated with this gear.

Hook and Line (Rod and Reel)

During 2015–2019, five mortalities and one live release without serious injury involving hook and line gear entanglement or ingestion were documented. The mortalities occurred in 2015 (n=1), 2018 (n=1), and 2019 (n=3). For two of the five mortalities, available evidence from the stranding data suggested the hook and line gear interaction contributed to the cause of death. For two mortalities, available evidence suggested the gear interaction did not contribute to cause of death, and for the remaining mortality, it could not be determined if the gear contributed to cause of death. During 2015, one animal was released alive. For the live animal, it was initially seriously injured, but due to mitigation efforts, was released without serious injury (Maze-Foley and Garrison 2020). All six cases were included in the stranding database (NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 25 August 2020) and are included in the stranding totals presented in Table 4. The two mortalities for which evidence suggested the gear contributed to cause of death were included in the annual human-caused mortality and serious injury total for this stock (Table 3).

In addition to animals included in the stranding database, during 2015–2019, there were three at-sea observations in the Eastern Coastal Stock area of live common bottlenose dolphins entangled in hook and line fishing gear. In two cases, the animals were considered seriously injured (2015, 2018), and for the remaining case (2018), it could not be determined if the animal was seriously injured (Maze-Foley and Garrison 2020). The two serious injuries were included in the annual human-caused mortality and serious injury total for this stock (Table 3).

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 observer program. The documented interactions in this gear represent a minimum known count of interactions in the last five years.

Other Mortality

In addition to animals included in the stranding database and those mentioned above, during 2015–2019 in the Eastern Coastal Stock area, there were six at-sea observations of common bottlenose dolphins entangled in unidentified rope, unidentified line, a mesh net, and a cast net, an animal reported to be anchored/tethered, and an animal that was encircled in a net by the public. Two of these animals were considered seriously injured, and for the remaining four animals, it could not be determined whether they were seriously injured (Maze-Foley and Garrison 2020). The two serious injuries were included in the annual human-caused mortality and serious injury total for this stock (Table 3).

Depredation of fishing catch and/or bait is a growing problem in Gulf of Mexico coastal and estuarine waters and globally, and can lead to serious injury or mortality via ingestion of or entanglement in gear (e.g., Zollett and Read 2006; Read 2008; Powell and Wells 2011; Vail 2016), as well as changes in dolphin activity patterns, such as decreases in natural foraging (Powell and Wells 2011). It has been suggested that provisioning, or the illegal feeding, of wild common bottlenose dolphins, may encourage depredation because provisioning conditions dolphins to approach humans and vessels, where they then may prey on bait and catches (Vail 2016). Illegal feeding/provisioning has been documented in the literature in Florida and Texas (Bryant 1994; Samuels and Bejder 2004; Cunningham-Smith *et al.* 2006; Powell and Wells 2011; Powell *et al.* 2018). Such conditioning increases risks of subsequent injury or mortality (Christiansen *et al.* 2016).

Feeding or provisioning of wild common bottlenose dolphins has been documented in Florida, particularly near Panama City Beach in the Panhandle (Samuels and Bejder 2004) and south of Sarasota Bay (Cunningham-Smith *et al.* 2006; Powell and Wells 2011), and also in Texas near Corpus Christi (Bryant 1994). Feeding wild dolphins is defined under the MMPA as a form of 'take' because it can alter their natural behavior and increase their risk of injury or death. There are emerging questions regarding potential linkages between provisioning and depredation of recreational fishing gear and associated entanglement and ingestion of gear, which is increasing through much of

Florida. During 2006, an estimated 2% of the long-term resident dolphins of Sarasota Bay, immediately inshore of the Eastern Coastal Stock, died from ingestion of recreational fishing gear (Powell and Wells 2011).

Swimming with wild common bottlenose dolphins has also been documented in Florida, including Key West (Samuels and Engleby 2007) and Panama City Beach (Samuels and Bejder 2004), but to date, there are no records for this stock area.

All mortalities and serious injuries from known sources for the Eastern Coastal Stock are summarized in Table 3.

Table 3. Summary of the incidental mortality and serious injury of common bottlenose dolphins (Tursiops truncatus) of the Eastern Coastal Stock. For fisheries that do not have an ongoing, federal observer program, 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 strandings, at-sea counts, and fisherman self-reported takes, the number reported is a minimum because not all strandings, at-sea cases, or gear interactions are detected. See the Annual Human-Caused Mortality and Serious Injury section for biases and limitations of mortality estimates, and the Strandings section for limitations of stranding data. NA = not applicable. *Indicates the count would have been higher (5 instead of 4) had it not been for mitigation efforts (see text for that specific fishery for further details).

Fishery	Years	Data Type	Mean Annual Estimated Mortality and Serious Injury Based on Observer Data	5-year Minimum Count Based on Stranding, At-Sea, MMAP, and/or Observer Data
Shrimp Trawl	2015–2019	Observer Data	7.6 (CV=1.05)	NA
Gillnet	2015–2019	Observer Data (minimum count only, no estimate available)	NA	1
Crab Trap/Pot	2015–2019	Stranding Data	NA	1
Shark Bottom Longline	2015–2019	Observer Data	0	NA
Florida West Coast Sardine Purse Seine	2015–2019	Stranding Data and MMAP Data	NA	0
Hook and Line	2015–2019	Stranding Data and At-Sea Observations	NA	4*
Mean Annual Mortality due to commercial fisheries (2015–2019)			**8.8**	
Mean Annual Mortality due to other takes (2015–2019)			**0.4**	
Minimum Total Mean Annual Human-Caused Mortality and Serious Injury (2015–2019)			**9.2**	

Strandings

During 2015–2019, 154 common bottlenose dolphins were found stranded in Eastern Coastal waters of the northern Gulf of Mexico (Table 4; NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 25 August 2020). There was evidence of human interaction (HI) for 16 of the strandings. No evidence of human interaction was detected for three strandings, and for the remaining 135 strandings, it could not be determined if there was evidence of human interaction. Human interactions were from several sources, including six entanglements with hook and line gear, one entanglement with commercial stone crab trap/pot gear, and two animals with evidence of a vessel strike (Table 4). It should be noted that evidence of human interaction does not necessarily mean the interaction caused the animal's stranding or death.

The assignment of animals to a single stock is impossible in some regions where stocks overlap, especially in nearshore coastal waters (Maze-Foley *et al.* 2019). Of the 154 strandings ascribed to the Eastern Coastal Stock, 149 were ascribed solely to this stock. The counts in Table 4 may include some animals from the St. Joseph Sound, Clearwater Harbor Stock or Tampa Bay Stock and thereby overestimate the number of strandings for the Eastern Coastal Stock. 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).

There are a number of other difficulties associated with the interpretation of stranding data. Stranding data underestimate the extent of human and fishery-related mortality and serious injury because not all of the dolphins that die or are seriously injured in human interactions wash ashore, or, if they do, they are not all recovered (Peltier *et al*. 2012; Wells *et al*. 2015; Carretta *et al.* 2016). Additionally, not all carcasses will show evidence of human interaction, entanglement or other fishery-related interaction due to decomposition, scavenger damage, etc. (Byrd *et al*. 2014). Finally, the level of technical expertise among stranding network personnel varies widely as does the ability to recognize signs of human interaction.

Since 1990, there have been 15 common bottlenose dolphin die-offs or Unusual Mortality Events (UMEs) in the northern Gulf of Mexico (http://www.nmfs.noaa.gov/pr/health/mmume/events.html, accessed 5 November 2020), and 5 of these have occurred within the boundaries of the Eastern Coastal Stock and may have affected the stock. 1) From January through May 1990, a total of 344 bottlenose dolphins stranded in the northern Gulf of Mexico. Overall this represented a two-fold increase in the prior maximum recorded 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). 2) An unusual mortality event was declared for Sarasota Bay, Florida, in 1991involving 31 bottlenose dolphins. The cause was not determined, but it is believed biotoxins may have contributed to this event (Litz *et al.* 2014). 3) In 2005, a particularly destructive red tide (*Karenia brevis*) bloom occurred off of central west Florida. Manatee, sea turtle, bird and fish mortalities were reported in the area in early 2005 and a manatee UME had been declared. Dolphin mortalities began to rise above the historical averages by late July 2005, continued to increase through October 2005, and were then declared to be part of a multi-species UME. The multi-species UME extended into 2006, and ended in November 2006. In total, 190 dolphins were involved, primarily bottlenose dolphins (plus strandings of 1 Atlantic spotted dolphin, *S. frontalis*, and 23 unidentified dolphins). The evidence suggests the effects of a red tide bloom contributed to the cause of this event (Litz *et al.* 2014). 4) A common bottlenose dolphin UME occurred in southwest Florida from 1 July 2018 through 30 June 2019, with peak strandings occurring between 1 July 2018 and 30 April 2019. In total, 183 dolphins were reported (note the dates and numbers are subject to change as the closure package has not yet been approved by the UME Working Group). All age classes of dolphins were represented and the majority of the animals recovered were in moderate to advanced stages of decomposition. The cause of the bottlenose dolphin UME was determined to be due to biotoxin exposure from the *K. brevis* harmful algal bloom off the coast of southwest Florida. The additional supporting evidence of fish kills and other species die-offs linked to brevetoxin during the same time and space support that the impacts of the harmful algal bloom caused the dolphin mortalities. 5) During 1 February 2019 to 30 November 2019, a UME was declared for the area from the eastern border of Taylor County, Florida, west through Alabama, Mississippi, and Louisiana (http://www.nmfs.noaa.gov/pr/health/mmume/cetacean_gulfofmexico.htm, accessed 5 November 2020). No strandings were reported within the Eastern Coastal Stock range during this event.

Table 4. Common bottlenose dolphin strandings occurring in the Eastern Coastal Stock area from 2015 to 2019, 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 25 August 2020). Please note HI does not necessarily mean the interaction caused the animal's death.

Stock	Category	2015	2016	2017	2018	2019	Total
Eastern Coastal Stock	Total Stranded	13	15	8	96	22	154

	Human Interaction						
	---Yes	5[a]	2[b]	0	5[c]	4[d]	16
	---No	0	1	1	1	0	3
	---CBD	8	12	7	90	18	135

a. Includes 3 fisheries interactions (FIs), 2 of which were entanglement interactions with hook and line gear (1 mortality, 1 released alive without serious injury), and 1 mortality with evidence of a vessel strike.
b. Includes 1 mortality with evidence of a vessel strike and 1 FI (mortality).
c. Includes 3 FIs, 1 of which was an entanglement interaction with hook and line gear (mortality) and 1 was an entanglement interaction with commercial stone crab trap/pot gear (released alive, CBD if seriously injured).
d. Includes 3 FIs, all of which were entanglement interactions with hook and line gear (mortalities).

HABITAT ISSUES

The *Deepwater Horizon* (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). Because the range of the Eastern Coastal Stock of common bottlenose dolphins does not extend west of 84°W longitude, this stock is not thought to have experienced oil exposure due to the DWH event.

The nearshore habitat occupied by the three coastal stocks is adjacent to areas of high human population and in some areas, such as Tampa Bay, Florida, Galveston, Texas, and Mobile, Alabama, is highly industrialized. Concentrations of anthropogenic chemicals such PCBs and DDT and its metabolites vary from site to site, and can reach levels of concern for bottlenose dolphin health and reproduction in the southeastern U.S. (Schwacke *et al.* 2002). PCB concentrations in three stranded dolphins sampled from the Eastern Coastal Stock area ranged from 16-46µg/g wet weight. Two stranded dolphins from the Northern Coastal Stock area had the highest levels of DDT derivatives of any of the bottlenose dolphin liver samples analyzed in conjunction with a 1990 mortality investigation conducted by NMFS (Varanasi *et al.* 1992). The significance of these findings is unclear, but there is some evidence that increased exposure to anthropogenic compounds may reduce immune function in bottlenose dolphins (Lahvis *et al.* 1995), or impact reproduction through increased first-born calf mortality (Wells *et al.* 2005).

STATUS OF STOCK

The common bottlenose dolphin is not listed as threatened or endangered under the Endangered Species Act, and the Eastern Coastal Stock is not considered strategic under the MMPA. Total U.S. fishery-related mortality and serious injury for this stock is unknown. The minimum estimate of fishery-related mortality and serious injury is less than 10% of PBR, but there is insufficient information (see Annual Human-Caused Mortality and Serious Injury section) available to determine whether the total fishery-related mortality and serious injury is insignificant and approaching the zero mortality and serious injury rate. The status of this stock relative to optimum sustainable population in the Gulf of Mexico EEZ is unknown. There are insufficient data to determine the population trends for this stock.

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COMMON BOTTLENOSE DOLPHIN (*Tursiops truncatus truncatus*): Gulf of Mexico Northern Coastal Stock

STOCK DEFINITION AND GEOGRAPHIC RANGE

Common bottlenose dolphins inhabit coastal waters throughout the northern Gulf of Mexico (i.e., U.S. Gulf of Mexico; Mullin *et al.* 1990). As a working hypothesis, it is assumed that the dolphins occupying habitats with dissimilar climatic, coastal and oceanographic characteristics might be restricted in their movements between habitats, and thus constitute separate stocks. Therefore, northern Gulf of Mexico coastal waters have been divided for management purposes into three stock areas: eastern, northern and western, with coastal waters defined as waters between the shore, barrier islands or presumed outer bay boundaries out to the 20-m isobath (Figure 1). The 20-m depth seaward boundary corresponds to survey strata (Scott 1990; Blaylock and Hoggard 1994; Fulling *et al.* 2003), and thus represents a management boundary rather than an ecological boundary. The Northern Coastal common bottlenose dolphin stock area extends from 84ºW longitude to the Mississippi River Delta. This region is characterized by a temperate climate, barrier islands, sand beaches, coastal marshes and marsh islands, and has a relatively high level of freshwater input. It is bordered on the east by an extensive area of coastal marsh and marsh islands typical of Florida's Apalachee Bay. Dolphins belonging to this stock are all expected to be of the coastal ecotype (Vollmer 2011). Recently, genetic analyses of population structure in coastal, shelf, and oceanic waters of the Gulf of Mexico revealed seven demographically independent populations in the northern Gulf of Mexico, suggesting the current stock designations and boundaries in these waters do not accurately reflect the population structure (Vollmer and Rosel 2017). Sampling within the range of the Northern Coastal Stock was limited and further work is necessary to determine the boundaries of these demographically independent populations.

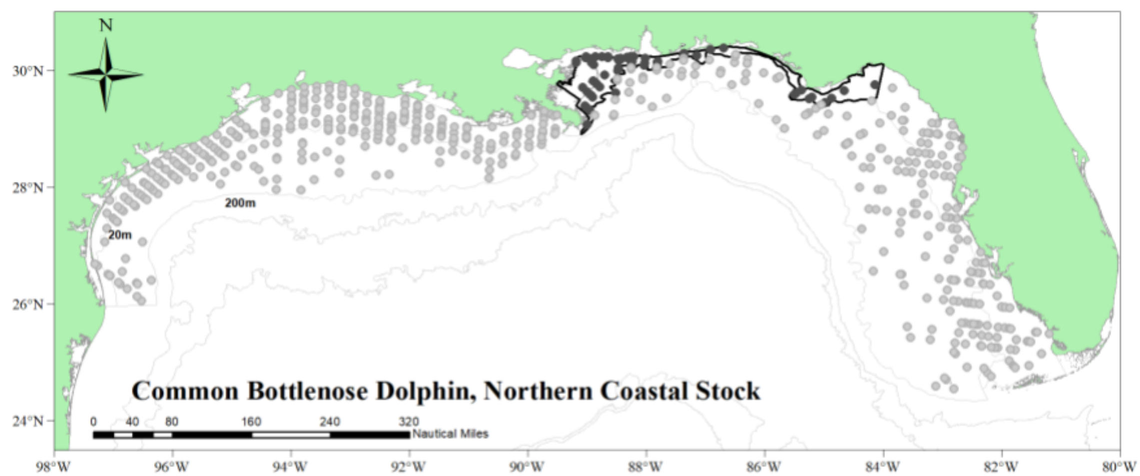

Figure 1. Distribution of common bottlenose dolphin on-effort sightings in coastal and continental shelf waters during SEFSC aerial surveys in summer 2017, winter 2018, and fall 2018. Sightings within the boundaries of the Northern Coastal Stock are denoted by the black circles. Isobaths are the 20-m, 200-m, 1,000-m, and 2,000-m depth contours.

This stock's boundaries abut other common bottlenose dolphin stocks, namely the Continental Shelf Stock, the Eastern and Western Coastal Stocks, and several bay, sound and estuary stocks in Louisiana, Mississippi, Alabama and Florida, and while individuals from different stocks may occasionally overlap, it is not thought that significant mixing or interbreeding occurs between them. Fazioli *et al.* (2006) conducted photo-identification surveys of coastal waters off Tampa Bay, Sarasota Bay and Lemon Bay, Florida, over 14 months. They found both 'inshore' and 'Gulf' dolphins inhabited coastal waters but the two types used coastal waters differently. Dolphins from the inshore communities were observed occasionally in Gulf near-shore waters adjacent to their inshore range, whereas ‘Gulf’ dolphins were found primarily in open Gulf of Mexico waters with some displaying seasonal variations in their use of the study area. The ‘Gulf’ dolphins did not show a preference for waters near passes as was seen for ‘inshore’ dolphins, but moved throughout the study area and made greater use of waters offshore of waters used by ‘inshore’ dolphins. During winter months abundance of ‘Gulf’ groups decreased while abundance for ‘inshore’ groups increased. These findings support an earlier report by Irvine *et al.* (1981) of increased use of pass and coastal waters by Sarasota Bay dolphins in winter. Seasonal movements of identified individuals and abundance indices suggested that part of the ‘Gulf’ dolphin community moved out of the study area during winter, but their destination is unknown (Fazioli *et al.* 2006). In a follow-up study, Sellas *et al.* (2005) examined genetic population subdivision in the study area of Fazioli *et al.* (2006), and found evidence of significant population structure among all areas. Rosel *et al.* (2017) also identified significant genetic differentiation between estuarine residents of Barataria Bay and the adjacent coastal stock, further supporting separation of coastal and estuarine stocks. Finally, off Galveston, Texas, Beier (2001) reported an open population of individual dolphins in coastal waters, but several individual dolphins had been sighted previously by other researchers over a 10-year period. Some coastal animals may move relatively long distances alongshore. Two bottlenose dolphins previously seen in the South Padre Island area in Texas were seen in Matagorda Bay, 285 km north, in May 1992 and May 1993 (Lynn and Würsig 2002).

POPULATION SIZE

The best abundance estimate available for the northern Gulf of Mexico Northern Coastal Stock of common bottlenose dolphins is 11,543 (CV=0.19; Table 1; Garrison *et al.* 2021). This estimate is from an inverse-variance weighted average of seasonal abundance estimates from aerial surveys conducted during summer 2017, winter 2018, and fall 2018.

Earlier Abundance Estimates

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

Recent Surveys and Abundance Estimates

The Southeast Fisheries Science Center conducted aerial surveys of continental shelf waters (shoreline to 200 m depth) along the U.S. Gulf of Mexico coast from the Florida Keys to the Texas/Mexico border during summer (June–August) 2017 and fall (October–November) 2018, and from Tampa, Florida, to Port O’Connor, Texas, during winter (January–March) 2018 (Garrison *et al.* 2021). The surveys were conducted along tracklines oriented perpendicular to the shoreline and spaced 20 km apart. The total survey effort varied during each survey due to weather conditions, but ranged between 8,046 and 14,590 km. 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). Abundance was calculated using mark-recapture distance sampling implemented in package mrds (version 2.21; Laake *et al.* 2020) in the R statistical programming language. This approach estimates both the probability of detection on the trackline and within the surveyed strip accounting for the effects of sighting conditions (e.g., sea state, glare, turbidity, and cloud cover). A different detection probability model was used for each seasonal survey (Garrison *et al.* 2021). The survey data were post-stratified into spatial boundaries corresponding to the defined boundaries of common bottlenose dolphin stocks within the surveyed area. The abundance estimates for the Northern Coastal Stock of common bottlenose dolphins were based upon tracklines and sightings in waters from the shoreline to the 20-m isobath and between the Mississippi River Delta and 84°W longitude. The seasonal abundance estimates for this stock were: summer – 4,671 (CV=0.49), winter – 18,194 (CV=0.24), and fall – 7,152 (CV=0.32). Due to the uncertainty in stock movements and apparent seasonal variability in the abundance of the stock, a weighted average of these seasonal estimates was taken where the weighting was the inverse of the CV. This approach weights estimates with higher precision more heavily in the final weighted mean. The resulting weighted mean and best estimate of abundance for the Northern Coastal Stock of common bottlenose dolphins was 11,543 (CV=0.19; Table 1; Garrison *et al.* 2021).

Table 1. Most recent abundance estimate (N_{est}) and coefficient of variation (CV) of the northern Gulf of Mexico Northern Coastal Stock of common bottlenose dolphins (0 – 20-m isobaths) based on summer 2017, winter 2018, and fall 2018 aerial surveys.

Years	Area	N_{est}	CV
2017, 2018	Gulf of Mexico	11,543	0.19

Minimum Population Estimate

The minimum population estimate is the lower limit of the two-tailed 60% confidence interval of the log-normally distributed 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 Northern Coastal Stock of common bottlenose dolphins is 11,543 (CV=0.19). The minimum population estimate for the Northern Coastal Stock is 9,881 common bottlenose dolphins (Table 2).

Current Population Trend

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). Two point estimates of common bottlenose dolphin abundance for the Northern Coastal Stock have been made based on aerial data from surveys during 2011−2012 and 2017−2018 (Garrison *et al.* 2021). Each of these surveys had a similar design and was conducted using the same aircraft. The model for detection probability on the trackline from the 2017/2018 survey was applied to the abundance estimates from the 2011 and 2012 surveys. The resulting inverse variance weighted best abundance estimates for seasonal surveys were: 2011–2012 – 7,569 (CV=0.22) and 2017−2018 – 11,543 (CV=0.19). A trends analysis is not possible because there are only two abundance estimates available. For further information on comparisons of old and current abundance estimates for this stock see Garrison *et al.* (2021).

CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

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

POTENTIAL BIOLOGICAL REMOVAL

Potential Biological Removal (PBR) is the product of minimum population size, one-half the maximum productivity rate and a recovery factor (Wade and Angliss 1997). The minimum population size is 9,881. The maximum productivity rate is 0.04, the default value for cetaceans. The recovery factor is 0.45 because the CV of the shrimp trawl mortality estimate is greater than 0.6 (Wade and Angliss 1997). PBR for the northern Gulf of Mexico Northern Coastal Stock of common bottlenose dolphins is 89 (Table 2).

Table 2. Best and minimum abundance estimates of the northern Gulf of Mexico Northern Coastal Stock of common bottlenose dolphins with Maximum Productivity Rate (R_{max}), Recovery Factor (F_r) and PBR.

N_{est}	N_{est} CV	N_{min}	F_r	R_{max}	PBR
11,543	0.19	9,881	0.45	0.04	89

ANNUAL HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

The total annual human-caused mortality and serious injury for the Northern Coastal Stock of common bottlenose dolphins during 2015–2019 is unknown because this stock is known to interact with unobserved fisheries (see below). The five-year unweighted mean annual mortality estimate for 2015–2019 for the commercial shrimp trawl fishery was 6.5 (CV=0.64; see Shrimp Trawl section below). The mean annual fishery-related mortality and serious injury during 2015–2019 for strandings identified as fishery-caused was 1.4. Mean annual mortality and serious injury during 2015–2019 due to other human-caused actions (the *Deepwater Horizon* oil spill) was predicted to be 20. The minimum total mean annual human-caused mortality and serious injury for this stock during 2015–2019 was 28 (Table 3). This is considered 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 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) various assumptions were made in the population model used to estimate population decline for the northern Gulf of Mexico Bay Stock and Estuaries (BSE) stocks impacted by the *Deepwater Horizon* (DWH) oil spill.

Fisheries Information

There are seven commercial fisheries that interact, or that potentially could interact, with this stock. These include four Category II fisheries (Southeastern U.S. Atlantic, Gulf of Mexico shrimp trawl; Southeastern U.S. Atlantic, Gulf of Mexico stone crab trap/pot; Gulf of Mexico menhaden purse seine; and Gulf of Mexico gillnet); and two Category III fisheries (Gulf of Mexico blue crab trap/pot; and Atlantic Ocean, Gulf of Mexico, Caribbean commercial passenger fishing vessel (hook and line)). Detailed fishery information is presented in Appendix III.

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

Shrimp Trawl

Between 1997 and 2019, 13 common bottlenose dolphins and nine unidentified dolphins, which could have been either common bottlenose dolphins or Atlantic spotted dolphins, became entangled in the lazy line, turtle excluder device or tickler chain gear in observed trips of the commercial shrimp trawl fishery in the Gulf of Mexico (Soldevilla *et al.* 2021). All dolphin bycatch interactions resulted in mortalities except for one unidentified dolphin that was released alive in 2009 (Maze-Foley and Garrison 2016). Soldevilla *et al.* (2015, 2016, 2021) provided mortality estimates calculated from analysis of shrimp fishery effort data and NMFS's Observer Program bycatch data. Annual mortality estimates were calculated for the years 2015–2019 from stratified annual fishery effort and bycatch rates, and the five-year unweighted mean mortality estimate was calculated for Gulf of Mexico dolphin stocks (Soldevilla *et al.* 2021). The four-area (TX, LA, MS/AL, FL) stratification method was chosen because it best approximates how fisheries operate (Soldevilla *et al.* 2015, 2016, 2021). The mean annual mortality estimate for the Northern Coastal Stock of common bottlenose dolphins is 6.5 (CV=0.64). Limitations and biases of annual bycatch mortality estimates are described in detail in Soldevilla *et al.* (2015, 2016, 2021).

Blue Crab and Stone Crab Trap/Pot

During 2015–2019, one entanglement associated with the commercial blue crab trap/pot fishery was documented which was ascribed to the Northern Coastal Stock or the Mississippi Sound, Lake Borgne, Bay Boudreau Stock. This mortality was included in the stranding database (NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 25 August 2020) and in the stranding totals presented in Table 4, and it is also included in the annual human-caused mortality and serious injury total for this stock (Table 3).

Since there is no observer program for these fisheries, it is not possible to estimate the total number of interactions or mortalities associated with crab traps/pots. The documented mortality in this gear represents a minimum known count of interactions in the last five years.

Menhaden Purse Seine

During 2015–2019, one interaction between the Northern Coastal Stock and the menhaden purse seine fishery was documented (in 2018) through the Marine Mammal Authorization Program (MMAP). There is currently no observer program for the Gulf of Mexico menhaden purse seine fishery. 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 communities from which bottlenose dolphins are being taken. The documented interaction in this gear represents a minimum known count of interactions in the last five years.

Gillnet

No marine mammal mortalities associated with gillnet fisheries have been reported or observed for the Northern Coastal Stock. There is limited observer coverage of gillnet fisheries in federal waters (e.g., Mathers *et al.* 2020), but none currently in state waters, although during 2012–2018 NMFS placed observers on commercial vessels (state permitted gillnet vessels) in the coastal state waters of Alabama, Mississippi, and Louisiana (Mathers *et al.* 2016). No takes were observed in state coastal waters during that time. However, stranding data suggest that gillnet and marine mammal interactions do occur (Read and Murray 2000), causing mortality and serious injury. During 2015–2019, nine stranded common bottlenose dolphins were recovered with markings indicative of interaction with gillnet gear (Read and Murray 2000), but no gear was attached to the carcasses and it is unknown whether the interactions with the gear contributed to the death of these animals. Seven of the nine cases were ascribed to the Northern Coastal Stock alone, and two were ascribed to both the Northern Coastal and Mississippi Sound, Lake Borgne, Bay Boudreau stocks. Because there is no observer program within this stock's boundaries, it is not possible to estimate the total number of interactions or mortalities associated with gillnet gear.

Hook and Line (Rod and Reel)

During 2015–2019, two mortalities involving hook and line gear entanglement or ingestion were documented for the Northern Coastal Stock. The mortalities occurred in 2015 and 2017, and available evidence from the stranding records suggested the hook and line gear interactions contributed to the cause of death. The mortalities were included in the stranding database (NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 25 August 2020) and in the stranding totals presented in Table 4, and were included in the annual human-caused mortality and serious injury total for this stock (Table 3).

In addition to animals included in the stranding database, during 2015–2019, there were three at-sea observations in the Northern Coastal Stock area of live common bottlenose dolphins entangled in hook and line fishing gear, and all three were considered seriously injured (Maze-Foley and Garrison 2020). The serious injuries occurred in 2015, 2016, and 2017, and were included in the annual human-caused mortality and serious injury total for this stock (Table 3).

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 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 *Deepwater Horizon* (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 Northern Coastal Stock, this model predicted the stock will have experienced a 50% (95%CI: 32–73) maximum reduction in population size (DWH MMIQT 2015; Schwacke *et al.* 2017), and for the years 2015–2019, the model projected 101 mortalities (Table 3). 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).

NOAA's Office of Law Enforcement has been investigating increasing numbers of reports from the northern Gulf of Mexico coast of violence against common bottlenose dolphins, including shootings using guns and bows and arrows, throwing pipe bombs and cherry bombs, and stabbings (Vail 2016). During 2015–2019, one mortality ascribed to the Northern Coastal Stock was documented with a bullet present just behind the head. This animal was included within the stranding database (NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 25 August 2020) and in the stranding totals presented in Table 4, but was not included in the annual human-caused mortality and serious injury total for this stock (Table 3) due to the bullet was not believed to be the definitive cause of death. 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 of fishing catch and/or bait is a growing problem in Gulf of Mexico coastal and estuarine waters and globally, and can lead to serious injury or mortality via ingestion of or entanglement in gear (e.g., Zollett and Read 2006; Read 2008; Powell and Wells 2011; Vail 2016), as well as changes in dolphin activity patterns, such as decreases in natural foraging (Powell and Wells 2011). It has been suggested that provisioning, or the illegal feeding, of wild common bottlenose dolphins, may encourage depredation because provisioning conditions dolphins to approach humans and vessels, where they then may prey on bait and catches (Vail 2016). Such conditioning increases risks of subsequent injury and mortality (Christiansen *et al.* 2016). Illegal feeding/provisioning has been documented in the literature in Florida and Texas (Bryant 1994; Samuels and Bejder 2004; Cunningham-Smith *et al.* 2006; Powell and

Wells 2011; Powell *et al.* 2018).

Feeding or provisioning of wild common bottlenose dolphins has been documented in Florida, particularly near St. Andrew Bay (Panama City Beach) in the Panhandle (Samuels and Bejder 2004; Powell *et al.* 2018) and south of Sarasota Bay (Cunningham-Smith *et al.* 2006; Powell and Wells 2011), and also in Texas near Corpus Christi (Bryant 1994). Feeding wild dolphins is defined under the MMPA as a form of 'take' because it can alter their natural behavior and increase their risk of injury or death. Nevertheless, a high rate of provisioning has been observed south of Sarasota Bay since 1990 (Cunningham-Smith *et al.* 2006; Powell and Wells 2011), and near St. Andrew Bay in 1998 (Samuels and Bejder 2004) and in 2014 (Powell *et al.* 2018). For many years within certain areas of St. Andrew Bay and adjacent coastal waters, it has been typical to see wild dolphins surrounded by multiple boats, multiple personal watercraft, and multiple swimmers. Studies have documented a high rate of unregulated food provisioning and recorded many interactions with humans that put dolphins at risk of injury, illness, or death (Samuels and Bejder 2004; Powell *et al.* 2018). Research by Powell *et al.* (2018) during 2014 indicated the number of conditioned individual dolphins (conditioned to human interaction by food reinforcement; animals that accepted food handouts from people on a regular basis) tripled (n=21) compared to those documented in 1998 by Samuels and Bejder (2004; n=7), and that overall the problems of illegal feeding and harassment had increased. Powell *et al.* (2018) found that conditioned dolphins spent the majority of their time approaching boats to beg for food and patrolling among boats and swimmers looking for handouts, which in turn increases their risk of boat strike, entanglement in or hooking by fishing gear, or retaliation by angry fishermen (Wells and Scott 1997; Powell and Wells 2011; Adimey *et al.* 2014; Powell *et al.* 2018).

Swimming with wild common bottlenose dolphins has also been documented in Florida in Key West (Samuels and Engleby 2007) and near Panama City Beach (Samuels and Bejder 2004). Near Panama City Beach, Samuels and Bejder (2004) concluded that dolphins were amenable to swimmers due to illegal provisioning. Swimming with wild dolphins may cause harassment, and harassment is illegal under the MMPA.

All mortalities and serious injuries from known sources for the Northern Coastal Stock are summarized in Table 3.

Table 3. Summary of the incidental mortality and serious injury of common bottlenose dolphins (Tursiops truncatus) of the Northern Coastal Stock. For fisheries that do not have an ongoing, federal observer program, 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 strandings, at-sea counts, and fisherman self-reported takes, the number reported is a minimum because not all strandings, at-sea cases, or gear interactions are detected. See the Annual Human-Caused Mortality and Serious Injury section for biases and limitations of mortality estimates, and the Strandings section for limitations of stranding data. 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	2015–2019	Observer Data	6.5 (CV=0.64)	NA
Crab Trap/Pot	2015–2019	Stranding Data	NA	1
Menhaden Purse Seine	2015–2019	MMAP Fisherman self-reported takes	NA	1
Gillnet	2015–2019	Observer Data and Stranding Data	NA	0
Hook and Line	2015–2019	Stranding Data and At-Sea Observations	NA	5

Mean Annual Mortality due to commercial fisheries (2015–2019)	**7.9**
Mortality due to DWH (5-year Projection)	**101**
Mean Annual Mortality due to DWH (2015–2019)	**20**
Minimum Total Mean Annual Human-Caused Mortality and Serious Injury (2015–2019)	**28**

Strandings

A total of 137 common bottlenose dolphins were found stranded in Northern Coastal Stock waters of the Gulf of Mexico from 2015 through 2019 (Table 4; NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 25 August 2020). There was evidence of human interaction (HI) for 22 of the strandings. No evidence of human interaction was detected for five strandings, and for the remaining 110 strandings, it could not be determined if there was evidence of human interaction. Human interactions were from several sources, including nine with markings indicative of interaction with gillnet gear, two entanglements with hook and line gear, one entanglement in commercial blue crab trap/pot gear, two animals with evidence of a vessel strike, and one animal with a gunshot wound. It should be noted that evidence of human interaction does not necessarily mean the interaction caused the animal's stranding or death.

The assignment of animals to a single stock is impossible in some regions where stocks overlap, especially in nearshore coastal waters (Maze-Foley *et al.* 2019). Of the 137 strandings ascribed to the Northern Coastal Stock, 78 were ascribed solely to this stock. The counts in Table 4 may include some animals from the Mississippi Sound, Lake Borgne, Bay Boudreau Stock and/or the St. Joseph Bay Stock and thereby overestimate the number of strandings for the Northern Coastal Stock. 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).

There are a number of other difficulties associated with the interpretation of stranding data. Stranding data underestimate the extent of human and fishery-related mortality and serious injury because not all of the dolphins that die or are seriously injured in human interactions wash ashore, or, if they do, they are not all recovered (Peltier *et al*. 2012; Wells *et al*. 2015; Carretta *et al.* 2016). Additionally, not all carcasses will show evidence of human interaction, entanglement or other fishery-related interaction due to decomposition, scavenger damage, etc. (Byrd *et al*. 2014). Finally, the level of technical expertise among stranding network personnel varies widely as does the ability to recognize signs of human interaction.

Since 1990, there have been 15 bottlenose dolphin die-offs or Unusual Mortality Events (UMEs) in the northern Gulf of Mexico (http://www.nmfs.noaa.gov/pr/health/mmume/events.html, accessed 5 November 2020), and eight of these have occurred within the boundaries of the Northern Coastal Stock and may have affected the stock. 1) From January through May 1990, a total of 344 bottlenose dolphins stranded in the northern Gulf of Mexico. Overall this represented a two-fold increase in the prior maximum recorded 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). 2) In 1993–1994 a UME of bottlenose dolphins caused by morbillivirus started in the Florida Panhandle and spread west with most of the mortalities occurring in Texas (Lipscomb 1993; Lipscomb *et al.* 1994; Litz *et al.* 2014). From February through April 1994, 236 bottlenose dolphins were found dead on Texas beaches, of which 67 occurred in a single 10-day period. 3) 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. 4) Between August 1999 and May 2000, 150 bottlenose dolphins died coincident with *K. brevis* blooms and fish kills in the Florida Panhandle (additional strandings included three Atlantic spotted dolphins, *Stenella frontalis*, one Risso's dolphin, *Grampus griseus*, two Blainville's beaked whales, *Mesoplodon densirostris*, and four unidentified dolphins). Brevetoxin was determined to be the cause of this event (Twiner *et al.* 2012; Litz *et* *al.* 2014). 5) In March and April 2004, in another Florida Panhandle UME attributed to *K. brevis* blooms, 105 bottlenose dolphins and two unidentified dolphins stranded dead (Litz *et al.* 2014). Although there was no indication of a *K. brevis* bloom at the time, high levels of brevetoxin were found in the stomach contents of the stranded dolphins (Flewelling *et al.* 2005; Twiner *et al.* 2012). 6) A separate UME was declared in the Florida Panhandle after elevated numbers of dolphin strandings occurred in association with a *K. brevis* bloom in September 2005. Dolphin strandings remained elevated through the spring of 2006 and brevetoxin was again detected in the tissues of most of the stranded dolphins and determined to be the cause of the event (Twiner *et al.* 2012; Litz *et al.* 2014). Between September 2005 and April 2006 when the event was officially declared over, a total of 88 bottlenose dolphin strandings occurred (plus strandings of five unidentified dolphins). 7) 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 2 December 2020). It includes cetaceans that stranded prior to the DWH oil spill (see Habitat Issues section below), during the spill, and after. Exposure to the DWH oil spill was determined to be the primary underlying cause of the elevated stranding numbers in the northern Gulf of Mexico after the spill (e.g., Schwacke *et al.* 2014; Venn-Watson *et al.* 2015; Colegrove *et al.* 2016; DWH NRDAT 2016). During 2011–2014, nearly all stranded dolphins from this stock were considered to be part of the UME. 8) During 1 February 2019 to 30 November 2019, a UME was declared for the area from the eastern border of Taylor County, Florida, west through Alabama, Mississippi, and Louisiana (http://www.nmfs.noaa.gov/pr/health/mmume/cetacean_gulfofmexico.htm, accessed 5 November 2020). A total of 337 common bottlenose dolphins stranded during this event, with 45 of them being from the Northern Coastal Stock. The largest number of mortalities occurred in eastern Louisiana and Mississippi. An investigation concluded the event was caused by exposure to low salinity waters as a result of extreme freshwater discharge from rivers. The unprecedented amount of freshwater discharge during 2019 (e.g., Gasparini and Yuill 2020) resulted in low salinity levels across the region.

Table 4. Common bottlenose dolphin strandings occurring in the Northern Coastal Stock area from 2015 to 2019, 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 25 August 2020). Please note HI does not necessarily mean the interaction caused the animal's death.

Stock	Category	2015	2016	2017	2018	2019	Total
Northern Coastal Stock	Total Stranded	23	26	24	15	49[e]	137
	Human Interaction						
	---Yes	5[a]	5[b]	6[c]	2[d]	4[f]	22
	---No	1	1	1	2	0	5
	---CBD	17	20	17	11	45	110

a. Includes 1 entanglement interaction with hook and line gear (mortality) and 2 mortalities with evidence of a vessel strike.
b. Includes 4 fisheries interactions (FIs), 2 of which were mortalities with markings indicative of interaction with gillnet gear; also includes 1 mortality with a gunshot wound.
c. All 6 are FIs, including 1 entanglement interaction with hook and line gear (mortality) and 3 mortalities with markings indicative of interaction with gillnet gear.
d. Both are FIs, including 1 entanglement interaction with commercial blue crab trap/pot gear (mortality).
e. 45 strandings were part of the UME event in the northern Gulf of Mexico.
f. All 4 are FIs, all of which were mortalities with markings indicative of interaction with gillnet gear.

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 Natural Resource Damage Assessment (NRDA) process was initiated under the Oil Pollution Act of 1990. A variety of NRDA research studies were conducted to determine potential impacts of the spill on marine mammals. These studies estimated that 82% (95%CI: 55–100) of the Northern Coastal Stock of common bottlenose dolphins in the Gulf were exposed to oil, that 37% (95%CI: 17–53) of females suffered from reproductive failure, and 30% (95%CI: 11–47) suffered adverse health effects (DWH MMIQT 2015). A population model estimated that the stock experienced a 50% maximum reduction in population size (see Other Mortality section above).

The nearshore habitat occupied by the three coastal stocks is adjacent to areas of high human population and in some areas, such as Tampa Bay, Florida, Galveston, Texas, and Mobile, Alabama, is highly industrialized. Concentrations of anthropogenic chemicals such PCBs and DDT and its metabolites vary from site to site, and can reach levels of concern for bottlenose dolphin health and reproduction in the southeastern U.S. (Schwacke et al. 2002). PCB concentrations in three stranded dolphins sampled from the Eastern Coastal Stock area ranged from 16-46µg/g wet weight. Two stranded dolphins from the Northern Coastal Stock area had the highest levels of DDT derivatives of any of the bottlenose dolphin liver samples analyzed in conjunction with a 1990 mortality investigation conducted by NMFS (Varanasi et al. 1992). The significance of these findings is unclear, but there is some evidence that increased exposure to anthropogenic compounds may reduce immune function in bottlenose dolphins (Lahvis *et al.* 1995), or impact reproduction through increased first-born calf mortality (Wells *et al.* 2005).

The Mississippi River, which drains about two-thirds of the continental U.S., flows into the north-central Gulf of Mexico and deposits its nutrient load which is linked to the formation of one of the world's largest areas of seasonal hypoxia (Rabalais *et al.* 1999). This area is located in Louisiana coastal waters west of the Mississippi River delta. How it affects common bottlenose dolphins is not known.

STATUS OF STOCK

The common bottlenose dolphin is not listed as threatened or endangered under the Endangered Species Act, and the Gulf of Mexico Northern Coastal Stock is not considered strategic under the Marine Mammal Protection Act. However, the occurrence of a UME of unprecedented size and duration has impacted the Northern Coastal Stock area and is a cause for concern. The DWH damage assessment estimated that the stock experienced a 50% (95%CI: 32–73) maximum reduction in population size due to the oil spill (DWH MMIQT 2015; Schwacke *et al.* 2017). Total U.S. fishery-related 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. The status of this stock relative to optimum sustainable population in the Gulf of Mexico EEZ is unknown. There are insufficient data to determine the population trends for this stock.

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COMMON BOTTLENOSE DOLPHIN (*Tursiops truncatus truncatus*): Gulf of Mexico Western Coastal Stock

STOCK DEFINITION AND GEOGRAPHIC RANGE

Common bottlenose dolphins inhabit coastal waters throughout the northern Gulf of Mexico (i.e., U.S. Gulf of Mexico; Mullin *et al.* 1990). As a working hypothesis, it is assumed that the dolphins occupying habitats with dissimilar climatic, coastal and/or oceanographic characteristics might be restricted in their movements between habitats, and thus constitute separate stocks. Therefore, northern Gulf of Mexico coastal waters have been divided for management purposes into three stock areas: eastern, northern and western, with coastal waters defined as waters between the shore, barrier islands or presumed outer bay boundaries out to the 20-m isobath (Figure 1). The 20-m depth seaward boundary corresponds to survey strata (Scott 1990; Blaylock and Hoggard 1994; Fulling *et al.* 2003) and thus represents a management boundary rather than an ecological boundary. The Western Coastal common bottlenose dolphin stock area extends from the Mississippi River Delta to the Texas-Mexico border. This region is characterized by an arid to temperate climate, sand beaches in southern Texas, extensive coastal marshes in northern Texas and Louisiana, and varying amounts of freshwater input. Dolphins belonging to this stock are all expected to be of the coastal ecotype (Vollmer 2011). The Western Coastal Stock is trans-boundary with Mexico; however, there is no information available for abundance estimation, nor for estimating fishery-related mortality in Mexican waters.

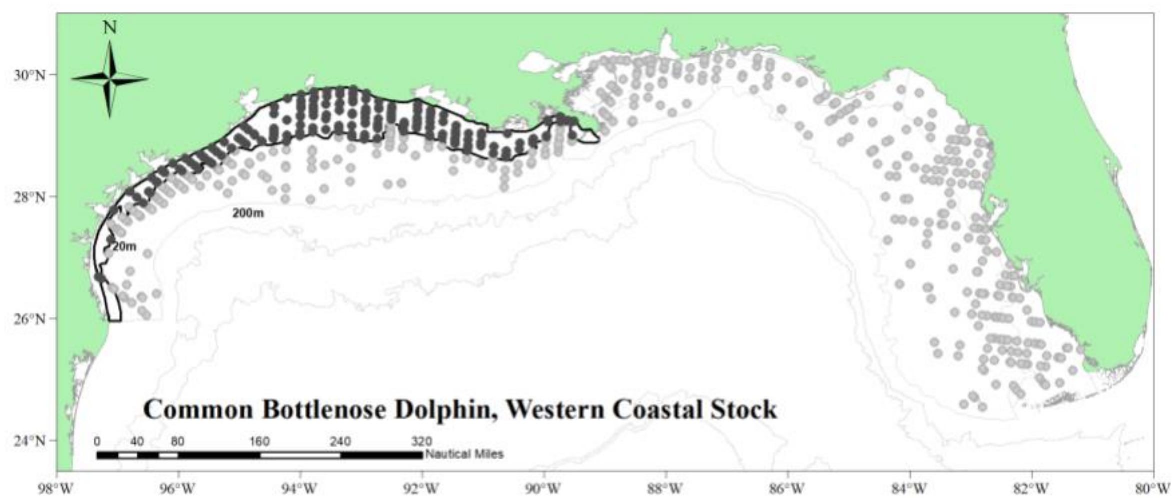

Figure 1. Distribution of common bottlenose dolphin on-effort sightings in coastal and continental shelf waters during SEFSC aerial surveys in summer 2017, winter 2018, and fall 2018. Sightings within the boundaries of the Western Coastal Stock are denoted by the black circles. Isobaths are the 20-m, 200-m, 1,000-m, and 2,000-m depth contours.

Recently, genetic analyses of population structure in coastal, shelf, and oceanic waters of the Gulf of Mexico revealed seven demographically independent populations in the northern Gulf of Mexico, suggesting the current stock designations and boundaries in these waters do not accurately reflect the population structure (Vollmer and Rosel 2017). Sampling within the range of the Western Coastal Stock was limited and further work is necessary to determine the boundaries of these demographically independent populations.

This stock's boundaries abut other common bottlenose dolphin stocks, namely the Northern Coastal Stock, Continental Shelf Stock and several bay, sound and estuary stocks in Texas and Louisiana, and while individuals from different stocks may occasionally overlap, it is not thought that significant mixing or interbreeding occurs between them. Fazioli *et al.* (2006) conducted photo-identification surveys of coastal waters off Tampa Bay, Sarasota Bay and Lemon Bay, Florida, over 14 months. They found both 'inshore' and 'Gulf' dolphins inhabited coastal waters but the two types used coastal waters differently. Dolphins from the inshore communities were observed occasionally in Gulf near-shore waters adjacent to their inshore range, whereas 'Gulf' dolphins were found primarily in open Gulf of Mexico waters with some displaying seasonal variations in their use of the study area. The 'Gulf' dolphins did not show a preference for waters near passes as was seen for 'inshore' dolphins, but moved throughout the study area and made greater use of waters offshore of waters used by 'inshore' dolphins. During winter months abundance of 'Gulf' groups decreased while abundance for 'inshore' groups increased. These findings support an earlier report by Irvine *et al.* (1981) of increased use of pass and coastal waters by Sarasota Bay dolphins in winter. Seasonal movements of identified individuals and abundance indices suggested that part of the 'Gulf' dolphin community moved out of the study area during winter, but their destination is unknown (Fazioli *et al.* 2006). In a follow-up study, Sellas *et al.* (2005) examined genetic population subdivision in the study area of Fazioli *et al.* (2006), and found evidence of significant population structure among all areas. Rosel *et al.* (2017) also identified significant genetic differentiation between estuarine residents of Barataria Bay and the adjacent coastal stock, further supporting separation of coastal and estuarine stocks.

Finally, off Galveston, Texas, Beier (2001) reported an open population of individual dolphins in coastal waters, but several individual dolphins had been sighted previously by other researchers over a 10-year period. Some coastal animals may move relatively long distances alongshore. Two bottlenose dolphins previously seen in the South Padre Island area in Texas were seen in Matagorda Bay, 285 km north, in May 1992 and May 1993 (Lynn and Würsig 2002).

POPULATION SIZE

The best abundance estimate available for the northern Gulf of Mexico Western Coastal Stock of common bottlenose dolphins is 20,759 (CV=0.13; Table 1; Garrison *et al.* 2021). This estimate is from an inverse-variance weighted average of seasonal abundance estimates from aerial surveys conducted during summer 2017 and fall 2018.

Earlier Abundance Estimates

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

Recent Surveys and Abundance Estimates

The Southeast Fisheries Science Center conducted aerial surveys of continental shelf waters (shoreline to 200 m depth) along the U.S. Gulf of Mexico coast from the Florida Keys to the Texas/Mexico border during summer (June–August) 2017 and fall (October–November) 2018. The stock was only partially surveyed during a winter 2018 aerial survey, and therefore this survey was not included in the current abundance estimates (Garrison *et al.* 2021). The surveys were conducted along tracklines oriented perpendicular to the shoreline and spaced 20 km apart. The total survey effort varied during each survey due to weather conditions, and was 10,781 km (fall) and 14,590 km (summer). 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). Abundance was calculated using mark-recapture distance sampling implemented in package mrds (version 2.21; Laake *et al.* 2020) in the R statistical programming language. This approach estimates both the probability of detection on the trackline and within the surveyed strip accounting for the effects of sighting conditions (e.g., sea state, glare, turbidity, and cloud cover). A different detection probability model was used for each seasonal survey (Garrison *et al.* 2021). The survey data were post-stratified into spatial boundaries corresponding to the defined boundaries of common bottlenose dolphin stocks within the surveyed area. The abundance estimates for the Western Coastal Stock of common bottlenose dolphins were based upon tracklines and sightings in waters from the shoreline to the 20-m isobath and between the Texas-Mexico border and the Mississippi River Delta. The seasonal abundance estimates for this stock were: summer – 18,601 (CV=0.30) and fall – 21,766 (CV=0.14). Due to the uncertainty in stock movements and apparent seasonal variability in the abundance of the stock, a weighted average of these seasonal estimates was taken where the weighting was the inverse of the CV. This approach weights estimates with higher precision more heavily in the final weighted mean. The resulting weighted mean and best estimate of abundance for the Western Coastal Stock of common bottlenose dolphins was 20,759 (CV=0.13; Table 1; Garrison *et al.* 2021).

Table 1. Most recent abundance estimate (N_{est}) and coefficient of variation (CV) of the northern Gulf of Mexico Western Coastal Stock of common bottlenose dolphins (0–20-m isobaths) based on summer 2017, winter 2018, and fall 2018 aerial surveys.

Years	Area	Nest	CV
2017, 2018	Gulf of Mexico	20,759	0.13

Minimum Population Estimate

The minimum population estimate is the lower limit of the two-tailed 60% confidence interval of the log-normally distributed 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 Coastal Stock of common bottlenose dolphins is 20,759 (CV=0.13). Therefore, the minimum population estimate for the northern Gulf of Mexico Western Coastal Stock is 18,585 (Table 2).

Current Population Trend

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). Two point estimates of common bottlenose dolphin abundance for the Western Coastal Stock have been made based on aerial data from surveys during 2011−2012 and 2017−2018 (Garrison *et al.* 2021). Each of these surveys had a similar design and was conducted using the same aircraft and observer configuration. The resulting inverse variance weighted best abundance estimates for seasonal surveys were: 2011–2012 – 19,381 (CV=0.20) and 2017−2018 – 20,759 (CV=0.13). A trends analysis is not possible because there are only two abundance estimates available. For further information on comparisons of old and current abundance estimates for this stock see Garrison *et al.* (2021).

CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

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

POTENTIAL BIOLOGICAL REMOVAL

Potential Biological Removal (PBR) is the product of minimum population size, one-half the maximum productivity rate and a recovery factor (Wade and Angliss 1997). The minimum population size is 18,585. The maximum productivity rate is 0.04, the default value for cetaceans. The recovery factor is 0.45 because the CV of the shrimp trawl mortality estimate is greater than 0.6 (Wade and Angliss). PBR for the northern Gulf of Mexico Western Coastal Stock of common bottlenose dolphins is 167 (Table 2).

Table 2. Best and minimum abundance estimates of the northern Gulf of Mexico Western Coastal Stock of common bottlenose dolphins with Maximum Productivity Rate (R_{max}), Recovery Factor (F_r) and PBR.

N_{est}	N_{est} CV	N_{min}	F_r	R_{max}	PBR
20,759	0.13	18,585	0.45	0.04	167

ANNUAL HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

The total annual human-caused mortality and serious injury for the Western Coastal Stock of common bottlenose dolphins during 2015–2019 is unknown because this stock is known to interact with unobserved fisheries (see below). The five-year unweighted mean annual mortality estimate for 2015–2019 for the commercial shrimp trawl fishery was 32 (CV=0.65; see Shrimp Trawl section below). The mean annual fishery-related mortality and serious injury during 2015–2019 for strandings identified as fishery-caused was 0.4. Mean annual mortality and serious injury during 2015–2019 due to other human-caused actions (the *Deepwater Horizon* (DWH) oil spill and foreign fisheries) was predicted to be 3.2. The minimum total mean annual human-caused mortality and serious injury for this stock during 2015–2019 was 36 (Table 3). This is considered 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 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) various assumptions were made in the population model used to estimate population decline for the northern Gulf of Mexico Bay Stock and Estuaries (BSE) stocks impacted by the DWH oil spill.

Fisheries Information

There are five commercial fisheries that interact, or that potentially could interact, with this stock. These include three Category II fisheries (Southeastern U.S. Atlantic, Gulf of Mexico shrimp trawl; Gulf of Mexico menhaden purse seine; and Gulf of Mexico gillnet); and two Category III fisheries (Gulf of Mexico blue crab trap/pot; and Atlantic Ocean, Gulf of Mexico, Caribbean commercial passenger fishing vessel (hook and line)). Detailed fishery information is presented in Appendix III.

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

Shrimp Trawl

Between 1997 and 2019, 13 common bottlenose dolphins and nine unidentified dolphins, which could have been either common bottlenose dolphins or Atlantic spotted dolphins, became entangled in the lazy line, turtle excluder device or tickler chain gear in observed trips of the commercial shrimp trawl fishery in the Gulf of Mexico (Soldevilla *et al.* 2021). All dolphin bycatch interactions resulted in mortalities except for one unidentified dolphin that was released alive in 2009 (Maze-Foley and Garrison 2016). Soldevilla *et al.* (2015, 2016, 2021) provided mortality estimates calculated from analysis of shrimp fishery effort data and NMFS's Observer Program bycatch data. Annual mortality estimates were calculated for the years 2015–2019 from stratified annual fishery effort and bycatch rates, and the five-year unweighted mean mortality estimate was calculated for Gulf of Mexico dolphin stocks (Soldevilla *et al.* 2021). The four-area (TX, LA, MS/AL, FL) stratification method was chosen because it best approximates how fisheries operate (Soldevilla *et al.* 2015, 2016, 2021). The mean annual mortality estimate for the Western Coastal Stock of common bottlenose dolphins is 32 (CV=0.65). Limitations and biases of annual bycatch mortality estimates are described in detail in Soldevilla *et al.* (2015, 2016, 2021).

In addition, chaffing gear from a commercial shrimp trawl was recovered in a dolphin carcass that stranded during 2015. It is likely the animal ingested the gear while removing gilled fish that were caught in the trawl net. This animal was ascribed to both the Barataria Bay Estuarine System and Western Coastal stocks, and it was included in the stranding database (NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 25 August 2020; Table 4).

Menhaden Purse Seine

During 2015–2019, no interactions between the Western Coastal Stock and the menhaden purse seine fishery were documented. There is currently no observer program for the Gulf of Mexico menhaden purse seine fishery. Previously, interactions between the Western Coastal stock and this fishery have been documented by both a pilot observer program and the Marine Mammal Authorization Program. 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 communities from which bottlenose dolphins are being taken.

Gillnet

No marine mammal mortalities associated with U.S. gillnet fisheries have been reported or observed for the Western Coastal Stock. There is limited observer coverage of this fishery in federal waters (e.g., Mathers *et al.* 2020), but none currently in state waters, although during 2012–2018 NMFS placed observers on commercial vessels (state permitted gillnet vessels) in the coastal state waters of Alabama, Mississippi, and Louisiana (Mathers *et al.* 2016). No takes were observed in state waters during this time. Because there is no observer program within this stock's boundaries, it is not possible to estimate the total number of interactions or mortalities associated with gillnet gear.

Blue Crab Trap/Pot

During 2015–2019, no interactions were documented for the Western Coastal Stock with crab trap/pot fisheries.

An earlier interaction was documented for this stock (from 2008). Since there is no observer program, it is not possible to estimate the total number of interactions or mortalities associated with crab traps/pots.

Hook and Line (Rod and Reel)

During 2015–2019, one mortality involving hook and line gear entanglement was documented for the Western Coastal Stock. The mortality occurred in 2018, and available evidence from the stranding record suggested the hook and line gear interaction contributed to the cause of death. The mortality was included in the stranding database (NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 25 August 2020) and in the stranding totals presented in Table 4, and in the annual human-caused mortality and serious injury total (Table 3).

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 observer program. The documented interaction in this gear represents 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 Western Coastal Stock, the model predicted the stock experienced a 5% (95%CI: 3–9) maximum reduction in population size due to the oil spill (DWH MMIQT 2015; DWH NRDAT 2016; Schwacke *et al.* 2017), and for the years 2015–2019, the model projected 16 mortalities (Table 3). 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 addition to the fishery interactions discussed above, two additional fishery-related mortalities were documented during 2015–2019. One mortality was documented in 2017 as a result of entanglement in monofilament line. It could not be determined if the line was part of a net or not. In 2018, an additional mortality was documented near the Texas/Mexico border in Mexican shark gillnet gear. Both of these interactions were included in the stranding database (NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 25 August 2020) and in the totals presented in Table 4, and also in the annual human-caused mortality and serious injury total for this stock (Table 3).

NOAA's Office of Law Enforcement has been investigating increased reports from along the northern Gulf of Mexico coast of violence against common bottlenose dolphins, including shootings via guns and bows and arrows, throwing pipe bombs and cherry bombs, and stabbings (Vail 2016). During 2015–2019, for one mortality, gunshot pellets were found during the necropsy. The gunshot occurred pre-mortem but was not believed 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 25 August 2020) and in the totals presented in Table 4, but was not included in the annual human-caused mortality and serious injury total for this stock (Table 3). 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 2019 shooting involved depredation.

Depredation of fishing catch and/or bait 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 common bottlenose dolphins, may encourage depredation because provisioning conditions dolphins to approach humans and vessels, where they then may prey on bait and catches (Vail 2016). Such conditioning increases risks of subsequent injury and mortality (Christiansen *et al.* 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.

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. No interactions have been documented during the most recent five years, 2015–2019. Historically, two mortalities were documented involving relocation trawling activities and common bottlenose dolphins likely belonging to the Western Coastal Stock (2005, 2007).

All mortalities and serious injuries from known sources for the Western Coastal Stock are summarized in Table 3**.**

Table 3. Summary of the incidental mortality and serious injury of common bottlenose dolphins (Tursiops truncatus) of the Western Coastal Stock. For fisheries that do not have an ongoing, federal observer program, 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 strandings, at-sea counts, and fisherman self-reported takes, the number reported is a minimum because not all strandings, at-sea cases, or gear interactions are detected. See the Annual Human-Caused Mortality and Serious Injury section for biases and limitations of mortality estimates, and the Strandings section for limitations of stranding data. 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	2015–2019	Observer Data	32 (CV=0.65)	NA
Menhaden Purse Seine	2015–2019	MMAP Fisherman self-reported takes	NA	0
Gillnet	2015–2019	Observer Data and Stranding Data	NA	0
Crab Trap/Pot	2015–2019	Stranding Data	NA	0
Hook and Line	2015–2019	Stranding Data and At-Sea Observations	NA	1
Unknown Gear	2015–2019	Stranding Data	NA	1
Mean Annual Mortality due to commercial fisheries (2015–2019)			**32.4**	
Other Takes (foreign fishing gear, 5-year Count)			1	
Mortality due to DWH (5-year Projection)			16	
Mean Annual Mortality due to other takes and DWH (2015–2019)			**3.2**	
Minimum Total Mean Annual Human-Caused Mortality and Serious Injury (2015–2019)			**36**	

Strandings

A total of 586 common bottlenose dolphins were found stranded in Western Coastal Stock waters of the northern Gulf of Mexico from 2015 through 2019 (Table 4; NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 25 August 2020). There was evidence of human interaction (HI) for 26 of the strandings. No evidence of human interaction was detected for 63 strandings, and for the remaining 497 strandings, it could not be determined if there was evidence of human interaction. Human interactions were from several sources, including an entanglement with hook and line gear, an entanglement in a Mexican shark gillnet, and an animal with gunshot wounds. It should be noted that evidence of human interaction does not necessarily mean the interaction caused the animal's stranding or death.

The assignment of animals to a single stock is impossible in some regions where stocks overlap, especially in nearshore coastal waters (Maze-Foley *et al.* 2019). Of the 586 strandings ascribed to the Western Coastal Stock, 441 were ascribed solely to this stock. The counts in Table 4 may include some animals from the Barataria Bay Estuarine System; Terrebonne-Timbalier Bay Estuarine System; Galveston Bay, East Bay, Trinity Bay; West Bay; Calcasieu Lake; Nueces Bay, Corpus Christi Bay; or Laguna Madre stocks, and thereby overestimate the number of strandings for the Western Coastal Stock. 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).

There are a number of other difficulties associated with the interpretation of stranding data. Stranding data underestimate the extent of human and fishery-related mortality and serious injury because not all of the dolphins that die or are seriously injured in human interactions wash ashore, or, if they do, they are not all recovered (Peltier *et al*. 2012; Wells *et al*. 2015; Carretta *et al.* 2016). Additionally, not all carcasses will show evidence of human interaction, entanglement or other fishery-related interaction due to decomposition, scavenger damage, etc. (Byrd *et al*. 2014). Finally, the level of technical expertise among stranding network personnel varies widely as does the ability to recognize signs of human interaction.

Since 1990, there have been 15 bottlenose dolphin die-offs or Unusual Mortality Events (UMEs) in the northern Gulf of Mexico (http://www.nmfs.noaa.gov/pr/health/mmume/events.html, accessed 5 November 2020), and eight of these have occurred within the boundaries of the Western Coastal Stock and may have affected the stock. 1) From January through May 1990, a total of 344 bottlenose dolphins stranded in the northern Gulf of Mexico. Overall this represented a two-fold increase in the prior maximum recorded 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). 2) In March and April 1992, 119 bottlenose dolphins stranded in Texas, about nine times the average number. The cause of this event was not determined, but low salinity due to record rainfall combined with pesticide runoff and exposure to morbillivirus were suggested as potential contributing factors (Duignan *et al.* 1996; Colbert *et al.* 1999; Litz *et al.* 2014). 3) In 1993–1994 a UME of bottlenose dolphins likely caused by morbillivirus started in the Florida Panhandle and spread west with most of the mortalities occurring in Texas (Lipscomb 1993; Lipscomb *et al.* 1994; Litz *et al.* 2014). From February through April 1994, 236 bottlenose dolphins were found dead on Texas beaches, of which 67 occurred in a single 10-day period. 4) During February and March of 2007 an event was declared for northeast Texas and western Louisiana involving 64 bottlenose dolphins and two unidentified dolphins. Decomposition prevented conclusive analyses on most carcasses. 5) During February and March of 2008 an additional event was declared in Texas involving 111 bottlenose dolphin strandings (plus strandings of one unidentified dolphin and one melon-headed whale). Most of the animals recovered were in a decomposed state. The event has been closed, however, the investigation is ongoing. 6) 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 2 December 2020). It included cetaceans that stranded prior to the DWH oil spill (see Habitat Issues section below), during the spill, and after. Exposure to the DWH oil spill was determined to be the primary underlying cause of the elevated stranding numbers in the northern Gulf of Mexico after the spill (e.g., Schwacke *et al.* 2014; Venn-Watson *et al.* 2015; Colegrove *et al.* 2016; DWH NRDAT 2016). 7) A UME occurred from November 2011 to March 2012 across five Texas counties including 126 bottlenose dolphin strandings. Ninety-six animals from this stock were considered to be part of the UME. The strandings were coincident with a harmful algal bloom of *Karenia brevis*, but researchers have not determined that was the cause of the event. 8) During 1 February 2019 to 30 November 2019, a UME was declared for the area from the eastern border of Taylor County, Florida, west through Alabama, Mississippi, and Louisiana (http://www.nmfs.noaa.gov/pr/health/mmume/

cetacean_gulfofmexico.htm, accessed 5 November 2020). A total of 337 common bottlenose dolphins stranded during this event, with 44 of them being from the Western Coastal Stock. The largest number of mortalities occurred in eastern Louisiana and Mississippi. An investigation concluded the event was caused by exposure to low salinity waters as a result of extreme freshwater discharge from rivers. The unprecedented amount of freshwater discharge during 2019 (e.g., Gasparini and Yuill 2020) resulted in low salinity levels across the region.

Table 4. Common bottlenose dolphin strandings occurring in the Western Coastal Stock area from 2015 to 2019, 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 25 August 2020). Please note HI does not necessarily mean the interaction caused the animal's death.

Stock	Category	2015	2016	2017	2018	2019	Total
Western Coastal Stock	Total Stranded	94	100	143	123	126[d]	586
	Human Interaction						
	---Yes	3[a]	2	6[b]	8[c]	7[e]	26
	---No	3	6	22	16	16	63
	---CBD	88	92	115	99	103	497

a. Includes 1 interaction with chaffing gear from a commercial shrimp trawl (mortality).
b. Includes 2 fisheries interactions (FIs).
c. Includes FIs, including 1 interaction with hook and line gear (mortality) and 1 interaction with a Mexican shark gillnet (mortality).
d. 44 strandings were part of the UME event in the northern Gulf of Mexico.
e. Includes 1 FI and 1 animal with evidence of gunshot wounds (mortality).

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 Natural Resource Damage Assessment (NRDA) process was initiated under the Oil Pollution Act of 1990. A variety of NRDA research studies were conducted to determine potential impacts of the spill on marine mammals. These studies estimated that 23% (95%CI: 16–32) of the Western Coastal Stock of common bottlenose dolphins in the Gulf were exposed to oil, that 10% (95%CI: 5–15) of females suffered from reproductive failure, and 8% (95%CI: 3–13) suffered adverse health effects (DWH MMIQT 2015). A population model estimated that the stock experienced a 5% maximum reduction in population size (see Other Mortality section above).

In 2014, a vessel collision in Galveston Bay near Texas City released approximately 168,000 gallons of intermediate fuel oil. Through the National Resource Damage Assessment (NRDA) process, impacts of this spill are currently being evaluated and will include impacts to common bottlenose dolphins of the Western Coastal Stock (NOAA DAARP 2018).

The nearshore habitat occupied by the three coastal stocks is adjacent to areas of high human population and in some areas, such as Tampa Bay, Florida, Galveston, Texas, and Mobile, Alabama, is highly industrialized. Concentrations of anthropogenic chemicals such PCBs and DDT and its metabolites vary from site to site, and can reach levels of concern for bottlenose dolphin health and reproduction in the southeastern U.S. (Schwacke *et al.* 2002). PCB concentrations in three stranded dolphins sampled from the Eastern Coastal Stock area ranged from 16-46µg/g wet weight. Two stranded dolphins from the Northern Coastal Stock area had the highest levels of DDT derivatives of any of the bottlenose dolphin liver samples analyzed in conjunction with a 1990 mortality investigation conducted by NMFS (Varanasi *et al.* 1992). The significance of these findings is unclear, but there is some evidence that increased exposure to anthropogenic compounds may reduce immune function in bottlenose dolphins (Lahvis *et al.* 1995), or impact reproduction through increased first-born calf mortality (Wells *et al.* 2005). Concentrations of chlorinated hydrocarbons and metals were relatively low in most of the bottlenose dolphins examined in conjunction with an anomalous mortality event in Texas bays in 1990; however, some had concentrations at levels of possible toxicological concern (Varanasi *et al.* 1992). Agricultural runoff following periods of high rainfall in 1992 was implicated in a high level of bottlenose dolphin mortalities in Matagorda Bay, which is adjacent to the Western Coastal Stock area.

The Mississippi River, which drains about two-thirds of the continental U.S., flows into the north-central Gulf of Mexico and deposits its nutrient load which is linked to the formation of one of the world's largest areas of seasonal hypoxia (Rabalais *et al.* 1999). This area is located in Louisiana coastal waters west of the Mississippi River delta. How it affects bottlenose dolphins is not known.

STATUS OF STOCK

The common bottlenose dolphin is not listed as threatened or endangered under the Endangered Species Act, and the Gulf of Mexico Western Coastal Stock is not considered strategic under the Marine Mammal Protection Act. However, the occurrence of a UME of unprecedented size and duration has impacted the Western Coastal Stock area and is cause for concern. Total U.S. fishery-related 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. The status of this stock relative to optimum sustainable population in the Gulf of Mexico EEZ is unknown. There are insufficient data to determine the population trends for this stock.

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COMMON BOTTLENOSE DOLPHIN (*Tursiops truncatus truncatus*): West Bay Stock

NOTE – NMFS is in the process of writing individual stock assessment reports for each of the 31 bay, sound and estuary stocks of common bottlenose dolphins in the Gulf of Mexico. 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 estuarine site where photographic identification (photo-ID) or tagging studies have been conducted in the Gulf of Mexico (e.g., Irvine and Wells 1972; Shane 1977; Gruber 1981; Irvine *et al.* 1981; Wells 1986; Wells *et al.* 1987; Scott *et al.* 1990; Shane 1990; Wells 1991; Bräger 1993; Bräger *et al.* 1994; Fertl 1994; Wells *et al.* 1996a,b; Wells *et al.* 1997; Weller 1998; Maze and Würsig 1999; Lynn and Würsig 2002; Wells 2003; Hubard *et al.* 2004; Irwin and Würsig 2004; Shane 2004; Balmer *et al.* 2008; Urian *et al.* 2009; Bassos-Hull *et al.* 2013; Wells *et al.* 2017; Balmer *et al.* 2018). In many cases, residents occur predominantly within estuarine waters, with limited movements through passes to the Gulf of Mexico (Shane 1977; Gruber 1981; Irvine *et al*. 1981; Shane 1990; Maze and Würsig 1999; Lynn and Würsig 2002; Fazioli *et al.* 2006; Bassos-Hull *et al.* 2013; Wells *et al.* 2017). Genetic data also support the presence of discrete BSE stocks (Duffield and Wells 2002; Sellas *et al.* 2005; Rosel *et al.* 2017). 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 genetic population differentiation among all areas. Genetic data also indicate restricted genetic exchange between and demographic independence of BSE populations and those occurring in adjacent Gulf coastal waters (Sellas *et al.* 2005; Rosel *et al.* 2017). Differences in reproductive seasonality from site to site also suggest genetic-based distinctions among areas (Urian *et al.* 1996). Photo-ID and genetic data from several inshore areas of the southeastern United States also support the existence of resident estuarine animals and differentiation between animals biopsied along the Atlantic coast and those biopsied within estuarine systems at the same latitude (Caldwell 2001; Gubbins 2002; Zolman 2002; Mazzoil *et al.* 2005; Litz 2007; Rosel *et al.* 2009).

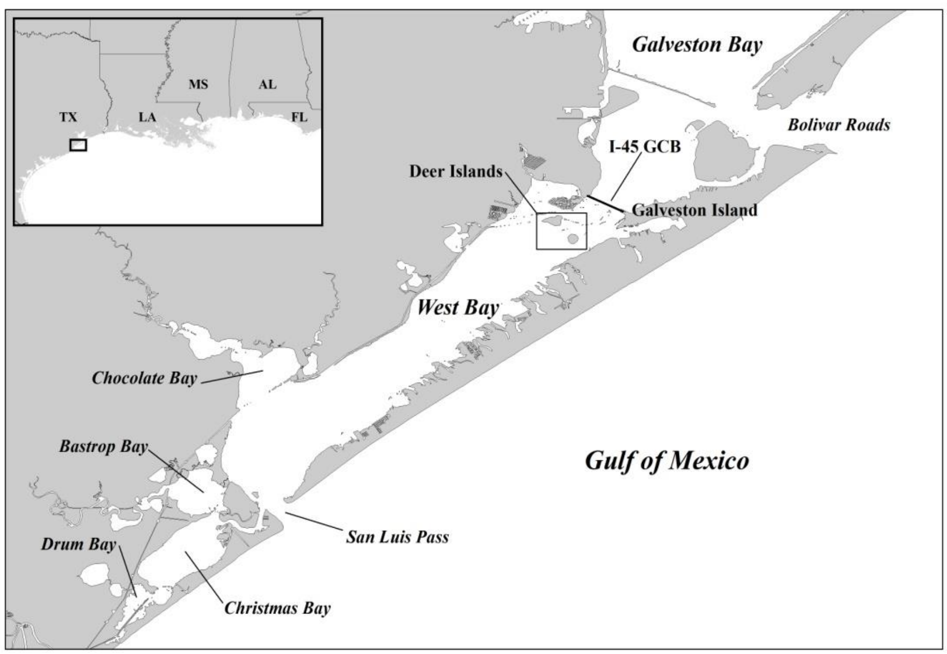

Figure 1. Geographic extent of the West Bay Stock, located within the Galveston Bay Estuary in Texas. I-45 GCB = I-45 Galveston Causeway Bridge.

West Bay, a bay within the Galveston Bay Estuary system, encompasses an area of approximately 180 km^2, and is a narrow, long bay averaging 1.2 m in depth (Diener 1975; Phillips and Rosel 2014; Figure 1). It tends to be more saline than Galveston Bay, with an average salinity of 15 to 32 ppt (Pulich and White 1991; Phillips and Rosel 2014). West Bay is separated from the Gulf of Mexico by Galveston Island, and connected to the Gulf via San Luis Pass in the southwest, and connected to Galveston Bay in the northeast via Bolivar Roads. The Galveston Bay Estuary has been selected as an estuary of national significance by the Environmental Protection Agency National Estuary Program (see http://www.gbep.state.tx.us/). Thus, a comprehensive conservation and management plan has been developed and is being implemented through a partnership of local, state, and federal representatives as well as community stakeholders, to restore and protect the estuary (Lester and Gonzalez 2011).

The West Bay Stock was designated in the first stock assessment reports published in 1995 (Blaylock *et al.* 1995) and common bottlenose dolphins are present within the bay. The stock boundaries extend from Drum Bay in the southwest to the I-45 Galveston Causeway Bridge in the northeast and include West Bay, Chocolate Bay, Bastrop Bay, Christmas Bay, Drum Bay, and San Luis Pass (Figure 1). However, Bastrop Bay, Christmas Bay, and Drum Bay are very shallow areas, and dolphins were not sighted there during recent exploratory surveys (Ronje *et al.* 2018). The area between the Deer Islands and the I-45 Galveston Causeway Bridge is being included in the West Bay Stock due to sightings of two animals that were also seen in southern West Bay (Litz *et al.* 2019), but this area may serve as a transition zone between the Galveston Bay/East Bay/Trinity Bay Stock and the West Bay Stock. Additional research may result in a revision to the northeastern boundary. Dolphins of this stock also are seen in nearshore coastal waters adjacent to San Luis Pass, where they may be exposed to additional threats. However, the extent to which they use these waters and whether there may be significant seasonality to that usage is unknown. To date, coastal waters approximately 3 km north and south of San Luis Pass and within 1 km of shore are included in the stock area. This coastal range is based on sightings data from a 2014–2015 photo-ID mark recapture survey (see Population Size). The range in coastal waters may be revised as new studies are conducted. Given the small size and relatively homogeneous habitat of West Bay, it is unlikely this stock contains multiple demographically independent populations, but a directed investigation of this question has never been conducted.

POPULATION SIZE

The best available abundance estimate for the West Bay Stock of common bottlenose dolphins is 37 (CV=0.05; Table 1), which is the result of vessel-based capture-recapture photo-ID surveys conducted during winter 2014 and summer 2015 (Ronje *et al.* 2020).

Earlier Abundance Estimates (>8 years old)

Boat-based photo-ID surveys in 1995 and 1996 conducted in southwestern West Bay, Chocolate Bay, San Luis Pass (SLP) and adjacent Gulf coastal waters outside SLP identified 28 year-round residents that utilized the bays, SLP, and nearshore coastal waters adjacent to SLP. During the summer dolphins were most frequently sighted furthest inland, mainly in Chocolate Bay, whereas during winter, sightings were concentrated near San Luis Pass and adjacent Gulf of Mexico coastal waters. In addition to resident animals, transient animals were sighted in Gulf coastal waters only (Maze and Würsig 1999). Additional boat-based surveys were conducted within the same area during 1997–2001 by Irwin and Würsig (2004) to compare three methods of assessing abundance: 1) counts based on photo-ID data; 2) capture-recapture analysis based on photo-ID data; and 3) line-transect surveys to estimate density using the program DISTANCE (Buckland *et al.* 1993). Photo-ID results based on counts yielded 34 resident animals displaying seasonal variation in their habitat use as described above. Capture-recapture analysis estimates of dolphin abundance in each year in warm months ranged from 28 (95% CI: 26–71) in 1998 to a high of 38 (95% CI: 33–55) in 2000. Line-transect density estimates ranged from 0.94 to 1.01 dolphins/km^2, with a warm-month abundance estimate of 108 dolphins (95% CI: 33-358). Irwin and Würsig (2004) suggested their density estimates were positively biased compared to estimates from other locations because the nonrandom distribution of dolphins in the study area makes the area unsuitable for line-transect surveys.

Recent Surveys and Abundance Estimates

Photo-ID capture-recapture surveys were conducted in two seasons (December 2014 and June 2015) with three surveys per season (Litz *et al.* 2019). The surveys covered the entirety of this stock's range including West Bay, Chocolate Bay, and San Luis Pass. Christmas Bay was surveyed in the summer but not the winter; there were no sightings in this bay. In addition, two 20-km segments of trackline were surveyed in the coastal waters off San Luis Pass (1 km from shore and 2 km from shore) (Litz *et al.* 2019). A Poisson-log normal Mark-Resight model

(McClintock *et al.* 2009) was used to estimate abundance for each season using MARK 8.2 (White and Burnham 1999). Six coastal sightings presumed to contain coastal stock animals (primarily 1–2 sightings of each animal and only in coastal waters) were removed from the analyses (Litz *et al.* 2019). The abundance estimate for winter (December 2014) was 51 dolphins (CV=0.04; 95% CI: 47–56) and the summer (June 2015) estimate was 44 dolphins (CV= 0.03; 95% CI: 43–47), and the mean of the estimates was 48 (CV=0.03; 95% CI: 45–50). The summer and winter estimates were averaged because there were no clear seasonal patterns in sighting distributions (Litz *et al.* 2019; Ronje *et al.* 2018). These estimates were corrected for the proportion of unmarked individuals. Capture probabilities were high for both seasons, and resighting data allowed for the exclusion of sightings of coastal stock animals from the abundance estimate. A key uncertainty is the possibility that coastal stock dolphins were present in estuarine waters and therefore could not be completely excluded from the abundance estimate.

Ronje *et al.* (2020) combined the West Bay survey data published by Litz *et al.* (2019) with data from two other study sites, Sabine Lake and Galveston Bay, into a single photo-ID catalog to compare inter-bay movements and incorporated results from that comparison when estimating abundance for each bay. As a part of this broader study, Ronje *et al.* (2020) also re-scored fin distinctiveness for the West Bay catalog of Litz *et al.* (2019) for consistency across study site catalogs, excluded dolphins that were sighted in more than one study site from analyses, and also used only on-effort sightings. Data were analyzed in MARK 9.0 (White and Burnham 1999) using the closed capture Huggins' p and c conditional likelihood approach and each season was analyzed independently. Using the selective dataset that included animals sighted only in coastal waters if sighted in both summer and winter seasons, and that removed animals sighted in more than one study site, (see Ronje *et al.* 2020), estimates for West Bay were 38 (CV=0.10; 95% CI: 29–47) in winter and 37 in summer (CV=0.02; CI: 33–40), and the mean of the estimates was 37 (CV=0.05). The summer and winter estimates were averaged because there were no clear seasonal patterns (Litz *et al.* 2019; Ronje *et al.* 2020). These estimates were corrected for the proportion of unmarked individuals (see Ronje *et al.* 2020).

The best estimate for the West Bay Stock is considered to be the average of the winter 2014 and summer 2015 estimates, 37 (CV=0.05; Table 1), as presented by Ronje *et al.* (2020). This is the most conservative estimate because it excluded animals sighted in more than one study area.

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 37 (CV=0.05). The minimum population estimate for the West Bay Stock is 35 common bottlenose dolphins (Table 1).

Current Population Trend

A population trend analysis has not been conducted for this stock. Older abundance estimates exist but data need to be examined for comparability to the 2014–2015 estimate.

CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

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

POTENTIAL BIOLOGICAL REMOVAL

Potential Biological Removal (PBR) is the product of the minimum population size, one-half the maximum productivity rate, and a recovery factor (MMPA Sec. 3. 16 U.S.C. 1362; Wade and Angliss 1997; Wade 1998). The minimum population size of the West Bay Stock of common bottlenose dolphins is 35. 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 Texas BSE stocks is greater than 0.8 (Wade and Angliss 1997). PBR for this stock of common bottlenose dolphins is 0.3 (Table 1).

Table 1. Best and minimum abundance estimates for the West Bay Stock of common bottlenose dolphins with Maximum Productivity Rate (Rmax), Recovery Factor (Fr) and PBR.

Nest	Nest CV	Nmin	Fr	Rmax	PBR
37	0.05	35	0.4	0.04	0.3

ANNUAL HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

The total annual human-caused mortality and serious injury for the West Bay Stock of common bottlenose dolphins during 2015–2019 is unknown. Across all Texas BSE stocks, the total annual estimated mortality for the shrimp trawl fishery was 0.4 (CV=1.62), but the portion of this attributed to the West Bay Stock is unknown (see Shrimp Trawl section). There were no recorded fishery-related mortalities or serious injuries during 2015–2019 based on strandings and at-sea observations. In addition, there were no recorded mortalities or serious injuries during 2015–2019 due to other human-caused sources. Therefore, the total mean annual human-caused mortality and serious injury for this stock during 2015–2019 was 0 (Table 2). However, the true value is likely non-zero because 1) not all fisheries that could interact with this stock are observed and/or observer coverage is very low, 2) stranding data are used as an indicator of fishery-related interactions and not all dead animals are recovered by the stranding network (Peltier *et al*. 2012; Wells *et al*. 2015), 3) cause of death is not (or cannot be) routinely determined for stranded carcasses, 4) the estimate of fishery-related interactions includes an actual count of verified fishery-caused deaths and serious injuries and should be considered a minimum (NMFS 2016), and 5) the estimate does not include shrimp trawl bycatch (see Shrimp Trawl section).

Fishery Information

There are three commercial fisheries that interact, or that potentially could interact, with this stock. These include one Category II fishery (Southeastern U.S. Atlantic, Gulf of Mexico shrimp trawl) and two Category III fisheries (Gulf of Mexico blue crab trap/pot; and Atlantic Ocean, Gulf of Mexico, Caribbean commercial passenger fishing vessel (hook and line)). Detailed fishery information is presented in Appendix III.

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

Shrimp Trawl

Between 1997 and 2019, 13 common bottlenose dolphins and nine 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 observed trips of the commercial shrimp trawl fishery in the Gulf of Mexico (Soldevilla *et al.* 2021). 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; 2021) provided mortality estimates calculated from analysis of shrimp fishery effort data and NMFS's Observer Program bycatch data. Mandated 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 2015–2019 based on inshore fishing effort (Soldevilla *et al.* 2021). Because the spatial resolution at which fishery effort is modeled is aggregated at the state level (e.g., Nance *et al.* 2008), the mortality estimate covers inshore waters of Texas from Galveston Bay, East Bay, Trinity Bay south to Laguna Madre. The mean annual mortality estimate for Texas BSE stocks for the years 2015–2019 was 0.4 (CV=1.62; Soldevilla *et al.* 2021). Limitations and biases of annual bycatch mortality estimates are described in detail in Soldevilla *et al.* (2015; 2016; 2021).

Blue Crab Trap/Pot

During 2015–2019, there were no documented interactions between commercial blue crab trap/pot gear and the West Bay Stock. There is no observer coverage of crab trap/pot fisheries in the Gulf of Mexico, so it is not possible to quantify total mortality.

Hook and Line (Rod and Reel)

During 2015–2019, there were no documented interactions between hook and line gear and the West Bay Stock

(NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 25 August 2020; Table 2). The most recent take occurred in 2014. It is not possible to estimate the total number of interactions with hook and line gear because there is no observer program in the Gulf of Mexico.

Other Mortality

NOAA's Office of Law Enforcement has been investigating increased reports from along the northern Gulf of Mexico coast of violence against common bottlenose dolphins, including shootings via guns and bows and arrows, pipe bombs and cherry bombs, and stabbings (Vail 2016). From recent cases that have been prosecuted, it has been shown that fishermen become frustrated and retaliate against dolphins for removing bait or catch, or depredating, their fishing gear. To date there are no records of acts of intentional harm for this stock area.

Depredation of fishing catch and/or bait is a growing problem in Gulf of Mexico coastal and estuary waters and globally, and can lead to serious injury or mortality via ingestion of or entanglement in gear (e.g., Zollett and Read 2006; Read 2008; Powell and Wells 2011; Vail 2016), as well as changes in dolphin activity patterns, such as decreases in natural foraging (Powell and Wells 2011). It has been suggested that provisioning, or the illegal feeding, of wild common bottlenose dolphins, may encourage depredation because provisioning conditions dolphins to approach humans and vessels, where they then may prey on bait and catches (Vail 2016). Such conditioning subsequently increases risks of injury and mortality (Christiansen *et al.* 2016). Provisioning has been documented in the literature in Florida and Texas (Bryant 1994; Samuels and Bejder 2004; Cunningham-Smith *et al.* 2006; Powell and Wells 2011). To date there are no records within the literature of provisioning for this stock area.

All mortalities and serious injuries from known sources for the West Bay Stock are summarized in Table 2.

Table 2. Summary of the incidental mortality and serious injury of common bottlenose dolphins (Tursiops truncatus) of the West Bay Stock. For the shrimp trawl fishery, the bycatch mortality for the West Bay Stock alone cannot be quantified at this time because mortality estimates encompass all estuarine waters of Texas pooled (see Shrimp Trawl section). The remaining fisheries do not have an ongoing, 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 strandings, at-sea counts, and fisherman self-reported takes, the number reported is a minimum because not all strandings, at-sea cases, or gear interactions are detected. See the Annual Human-Caused Mortality and Serious Injury section for biases and limitations of mortality estimates, and the Strandings section for limitations of stranding data. 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	2015–2019	Observer Data	Undetermined for this stock but may be non-zero (see Shrimp Trawl section)	NA
Atlantic Blue Crab Trap/Pot	2015–2019	Stranding Data and At-Sea Observations	NA	0
Hook and Line	2015–2019	Stranding Data and At-Sea Observations	NA	0
Mean Annual Mortality due to commercial fisheries (2015–2019)			**0**	
Research Takes (5-year Count)			0	

Other Takes (5-year Count)	0
Mean Annual Mortality due to research and other takes (2015–2019)	**0**
Minimum Total Mean Annual Human-Caused Mortality and Serious Injury (2015–2019)	**0**

Strandings

During 2015–2019, six common bottlenose dolphins were reported stranded within the West Bay area (Table 3; NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 25 August 2020). No evidence of human interaction (HI) was detected for four strandings, and for the remaining two strandings, it could not be determined if there was evidence of human interaction.

The assignment of animals to a single stock is impossible in some regions where stocks overlap, especially in nearshore coastal waters (Maze-Foley *et al.* 2019). Of the six strandings ascribed to the West Bay Stock, three were ascribed solely to this stock. It is likely, therefore, that the counts in Table 3 include some animals from the Western Coastal Stock and thereby overestimate the number of strandings for the West Bay Stock; those strandings that could not be definitively ascribed to the West Bay Stock were also included in the counts for the Western Coastal Stock 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).

There are a number of other difficulties associated with the interpretation of stranding data. Stranding data underestimate the extent of human and fishery-related mortality and serious injury because not all of the dolphins that die or are seriously injured in human interactions wash ashore, or, if they do, they are not all recovered (Peltier *et al*. 2012; Wells *et al*. 2015; Carretta *et al.* 2016). Additionally, not all carcasses will show evidence of human interaction, entanglement, or other fishery-related interaction due to decomposition, scavenger damage, etc. (Byrd *et al.* 2014). Finally, the level of technical expertise among stranding network personnel varies widely as does the ability to recognize signs of human interaction.

The West Bay Stock has likely been affected by five common bottlenose dolphin die-offs or Unusual Mortality Events (UMEs). 1) From January through May 1990, a total of 344 common bottlenose dolphins stranded in the northern Gulf of Mexico. Overall this represented a two-fold increase in the prior maximum recorded number of strandings for the same period in the northern Gulf of Mexico. The cause of the 1990 mortality event could not be determined (Hansen 1992), however, morbillivirus may have contributed to this event (Litz *et al.* 2014). One stranding occurred within West Bay and 25 others occurred along the ocean side of Galveston Island, some in the vicinity of West Bay, but the stock origin of those animals is unknown (Phillips and Rosel 2014). 2) In 1993–1994, a UME of common bottlenose dolphins caused by morbillivirus started in the Florida Panhandle and spread west with most of the mortalities occurring in Texas (Lipscomb 1993; Lipscomb *et al.* 1994; Litz *et al.* 2014). From February through April 1994, 236 common bottlenose dolphins were found dead on Texas beaches, of which 67 occurred in a single 10-day period. One stranding occurred within West Bay, and 51 others occurred along the ocean side of Galveston Island and may or may not have involved this stock (Phillips and Rosel 2014). 3) During February and March of 2007 a UME was declared for northeast Texas and western Louisiana involving 64 common bottlenose dolphins and two unidentified dolphins. Decomposition prevented conclusive analyses on most carcasses (Litz *et al.* 2014). Eighteen animals stranded along the ocean side of Galveston Island in the vicinity of West Bay, but the stock origin of the animals is unknown (Phillips and Rosel 2014). 4) During February and March of 2008 a UME was declared in Texas involving 111 common bottlenose dolphin strandings (plus strandings of one unidentified dolphin and one melon-headed whale, *Peponocephala electra*). Most of the animals recovered were in a decomposed state and a direct cause of the mortalities could not be identified. However, there were numerous, co-occurring harmful algal bloom toxins detected during the time period of this UME which may have contributed to the mortalities (Fire *et al.* 2011). Two strandings occurred within West Bay and 35 others occurred along the Gulf side of Galveston Island in the vicinity of West Bay, but the stock origin of the animals is unknown (Phillips and Rosel 2014). 5) A UME occurred from November 2011 to March 2012 across five Texas counties and included 126 common bottlenose dolphin strandings.

The strandings were coincident with harmful algal blooms of *Karenia brevis* and *Dinophysis sp.* The cause of the bottlenose dolphin UME was determined to be due to biotoxin exposure from brevetoxin and okadaic acid. The additional supporting evidence of fish kills and other species die-offs linked to brevetoxin during the same time and space made a strong case that the harmful algal blooms impacted the dolphins. Three animals from the West Bay Stock were considered to be part of the UME, and an additional 37 strandings occurred along the Gulf side of Galveston Island in the vicinity of West Bay, but the stock origin of the animals is unknown (Phillips and Rosel 2014).

Table 3. Common bottlenose dolphin strandings occurring in the West Bay Stock area from 2015 to 2019, 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 25 August 2020). Please note HI does not necessarily mean the interaction caused the animal's death.

Stock	Category	2015	2016	2017	2018	2019	Total
West Bay Stock	Total Stranded	0	2	0	3	1	6
	Human Interaction						
	---Yes	0	0	0	0	0	0
	---No	0	1	0	2	1	4
	---CBD	0	1	0	1	0	2

HABITAT ISSUES

The estuarine habitat occupied by this stock is adjacent to the highly populated and industrial areas of Houston and Galveston, Texas. The five coastal counties surrounding the Galveston Bay Estuary, which includes West Bay, have a population exceeding 5.4 million people as of January 1, 2018 (TDC 2019). This has been an area of continuous economic growth and development over most of the previous 50 years, with much of this growth attributed to the discovery of oil and the construction of the Houston Ship Channel (Lester and Gonzalez 2011).

There are over 3000 oil and natural gas production platforms in the five counties surrounding Galveston and West Bays, including pipelines for the transport of these products and many refining facilities (Lester and Gonzalez 2011). While most of the platforms are placed on the surrounding land in the West Bay area, several platforms reside in Chocolate Bay and the confluence of Chocolate Bay and West Bay (Lester and Gonzalez 2011). No major oil spills have occurred within West Bay itself, however, repeated spills, from minor to serious in nature, have occurred in the waters of Galveston Bay or in coastal waters off Galveston Island (see Phillips and Rosel 2014 for a summary). A recent oil spill in 2014, referred to as the Texas City Y incident, involved a vessel collision in Galveston Bay near Texas City and the subsequent release of approximately 168,000 gallons of intermediate fuel oil. Through the National Resource Damage Assessment (NRDA) process, impacts of this spill are currently being evaluated and will include impacts to common bottlenose dolphins of the West Bay Stock (NOAA DAARP 2018). No information is currently available on potential impacts to the West Bay Stock. In addition to being known as an area of oil and gas production, the area surrounding Galveston and West Bays produces more than 50% of all chemical products manufactured in the U.S. (Henningsen and Würsig 1991; Lester and Gonzalez 2011).

According to an agricultural census for 2007, over 7,700 farms consisting of >540,000 acres of cropland, were located within the five coastal counties surrounding the Galveston Bay Estuary (Lester and Gonzalez 2011). Raising of livestock is also common in this area. Agricultural impacts on West Bay include the introduction of pesticides, herbicides, and nutrients from crop management, as well as fecal coliform bacteria resulting from livestock waste (Lester and Gonzalez 2011). Due to high levels of fecal coliform bacteria, half of the Galveston Bay Estuary is provisionally or permanently closed to the harvesting of shellfish. Chocolate Bay and Bastrop Bay have been rated as "moderate" for bacterial contamination levels, and West Bay has been rated "good" with fewer than 10% of sampled sites exceeding threshold levels for coliform bacteria (Lester and Gonzalez 2011).

In addition to discharge from the petroleum and chemical refineries and facilities and agricultural sources and sewage, West Bay receives additional pollution from stormwater runoff and shipping traffic (Jackson *et al.* 1998; Santschi *et al.* 2001; Lester and Gonzalez 2011; Phillips and Rosel 2014). Analysis of sediment samples from Galveston and West Bays in 2009 and 2010 indicated low concentrations of heavy metals. However, in 2000, two sediment samples from West Bay exceeded safety thresholds for PCBs (lindane and chlordane) (Lester and Gonzalez 2011; Phillips and Rosel 2014). Heavy metal and chemical concentrations in sediments and fish tissues have historically been of concern, and advisories about seafood consumption have often been issued. For example, currently an advisory exists regarding catfish consumption in West Bay and Chocolate Bay due to concerns about dioxins and PCBs (TPWD 2020). Mercury concentrations from samples of blue crab, oysters, and finfish are typically below those considered to be of human health concern, however the second highest concentration of mercury within the Galveston Bay Estuary was measured in a sample of sheepshead collected in West Bay in 1999 (Lester and Gonzalez 2011; Phillips and Rosel 2014). Organic contaminants and trace metals have been monitored in oysters, and the resulting concentration of PCBs has typically surpassed the level for sub-lethal effects (Jackson *et al.* 1998; Phillips and Rosel 2014). The concentrations of lead found in oysters from West Bay and Back Bay (adjacent to West Bay, on the other side of the I-45 Galveston Causeway Bridge) have been higher than those reported from other sampling sites within the Galveston Bay Estuary (Jiann and Presley 1997). Polynuclear aromatic hydrocarbon (PAH) levels in Galveston Bay are higher than national levels and indicate contamination by petroleum products, industrial activities, and urban run-off (Qian *et al.* 2001; Phillips and Rosel 2014). Concentrations of chlorinated hydrocarbons and metals were examined in conjunction with an anomalous mortality event of common bottlenose dolphins in Texas bays (although not West Bay) in 1990 and found to be relatively low in most; however, some had concentrations at levels of possible toxicological concern (Varanasi *et al.* 1992).

Harmful algal blooms and low dissolved oxygen are habitat issues leading to fish kills almost annually in the summers for Galveston and West Bays (McInnes and Quigg 2010). For example, a fish kill occurred in 2005 near Galveston Island due to low dissolved oxygen and a cyanobacteria bloom, killing over 10,000 Gulf menhaden (Phillips and Rosel 2014). In August 2012, a bloom occurred killing approximately one million fish in Galveston and West Bays. Another *K. brevis* bloom occurred along the Texas coast during September 2011–January 2012 resulting in the temporary closure of all shellfish beds in Texas and fish kills in Galveston Bay (Phillips and Rosel 2014). Earlier algal blooms affecting West Bay and resulting in shellfish bed closures occurred in 1972, 1976, 1986, 1996, and 2000 (Magaña *et al.* 2003; Phillips and Rosel 2014). For the 2011–2012 UME mentioned above (Strandings section), the strandings were coincident with a large harmful algal bloom of *K. brevis*. The definitive cause of that event has not been determined, but the algal bloom could have contributed to the mortality event.

Loss of wetland habitat and seagrass beds, and fragmentation of these habitats, within West Bay is another important issue (Lester and Gonzalez 2011; Phillips and Rosel 2014). West Bay has suffered significant loss of wetland habitat since the 1950s, much through the conversion of wetlands to cropland. Subsidence is another leading cause of wetland loss, exacerbated by the removal of petroleum and groundwater in the area (Lester and Gonzalez 2011; Phillips and Rosel 2014). Seagrass beds have been lost due to a complex interaction of causes including shoreline development, dredging, subsidence, boat traffic, and severe storms (Lester and Gonzalez 2011). Conservation partners and resource managers have invested in habitat restoration efforts within West Bay and have begun to restore acres of intertidal marsh and seagrasses (Lester and Gonzalez 2011; Phillips and Rosel 2014).

Finally, West Bay and Galveston Bay experienced significant storm surge during Hurricane Ike in 2008. As a result, discussion and planning for improved coastal barriers to protect the region from storm surge is in the works. Part of this plan includes ecosystem restoration projects and possible construction of flood gates within the West Bay area (U.S. Army Corps of Engineers 2020).

STATUS OF STOCK

Common bottlenose dolphins are not listed as threatened or endangered under the Endangered Species Act, and the West Bay Stock is not a strategic stock under the MMPA. The total fishery-related mortality and serious injury for this stock is unknown. The minimum estimate of fishery-related mortality and serious injury is less than 10% of the calculated PBR, but there is insufficient information (see Annual Human-Caused Mortality and Serious Injury section) available to determine whether the total human-caused fishery-related mortality and serious injury is insignificant and approaching a zero mortality and serious injury rate. The status of this stock relative to optimum sustainable population is unknown and there are insufficient data to determine population trends for this stock.

Although this stock does not meet the criteria to qualify as strategic, NMFS has concerns regarding this stock due to the small stock size and the inability to determine the total human-caused mortality and serious injury.

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COMMON BOTTLENOSE DOLPHIN (*Tursiops truncatus truncatus*): Galveston Bay, East Bay, Trinity Bay Stock

NOTE – NMFS is in the process of writing individual stock assessment reports for each of the 31 bay, sound and estuary stocks of common bottlenose dolphins in the Gulf of Mexico. 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 estuarine site where photographic identification (photo-ID) or tagging studies have been conducted in the Gulf of Mexico (e.g., Irvine and Wells 1972; Shane 1977; Gruber 1981; Irvine *et al.* 1981; Wells 1986; Wells *et al.* 1987; Scott *et al.* 1990; Shane 1990; Wells 1991; Bräger 1993; Bräger *et al.* 1994; Fertl 1994; Wells *et al.* 1996a, 1996b; Wells *et al.* 1997; Weller 1998; Maze and Würsig 1999; Lynn and Würsig 2002; Wells 2003; Hubard *et al.* 2004; Irwin and Würsig 2004; Shane 2004; Balmer *et al.* 2008; Urian *et al.* 2009; Bassos-Hull *et al.* 2013; Wells *et al.* 2017; Balmer *et al.* 2018). In many cases, residents occur predominantly within estuarine waters, with limited movements through passes to the Gulf of Mexico (Shane 1977; Gruber 1981; Irvine *et al*. 1981; Shane 1990; Maze and Würsig 1999; Lynn and Würsig 2002; Fazioli *et al.* 2006; Bassos-Hull *et al.* 2013; Wells *et al.* 2017). However, several studies in the Bolivar Roads area of Galveston Bay, the primary entryway and ship channel into the Bay, have documented large numbers of dolphins using the deep-dredged channel and jetty habitat (e.g., Henningsen and Würsig 1991; Bräger 1993) and this area been shown to be a foraging "hotspot" (Moreno and Matthews 2018). How much movement of dolphins occurs from Bolivar Roads into the upper parts of the Bay is unknown.

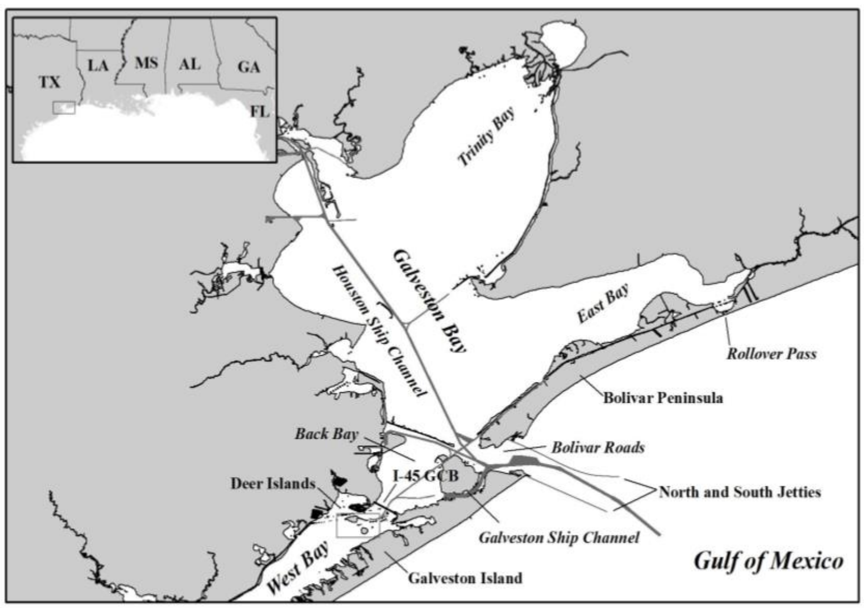

Figure 1. Geographic extent of the Galveston Bay, East Bay, Trinity Bay Stock, located on the northeast coast of Texas. I-45 GCB = I-45 Galveston Causeway Bridge.

Genetic data also support the concept of relatively discrete, demographically independent BSE populations in the Gulf of Mexico (Duffield and Wells 2002; Sellas *et al.* 2005; Rosel *et al.* 2017). 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 genetic population differentiation among all areas. Genetic data also indicate restricted genetic exchange between and demographic independence of BSE populations and those occurring in adjacent Gulf coastal waters (Sellas *et al.* 2005; Rosel *et al.* 2017). Photo-ID and genetic data from several inshore areas of the southeastern United States Atlantic coast also support the existence of resident estuarine animals and differentiation between animals biopsied along the Atlantic coast and those biopsied within estuarine systems at the same latitude (Caldwell 2001; Gubbins 2002; Zolman

2002; Mazzoil *et al.* 2005; Litz 2007; Rosel *et al.* 2009).

The Galveston Bay, East Bay, Trinity Bay stock area is part of the Galveston Bay Estuary, a large, shallow estuary located in northeast Texas. Encompassing a surface area of ~1,399 km^2, the estuary averages 2 m in depth (USEPA 1999; Phillips and Rosel 2014), but also includes dredged channels up to 15 m deep used for commercial navigation (Moreno and Matthews 2018; Ronje *et al.* 2018). During times of normal freshwater flow into the system (not drought or flood conditions), salinity ranges from less than 10 psu in Upper Trinity Bay to ~30 psu at Bolivar Roads (Lester and Gonzalez 2011). Galveston Bay, East Bay, and Trinity Bay are separated from the Gulf of Mexico by Galveston Island and Bolivar Peninsula, and connected to the Gulf via Bolivar Roads, also known as Bolivar Pass, and also Rollover Pass, a man-made pass through Bolivar Peninsula (Phillips and Rosel 2014; Figure 1). There are also north and south granite rock jetties extending from the Bolivar Peninsula and Galveston Island, respectively, 3 km into the Gulf of Mexico (Ronje *et al.* 2018). The Houston Ship Channel runs within Bolivar Roads, and the Galveston Ship Channel intersects Bolivar Roads. The Galveston Bay Estuary has been selected as an estuary of national significance by the Environmental Protection Agency National Estuary Program (see http://www.gbep.state.tx.us/). Thus, a comprehensive conservation and management plan has been developed and is being implemented through a partnership of local, state, and federal representatives as well as community stakeholders, to restore and protect the estuary (Lester and Gonzalez 2011).

The Galveston Bay, East Bay, Trinity Bay Stock of common bottlenose dolphins was designated in the first stock assessment reports published in 1995 (Blaylock *et al.* 1995). The stock boundaries extend from the I-45 Galveston Causeway Bridge in the southwest and includes Galveston Bay, East Bay, Trinity Bay, Back Bay, the Galveston Ship Channel, Bolivar Roads/Bolivar Pass (the area in between the jetties), and coastal waters 1 km around the jetties and 2 km from shore extending for 5 km on each side of the jetties (Figure 1). A recent photo-identification capture-mark-recapture study (Ronjet *et al.* 2020) observed some individuals in both this coastal strip and inside Galveston Bay. Bolivar Roads appears to serve as a transition zone or mixing area between the Galveston Bay, East Bay, Trinity Bay Stock, the Western Coastal Stock, and potentially also the West Bay and Sabine Lake stocks (Ronje *et al.* 2020). The area between the Deer Islands and the I-45 Galveston Causeway Bridge is being included in the West Bay Stock due to sightings of two animals that were also seen in southern West Bay (Litz *et al.* 2019), but this area may serve as a transition zone between the Galveston Bay, East Bay, Trinity Bay Stock and the West Bay Stock. Additional research may result in a revision to the stock boundaries. Photo-ID data indicate distinct ranging and habitat usage patterns (e.g., Galveston Ship Channel, Fertl 1994; Upper Galveston Bay, Fazioli and Mintzer 2020), suggesting that the stock may contain multiple demographically independent populations.

POPULATION SIZE

The best available abundance estimate for the Galveston Bay, East Bay, Trinity Bay Stock of common bottlenose dolphins is 842 (CV=0.08; 95%CI: 694–990; Table 1), which is the result of vessel-based capture-recapture photo-ID surveys conducted during winter (January) 2016 (Ronje *et al.* 2020).

Recent Surveys and Abundance Estimates

Photo-ID capture-recapture surveys were conducted in two seasons (winter (January) and summer (July) 2016) with three to four surveys per season (Ronje *et al.* 2018). The surveys covered the entirety of this stock's range including Galveston Bay, East Bay, Trinity Bay, Back Bay, the Galveston Ship Channel, and Bolivar Roads. In addition, two 20-km segments of trackline were surveyed in the coastal waters north and south of Bolivar Roads (500 m from shore and 2 km from shore; Ronje *et al.* 2018). Ronje *et al.* (2020) combined these survey data with data from two other study sites, Sabine Lake and West Bay, into a single catalog to compare inter-bay movements and incorporated results from that comparison when estimating abundance for each bay. As a part of this broader study, Ronje *et al.* (2020) excluded dolphins that were sighted in more than one study site from analyses. Data were analyzed with MARK 9.0 software (White and Burnham 1999) using the closed capture Huggins' p and c conditional likelihood approach and each season was analyzed independently. Using the selective dataset that included animals sighted only in coastal waters if sighted in both summer and winter seasons, and that removed animals sighted in more than one study site (see Ronje *et al.* 2020), estimates for Galveston Bay were 842 (CV=0.08; 95%CI: 694–990) in winter and 1,132 in summer (CV=0.13; 95%CI: 846–1,417). These estimates were corrected for the proportion of unmarked individuals.

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_{est} for this stock. The resulting best estimate for the Galveston Bay, East

Bay, Trinity Bay Stock is therefore the winter 2016 estimate, 842 (CV=0.08; 95%CI: 694–990; Table 1; Ronje *et al.* 2020). This is a conservative estimate because it excluded animals sighted in more than one study area.

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 842 (CV=0.08; 95%CI: 694–990). The minimum population estimate for the Galveston Bay, East Bay, Trinity Bay Stock is 787 common bottlenose dolphins (Table 1).

Current Population Trend

There are insufficient data to assess population trends for this stock.

CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

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

POTENTIAL BIOLOGICAL REMOVAL

Potential Biological Removal (PBR) is the product of the minimum population size, one-half the maximum productivity rate, and a recovery factor (MMPA Sec. 3. 16 U.S.C. 1362; Wade and Angliss 1997; Wade 1998). The minimum population size of the Galveston Bay, East Bay, Trinity Bay Stock of common bottlenose dolphins is 787. 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 Texas BSE stocks is greater than 0.8 (Wade and Angliss 1997). PBR for this stock of common bottlenose dolphins is 6.3 (Table 1).

Table 1. Best and minimum abundance estimates for the Galveston Bay, East Bay, Trinity Bay Stock of common bottlenose dolphins with Maximum Productivity Rate (R_{max}), Recovery Factor (F_r) and PBR.

N_{est}	N_{est} CV	N_{min}	F_r	R_{max}	PBR
842	0.08	787	0.4	0.04	6.3

ANNUAL HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

The total annual human-caused mortality and serious injury for the Galveston Bay, East Bay, Trinity Bay Stock of common bottlenose dolphins during 2015–2019 is unknown. Across all Texas BSE stocks, the total annual estimated mortality for the shrimp trawl fishery was 0.4 (CV=1.62), but the portion of this attributed to the Galveston Bay, East Bay, Trinity Bay Stock is unknown (see Shrimp Trawl section). The mean annual fishery-related mortality and serious injury during 2015–2019 based on strandings and at-sea observations identified as fishery-related was 0.4. Additional mean annual mortality and serious injury during 2015–2019 due to other human-caused sources was 0.6. The minimum total mean annual human-caused mortality and serious injury for this stock during 2015–2019 was therefore 1.0 (Table 2). This is considered 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 of fishery-related interactions includes an actual count of verified fishery-caused deaths and serious injuries and should be considered a minimum (NMFS 2016), and 5) the estimate does not include shrimp trawl bycatch (see Shrimp Trawl section).

Fishery Information

There are four commercial fisheries that interact, or that potentially could interact, with this stock. These include one Category II fishery (Southeastern U.S. Atlantic, Gulf of Mexico shrimp trawl) and three Category III fisheries (U.S. Atlantic, Gulf of Mexico trotline; Gulf of Mexico blue crab trap/pot; and Atlantic Ocean, Gulf of Mexico, Caribbean commercial passenger fishing vessel (hook and line)). Detailed fishery information is presented in Appendix III.

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

Shrimp Trawl

Between 1997 and 2019, 13 common bottlenose dolphins and nine 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 observed trips of the commercial shrimp trawl fishery in the Gulf of Mexico (Soldevilla *et al.* 2021). 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, 2021) provided mortality estimates calculated from analysis of shrimp fishery effort data and NMFS's Observer Program bycatch data. Mandated 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 2015–2019 based on inshore fishing effort (Soldevilla *et al.* 2021). Because the spatial resolution at which fishery effort is modeled is aggregated at the state level (e.g., Nance *et al.* 2008), the mortality estimate covers inshore waters of Texas from Galveston Bay, East Bay, Trinity Bay south to Laguna Madre. The mean annual mortality estimate for Texas BSE stocks for the years 2015–2019 was 0.4 (CV=1.62; Soldevilla *et al.* 2021). Limitations and biases of annual bycatch mortality estimates are described in detail in Soldevilla *et al.* (2015, 2016, 2021).

Trotline

During 2015–2019, one entanglement interaction between commercial trotline gear and the Galveston Bay, East Bay, Trinity Bay Stock was documented in the stranding database (NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 25 August 2020; Table 3). The entanglement occurred during 2018 and resulted in a mortality. There is no observer coverage of trotline fisheries in the Gulf of Mexico, so it is not possible to quantify total mortality. The documented interaction in this gear represents a minimum known count of interactions in the last five years.

Blue Crab Trap/Pot

During 2015–2019, there were no documented interactions between commercial blue crab trap/pot gear and the Galveston Bay, East Bay, Trinity Bay Stock. There is no observer coverage of crab trap/pot fisheries in the Gulf of Mexico, so it is not possible to quantify total mortality.

Hook and Line (Rod and Reel)

During 2015–2019, there were two at-sea observations of dolphins entangled in monofilament line. One occurred during 2015, and this animal was considered not seriously injured. The second case occurred during 2019, and this animal was considered to be seriously injured (Maze-Foley and Garrison 2021). The 2019 serious injury was included in the annual human-caused mortality and serious injury total for this stock (Table 2).

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

Other Mortality

One mortality was documented in 2018 in the Galveston Bay, East Bay, Trinity Bay Stock area as a result of an incidental entanglement in a fishery research gillnet. An additional interaction was documented in 2017 involving a live animal entangled in unidentified rope/line, and the animal was considered seriously injured. Both of these interactions were included in the stranding database (NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 25 August 2020) and in the annual human-caused mortality and serious injury total for this stock (Table 2).

During 2015–2019, there were two at-sea observations, one during 2015 and one during 2016, in Galveston Bay, of dolphins entangled in unidentified debris and gear. One of these animals (2016) was considered seriously injured (Maze-Foley and Garrison 2020), and it was included in the annual human-caused mortality and serious injury total for this stock (Table 2).

NOAA's Office of Law Enforcement has been investigating increased reports from along the northern Gulf of Mexico coast of violence against common bottlenose dolphins, including shootings via guns and bows and arrows, pipe bombs and cherry bombs, and stabbings (Vail 2016). From recent cases that have been prosecuted, it has been shown that fishermen become frustrated and retaliate against dolphins for removing bait or catch, or depredating their fishing gear. To date, there are no records of acts of intentional harm for this stock area.

Depredation of fishing catch and/or bait is a growing problem in Gulf of Mexico coastal and estuary waters and globally, and can lead to serious injury or mortality via ingestion of or entanglement in gear (e.g., Zollett and Read 2006; Read 2008; Powell and Wells 2011; Vail 2016), as well as changes in dolphin activity patterns, such as decreases in natural foraging (Powell and Wells 2011). It has been suggested that provisioning, or the illegal feeding, of wild common bottlenose dolphins, may encourage depredation because provisioning conditions dolphins to approach humans and vessels, where they then may prey on bait and catches (Vail 2016). Such conditioning increases risks of subsequent injury and mortality (Christiansen *et al.* 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 Galveston Bay, East Bay, Trinity Bay Stock are summarized in Table 2.

Table 2. Summary of the incidental mortality and serious injury of common bottlenose dolphins (Tursiops truncatus) of the Galveston Bay, East Bay, Trinity Bay Stock. For the shrimp trawl fishery, the bycatch mortality for the Galveston Bay, East Bay, Trinity Bay Stock alone cannot be quantified at this time because mortality estimates encompass all estuarine waters of Texas pooled (see Shrimp Trawl section). The remaining fisheries do not have an ongoing, 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 strandings, at-sea counts, and fisherman self-reported takes, the number reported is a minimum because not all strandings, at-sea cases, or gear interactions are detected. See the Annual Human-Caused Mortality and Serious Injury section for biases and limitations of mortality estimates, and the Strandings section for limitations of stranding data. 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	2015–2019	Observer Data	Undetermined for this stock but may be non-zero (see Shrimp Trawl section)	NA
Trotline	2015–2019	Stranding Data and At-Sea Observations	NA	1
Atlantic Blue Crab Trap/Pot	2015–2019	Stranding Data and At-Sea Observations	NA	0
Hook and Line	2015–2019	Stranding Data and At-Sea Observations	NA	1
Mean Annual Mortality due to commercial fisheries (2015–2019)			**0.4**	
Research Takes (5-year Count)			1	

Other Takes (5-year Count)	2
Mean Annual Mortality due to research and other takes (2015–2019)	**0.6**
Minimum Total Mean Annual Human-Caused Mortality and Serious Injury (2015–2019)	**1.0**

Strandings

During 2015–2019, 124 common bottlenose dolphins were reported stranded within the Galveston Bay, East Bay, Trinity Bay Stock area (Table 3; NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 25 August 2020). There was evidence of human interaction (HI) for 11 of the strandings. No evidence of human interaction was detected for 13 strandings, and for the remaining 100 strandings, it could not be determined if there was evidence of human interaction (Table 3). Human interactions were from numerous sources, including an entanglement in commercial trotline gear, an incidental take in a research gillnet, three animals with evidence of a vessel strike, and an entanglement with unidentified rope/line (Table 3). It should be noted that evidence of human interaction does not necessarily mean the interaction caused the animal's stranding or death.

The assignment of animals to a single stock is impossible in some regions where stocks overlap, especially in nearshore coastal waters (Maze-Foley *et al.* 2019). Of the 124 strandings ascribed to the Galveston Bay, East Bay, Trinity Bay Stock, 88 were ascribed solely to this stock. It is likely, therefore, that the counts in Table 3 include some animals from the Western Coastal Stock and thereby overestimate the number of strandings for the Stock; those strandings that could not be definitively ascribed to the Galveston Bay, East Bay, Trinity Bay Stock were also included in the counts for the Western Coastal Stock 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).

There are a number of other difficulties associated with the interpretation of stranding data. Stranding data underestimate the extent of human and fishery-related mortality and serious injury because not all of the dolphins that die or are seriously injured in human interactions wash ashore, or, if they do, they are not all recovered (Peltier *et al*. 2012; Wells *et al*. 2015; Carretta *et al.* 2016). 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 Galveston Bay, East Bay, Trinity Bay Stock has likely been affected by five common bottlenose dolphin die-offs or Unusual Mortality Events (UMEs). 1) From January through May 1990, a total of 344 common bottlenose dolphins stranded in the northern Gulf of Mexico. Overall this represented a two-fold increase in the prior maximum recorded number of strandings for the same period in the northern Gulf of Mexico. The cause of the 1990 mortality event could not be determined (Hansen 1992), however, morbillivirus may have contributed to this event (Litz *et al.* 2014). Three strandings occurred within Galveston Bay and one occurred in the ship channel just outside Galveston Bay. An additional 14 others stranded along the ocean side of Galveston Island and Bolivar Peninsula, but the stock origin of those animals is unknown (Phillips and Rosel 2014). 2) In 1993–1994, a UME of common bottlenose dolphins caused by morbillivirus started in the Florida Panhandle and spread west with most of the mortalities occurring in Texas (Lipscomb 1993; Lipscomb *et al.* 1994; Litz *et al.* 2014). From February through April 1994, 236 common bottlenose dolphins were found dead on Texas beaches, of which 67 occurred in a single 10-day period. Four strandings occurred within Galveston Bay, and 26 others occurred along the ocean side of Galveston Island and Bolivar Peninsula, but the stock origin of those animals is unknown (Phillips and Rosel 2014). 3) During February and March of 2007 a UME was declared for northeast Texas and western Louisiana involving 64 common bottlenose dolphins and two unidentified dolphins. Decomposition prevented conclusive analyses on most carcasses (Litz *et al.* 2014). One stranding occurred within Galveston Bay and one occurred in East Bay. Most of the other strandings occurred along the ocean side of Galveston Island or Bolivar Peninsula, but the stock origin of the animals is unknown (Phillips and Rosel 2014). 4) During February and March of 2008 a UME was declared in Texas involving 111 common bottlenose dolphin strandings (plus strandings of one unidentified dolphin and one melon-headed whale, *Peponocephala electra*). Most of the animals recovered were in a decomposed state and a direct cause of the mortalities could not be identified. However, there were numerous, co-occurring harmful algal bloom toxins detected during the time period of this UME which may have contributed to the mortalities (Fire *et al.* 2011). Twenty-four strandings occurred along the Gulf side of Galveston Island and Bolivar Peninsula in the vicinity of Galveston Bay, but the stock origin of the animals is unknown (Phillips and Rosel 2014). 5) A UME occurred from November 2011 to March 2012 across five Texas counties and included 126 common bottlenose dolphin strandings. The strandings were coincident with harmful algal blooms of *Karenia brevis* and *Dinophysis* sp. The cause of the bottlenose dolphin UME was determined to be due to biotoxin exposure from brevetoxin and okadaic acid. The additional supporting evidence of fish kills and other species die-offs linked to brevetoxin during the same time and space made a strong case that the harmful algal blooms impacted the dolphins. Three animals stranded within Galveston Bay and were considered to be part of the UME, and an additional 14 strandings occurred along the Gulf side of Galveston Island and Bolivar Peninsula in the vicinity of Galveston Bay, but the stock origin of the animals is unknown (Phillips and Rosel 2014).

Table 3. Common bottlenose dolphin strandings occurring in the Galveston Bay, East Bay, Trinity Bay Stock area from 2015 to 2019, 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 25 August 2020). Please note HI does not necessarily mean the interaction caused the animal's death.

Stock	Category	2015	2016	2017	2018	2019	Total
Galveston Bay, East Bay, Trinity Bay Stock	Total Stranded	18	19	31	26	30	124
	Human Interaction						
	---Yes	2	1[a]	1[b]	6[c]	1[d]	11
	---No	0	3	3	2	5	13
	---CBD	16	15	27	18	24	100

a. An animal with evidence of a vessel strike (mortality).
b. An entanglement interaction with unidentified rope/line (alive, seriously injured).
c. Includes 1 entanglement interaction in commercial trotline gear (mortality), 1 entanglement interaction in research gillnet gear (mortality), and 1 animal with evidence of a vessel strike (mortality).
d. An animal with evidence of a vessel strike (mortality).

HABITAT ISSUES

The estuarine habitat occupied by this stock is adjacent to the highly populated and industrial areas of Houston and Galveston, Texas and experiences impacts from a variety of anthropogenic sources. This has been an area of continuous economic growth and development over most of the previous 50 years, with much of this growth attributed to the discovery of oil and the construction of the Houston Ship Channel (Lester and Gonzalez 2011). This area is important for transportation, containing three major deep-draft ports within Galveston Bay: Port of Houston, Port of Texas City, and Port of Galveston (see Phillips and Rosel 2014 for a summary). There are over 3,000 oil and natural gas production platforms in all parts of Galveston Bay and the counties surrounding Galveston and West Bays, including pipelines for the transport of these products and many refining facilities (Lester and Gonzalez 2011). Repeated oil spills, from minor to serious in nature, have occurred in the waters of Galveston Bay or in coastal waters off Galveston Island (see Phillips and Rosel 2014 for a summary). Additional impacts to the Bay include discharge from petroleum and chemical refineries and facilities, and agricultural sources (Phillips and Rosel 2014), including high levels of fecal coliform bacteria that have provisionally or permanently closed parts of the Bay to the harvesting of shellfish (Lester and Gonzalez 2011).

Direct impacts to the stock include a recent oil spill, freshwater impacts and potentially harmful algal blooms. In 2014, a vessel collision in Galveston Bay near Texas City released approximately 168,000 gallons of intermediate fuel oil. Through the National Resource Damage Assessment (NRDA) process, impacts of this spill are currently being evaluated and will include impacts to common bottlenose dolphins (NOAA DAARP 2018). In 2017, Hurricane Harvey dropped record amounts of rainfall on the Texas coast leading to significant freshwater runoff and a lowering of the salinity in Galveston Bay. Fazioli and Mintzer (2020) found that skin lesion prevalence increased significantly after the event, and remained high for more than four months after the hurricane. In addition, most dolphins moved out of their common habitat in the upper portion of Galveston Bay, and others shifted their distribution to deeper channels in the bay where salinity increased with depth. Harmful algal blooms and low dissolved oxygen are habitat issues leading to fish kills almost annually in the summers for Galveston and West Bays (McInnes and Quigg 2010; Rosel and Phillips 2014). For the 2011–2012 UME mentioned above (Strandings section), the strandings were coincident with a large harmful algal bloom of *K. brevis*. The definitive cause of that event has not been determined, but the algal bloom could have contributed to the mortality event. Fire *et al.* (2020) examined common bottlenose dolphins stranded along the Texas coast from 2007–2017 and found a high prevalence of brevetoxin exposure regardless of the association of stranded animals with a *K. brevis* bloom. Their results demonstrated evidence of long-term recurring exposure to *K. brevis* bloom toxins, but the health impacts of such exposure are unknown.

Finally, Galveston Bay experienced significant storm surges during Hurricane Ike in 2008. As a result, discussion and planning for an improved coastal barrier to protect the region from storm surge is in the works. Part of this proposed project includes construction of massive flood gates across the mouth of Galveston Bay and the Houston Ship Channel. Construction of these gates across Bolivar Pass encompasses an area heavily used by common bottlenose dolphins (Ronje *et al.* 2020). In addition, the structure is projected to diminish tidal flow from Galveston Bay by as much as 10% (U.S. Army Corps of Engineers 2020).

STATUS OF STOCK

Common bottlenose dolphins are not listed as threatened or endangered under the Endangered Species Act, and the Galveston Bay, East Bay, Trinity Bay Stock is not a strategic stock under the MMPA. Total U.S. fishery-related 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 a zero mortality and serious injury rate The status of this stock relative to optimum sustainable population is unknown and there are insufficient data to determine population trends for this stock. However, NMFS has concern for this stock because of documented freshwater impacts, forthcoming large-scale ecosystem projects (e.g., floodwalls), oil spills (e.g., Texas City Y), and a potential underestimation of fishery impacts.

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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, 1996b; 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 presence 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 genetic population differentiation among all areas. The Sellas *et al.* (2005) findings support the identification of BSE populations distinct from those occurring in adjacent Gulf coastal waters. Rosel *et al.* (2017) also identified significant population differentiation between estuarine residents of Barataria Bay and the adjacent coastal stock. Differences in reproductive seasonality from site to site also suggest genetic-based distinctions among areas (Urian *et al.* 1996). Photo-ID and genetic data from several inshore areas of the southeastern United States also support the existence of resident estuarine animals and differentiation between animals biopsied along the Atlantic coast and those biopsied within estuarine systems at the same latitude (Caldwell 2001; Gubbins 2002; Zolman 2002; Mazzoil *et al.* 2005; Litz 2007; Rosel *et al*. 2009).

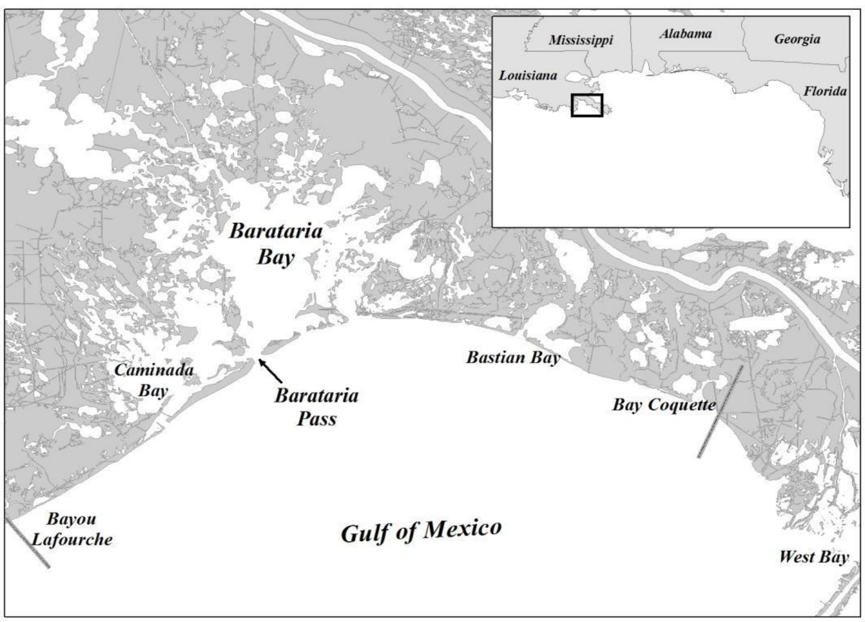

Figure 1. Geographic extent of the Barataria Bay Estuarine System Stock, located on the coast of Louisiana. The borders are denoted by solid lines.

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 designated 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 designated 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 2015–2019, no bottlenose dolphins were reported stranded to the southeast of BBES.

POPULATION SIZE

The best available abundance estimate for the BBES Stock of common bottlenose dolphins is 2,071 (CV=0.06; 95%CI: 1,832–2,309; Table 1), which is from vessel-based capture-recapture photo-ID surveys conducted during March and April 2019 (Garrison *et al.* 2020).

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.

McDonald *et al.* (2017) conducted vessel-based capture-mark-recapture (CMR) 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 CMR 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 CMR 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 CMR 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.

Recent Surveys and Abundance Estimates

Vessel-based CMR photo-ID surveys were conducted from 14 March to 1 April 2019 (Garrison *et al.* 2020). The surveyed area was expanded from that covered by DWH NRDA surveys (McDonald *et al.* 2017) to include the eastern and northern portions of the Bay. Data were analyzed with MARK version 9.0 software (White and Burnham 1999) using closed population CMR methods. Models were analyzed using the Full-Likelihood (Otis *et al.* 1978) and conditional (Huggins 1989) approaches, with similar results for both methods. The results of the Full-Likelihood approach are reported here. Abundance estimates were adjusted for the proportion of the population that had non-distinctive fins (see Garrison *et al.* 2020), and the resulting best estimate was 2,071 (CV=0.06; 95%CI: 1,832–2,309; Table 1).

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 2,071 (CV=0.06). The minimum population estimate for the BBES Stock is 1,971 bottlenose dolphins (Table 1).

Current Population Trend

There are insufficient data to assess population trends for this stock. The surveyed areas and methodology between the two available estimates are too different to allow a reliable evaluation of trends.

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 1,971. The maximum productivity rate is 0.04, the default value for cetaceans. The recovery factor is 0.45 because the CV of the shrimp trawl mortality estimate for Louisiana BSE stocks is greater than 0.6 (Wade and Angliss 1997). PBR for this stock of common bottlenose dolphins is 18 (Table 1).

Table 1. Best and minimum abundance estimates for the Barataria Bay Estuarine System Stock of common bottlenose dolphins with Maximum Productivity Rate (R_{max}), Recovery Factor (F_r) and PBR.

N_{est}	N_{est} CV	N_{min}	F_r	R_{max}	PBR
2,071	0.06	1,971	0.45	0.04	18

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 2015–2019 is unknown. Across Louisiana BSE stocks (from Sabine Lake east to Barataria Bay), the total annual estimated mortality for the shrimp trawl fishery was 45 (CV=0.65), but the portion of this attributed to the BBES Stock is unknown (see Shrimp Trawl section). The mean annual fishery-related mortality and serious injury during 2015–2019 for strandings and at-sea observations identified as fishery-related was 0. Additional mean annual mortality and serious injury during 2015–2019 due to other human-caused sources (fishery research, at-sea entanglements, gunshot wounds, and DWH oil spill) was 41. The minimum total mean annual human-caused mortality and serious injury for this stock during 2015–2019 was therefore 41 (Table 2). This is considered 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 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); and two Category III fisheries (Gulf of Mexico blue crab trap/pot; and Atlantic Ocean, Gulf of Mexico, Caribbean commercial passenger fishing vessel (hook and line)). Detailed fishery information is presented in Appendix III.

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

Shrimp Trawl

During 2015–2019, based on limited observer coverage in Louisiana BSE waters under the NMFS MARFIN program, there was one observed mortality and no observed serious injuries of common bottlenose dolphins from Gulf of Mexico BSE stocks by commercial shrimp trawls. Between 1997 and 2019, 13 common bottlenose dolphins and nine 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 observed trips of the commercial shrimp trawl fishery in the Gulf of Mexico (Soldevilla *et al.* 2021). 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; 2021) provided mortality estimates calculated from analysis of shrimp fishery effort data and NMFS's Observer Program bycatch data. Limited observer program coverage of Louisiana BSE waters started in 2015, but has not yet reached sufficient levels for estimating BSE bycatch rates; therefore time-area stratified bycatch rates were extrapolated into inshore waters to estimate a five-year unweighted mean mortality estimate for 2015–2019 based on inshore fishing effort (Soldevilla *et al.* 2021). Because the spatial resolution at which fishery effort is modeled is aggregated into four state areas (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 mean annual mortality estimate for Louisiana BSE stocks for the years 2015–2019 was 45 (CV=0.65; Soldevilla *et al.* 2021). 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 61% of shrimp fishery effort in western Louisiana 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; 2021).

In addition, chaffing gear from a commercial shrimp trawl was recovered in a dolphin carcass that stranded during 2015. It is likely the animal ingested the gear while removing gilled fish that were caught in the trawl net. This animal was ascribed to both the BBES and Western Coastal stocks, and it was included in the stranding database (NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 25 August 2020; Table 3).

Menhaden Purse Seine

During 2015–2019 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 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.

Blue Crab Trap/Pot

During 2015–2019 there were no documented interactions in commercial blue crab trap/pot gear for the BBES Stock. There is no observer coverage of crab trap/pot fisheries, so it is not possible to quantify total mortality.

Hook and Line (Rod and Reel)

During 2015–2019, two interactions with hook and line gear were documented within the stranding database (NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 25 August 2020; Table 3). In 2017, hook and line gear entanglement or ingestion were documented for one mortality and one animal released alive. For the live animal, it was initially seriously injured, but due to mitigation efforts, was released without serious injury (Maze-Foley and Garrison 2020). For the mortality, available evidence from the stranding data suggested the hook and line gear interaction did not contribute to the cause of death, and this animal was not included in the annual human-caused mortality and serious injury total for this stock (Table 2).

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 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 2015–2019, the model projected 204 mortalities (Table 2). 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 2015–2019, 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 25 August 2020) and in the totals presented in Table 3, as well as in the annual human-caused mortality and serious injury total for this stock (Table 2).

During 2015–2019, there was one at-sea observation during 2015 in Barataria Bay of a dolphin entangled around the head by a constricting strap. This animal was considered seriously injured (Maze-Foley and Garrison 2020) and was included in the annual human-caused mortality and serious injury total for this stock (Table 2).

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 2015–2019, for one mortality, gunshot pellets were found during the necropsy. The gunshot occurred pre-mortem but was not believed 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 25 August 2020) and in the totals presented in Table 3, but was not included within the annual human-caused mortality and serious injury total for this stock (Table 2). 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 2019 shooting involved depredation.

Depredation of fishing catch and/or bait 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 common bottlenose dolphins, may encourage depredation because provisioning conditions dolphins to approach humans and vessels, where they then may prey on bait and catches (Vail 2016). Such conditioning increases risks of subsequent injury and mortality (Christiansen *et al.* 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 2.

Table 2. 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, 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 strandings, at-sea counts, and fisherman self-reported takes, the number reported is a minimum because not all strandings, at-sea cases, or gear interactions are detected. See the Annual Human-Caused Mortality and Serious Injury section for biases and limitations of mortality estimates, and the Strandings section for limitations of stranding data. NA = not applicable. *Indicates the count would have been higher (1 instead of 0) had it not been for mitigation efforts (see text for that specific fishery for further details).

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	2015–2019	Observer Data	Undetermined for this stock but may be non-zero (see Shrimp Trawl section)	NA

Menhaden Purse Seine	2015–2019	Pilot Observer Program (2011); MMAP fisherman self-reported takes	NA	0
Atlantic Blue Crab Trap/Pot	2015–2019	Stranding Data	NA	0
Hook and Line	2015–2019	Stranding Data and At-Sea Observations	NA	0*
Mean Annual Mortality due to commercial fisheries (2015–2019)			**0**	
Research Takes (fishery research; 5-year Count)			1	
Other Takes (at-sea entanglements, gunshot wound; 5-year Count)			1	
Mortality due to DWH (5-year Projection)			204	
Mean Annual Mortality due to research takes, other takes, and DWH (2015–2019)			**41**	
Minimum Total Mean Annual Human-Caused Mortality and Serious Injury (2015–2019)			**41**	

Strandings

During 2015–2019, 138 common bottlenose dolphins were reported stranded within the BBES area (Table 3; NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 25 August 2020). There was evidence of human interaction (HI) for 10 of the strandings. No evidence of human interaction was detected for 14 strandings, and for the remaining 114 strandings, it could not be determined if there was evidence of human interaction. Human interactions were from numerous sources, including two entanglements with hook and line gear, one incidental take in a research gillnet, one mortality with evidence of gunshot wound, and one animal with evidence of a vessel strike (Table 3). It should be noted that evidence of human interaction does not necessarily mean the interaction caused the animal's stranding or death.

The assignment of animals to a single stock is impossible in some regions where stocks overlap, especially in nearshore coastal waters (Maze-Foley *et al.* 2019). Of the 138 strandings ascribed to the BBES Stock, 39 were ascribed solely to this stock. It is likely, therefore, that the counts in Table 3 include some animals from the Western Coastal Stock and the Terrebonne-Timbalier Bay Estuarine System (TTBES) Stock, and thereby overestimate the number of strandings for the BBES Stock; those strandings that could not be definitively ascribed to the BBES Stock were also included in the counts for the Western Coastal Stock or TTBES Stock 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).

There are a number of other difficulties associated with the interpretation of stranding data. Stranding data underestimate the extent of human and fishery-related mortality and serious injury because not all of the dolphins that die or are seriously injured in human interactions wash ashore, or, if they do, they are not all recovered (Peltier *et al*. 2012; Wells *et al*. 2015; Carretta *et al.* 2016). 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 three bottlenose dolphin die-offs or Unusual Mortality Events (UME). 1)

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). 2) 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. 3) During 1 February 2019 to 30 November 2019, a UME was declared for the area from the eastern border of Taylor County, Florida, west through Alabama, Mississippi, and Louisiana (http://www.nmfs.noaa.gov/pr/health/mmume/cetacean_gulfofmexico.htm, accessed 5 November 2020). A total of 337 common bottlenose dolphins stranded during this event, with 33 of them being from the BBES Stock. The largest number of mortalities occurred in eastern Louisiana and Mississippi. An investigation concluded the event was caused by exposure to low salinity waters as a result of extreme freshwater discharge from rivers. The unprecedented amount of freshwater discharge during 2019 (e.g., Gasparini and Yuill 2020) resulted in low salinity levels across the region.

Table 3. Common bottlenose dolphin strandings occurring in the Barataria Bay Estuarine System Stock area from 2015 to 2019, 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 25 August 2020). Please note HI does not necessarily mean the interaction caused the animal's death.

Stock	Category	2015	2016	2017	2018	2019	Total
Barataria Bay Estuarine System Stock	Total Stranded	33	29	36	5	35[c]	138
	Human Interaction						
	---Yes	3[a]	1	2[b]	1	3[d]	10
	---No	2	7	4	0	1	14
	---CBD	28	21	30	4	31	114

a. Includes 1 entanglement interaction in research gillnet gear (mortality), 1 interaction with chaffing gear from a commercial shrimp trawl (mortality), and 1 animal with healed vessel strike wounds (alive).
b. Includes 2 entanglement interactions with hook and line gear (1 mortality and 1 released alive without serious injury).
c. 33 strandings were part of the UME event in the northern Gulf of Mexico.
d. Includes 1 animal with evidence of gunshot wounds (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). 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, 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 (POPs) 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. A subsequent study by Balmer *et al.* (2018), using both blubber and blood samples collected during health assessments in 2011, 2013, and 2014, also examined POP concentrations. In comparison to Mississippi Sound and Sarasota Bay, dolphins from Barataria Bay had the lowest contaminant levels examined. 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 in Sarasota Bay, Florida (Wells *et al.* 2014). 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 prolonged low salinity exposure (e.g., Andersen 1973; Holyoake *et al.* 2010; Garrison *et al.* 2020).

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 the Barataria Bay Estuarine System Stock a strategic stock under the MMPA. The documented mean annual human-caused mortality for this stock for 2015–2019 was 41. 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 section), 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 section). 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, 1996b; 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

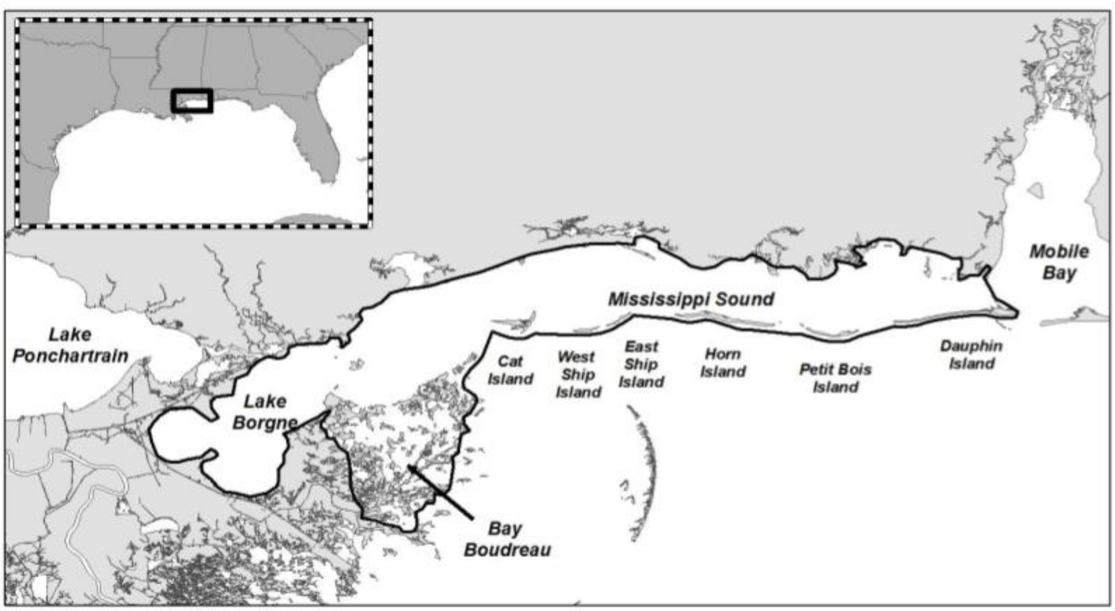

Figure 1. Geographic extent of the Mississippi Sound, Lake Borgne, Bay Boudreau Stock, located on the coasts of Alabama, Mississippi and Louisiana.

discrete, demographically independent bay, sound and estuary (BSE) populations (Duffield and Wells 2002; Sellas *et al.* 2005; Rosel *et al.* 2017). 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 genetic population structure among all areas. The Sellas *et al.* (2005) findings support the identification of BSE populations distinct from those occurring in adjacent Gulf coastal waters. Rosel *et al.* (2017) also identified significant population differentiation between estuarine residents of Barataria Bay and the adjacent coastal stock. 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).

The Mississippi Sound, Lake Borgne, Bay Boudreau Stock was designated 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, 1978b) 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 common bottlenose dolphins (Scott 1990). Mississippi Sound was once the site of the largest live-capture fishery of common bottlenose dolphins in North America (Reeves and Leatherwood 1984). Between 1973 and 1988, of the 533 common 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 common 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 common 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 common 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 common bottlenose dolphins were captured from Mississippi Sound, freeze-branded and released during 1982–1983 (Solangi and Dukes 1983; Lohoefener *et al*. 1990). Re-sighting efforts for these dolphins conducted from 1982–1985 by Lohoefener *et al.* (1990) 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 for 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.

In 2013, 19 dolphins (11 males and 8 females) were satellite tagged in Mississippi Sound with most (17) tagged near the mainland off eastern Mississippi and two tagged off the barrier islands (Mullin *et al.* 2017). Tag life averaged about 200 days. Dolphins tagged near the coast had a variety of ranges but generally remained in the region where they were tagged along the coast to mid-Mississippi Sound. One ranged into extreme eastern Mobile Bay and one other briefly into the Gulf of Mexico, but the others did not range outside of Mississippi Sound. Those tagged near the barrier islands ranged wider east to west but always in a very narrow corridor along both sides of the islands. While more work is needed, these tagging data indicate the potential for at least two dolphin communities, mainland and island, in the MS Sound Region.

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 and satellite tag 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 1,265 (CV=0.35; Table 1; Garrison *et al.* 2021). This estimate is from an aerial survey conducted during winter 2018.

Earlier Abundance Estimates (>8 years old)

Please see Appendix IV and Hayes *et al.* (2018) for a summary of abundance estimates, including earlier estimates and survey descriptions from NMFS surveys. In addition to NMFS surveys, 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 were 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=0.31) 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.

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 summer (June–August) 2017 and fall (October–November) 2018, and from Tampa, Florida, to Port O'Connor, Texas, during winter (January–March) 2018. The surveys were conducted along tracklines oriented perpendicular to the shoreline and spaced 20 km apart. The total survey effort varied during each survey due to weather conditions, but ranged between

8,046 and 14,590 km. The Mississippi Sound, Lake Borgne, Bay Boudreau Stock boundaries were surveyed completely in each season, and tracklines were spaced 5 km apart. Survey effort within the stock boundaries ranged between 487 and 750 km of effort (Garrison *et al.* 2021). 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). Abundance was calculated using mark-recapture distance sampling implemented in package mrds (version 2.21; Laake *et al.* 2020) in the R statistical programming language. This approach estimates both the probability of detection on the trackline and within the surveyed strip accounting for the effects of sighting conditions (e.g., sea state, glare, turbidity, and cloud cover). A different detection probability model was used for each seasonal survey (Garrison *et al.* 2021). The abundance estimates for the Mississippi Sound, Lake Borgne, Bay Boudreau Stock of bottlenose dolphins were based upon tracklines and sightings in waters along the Alabama, Mississippi, and Louisiana coasts inside of the barrier islands. The seasonal abundance estimates for this stock were: summer – 2,146 (CV=0.34), winter – 1,265 (CV=0.35), and fall – 4,337 (CV=0.16). 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_{est} 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 1,265 (CV=0.35; Table 1).

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 1,265 (CV=0.35). The minimum population estimate for the stock is 947 common bottlenose dolphins (Table 1).

Current Population Trend

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). Point estimates of common bottlenose dolphin abundance have been made based on aerial data from surveys during 2011−2012 and 2017−2018 (Garrison *et al.* 2021). Each of these surveys had a similar design and was conducted using the same aircraft and observer configuration. The resulting abundance estimates for winter seasonal surveys were: 2011–2012 – 1,104 (CV=0.59) and 2017−2018 – 1,265 (CV=0.35). A trends analysis is not possible because there are only two abundance estimates available. For further information on comparisons of old and current abundance estimates for this stock see Garrison *et al.* (2021).

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 *Deepwater Horizon* (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 947. The maximum productivity rate is 0.04, the default value for cetaceans. The recovery factor is 0.45 because the CV of the shrimp trawl mortality estimate for Mississippi and Alabama BSE stocks is greater than 0.6 (Wade and Angliss 1997). PBR for the Mississippi Sound, Lake Borgne, Bay Boudreau Stock of bottlenose dolphins is 8.5 (Table 1).

Table 1. Best and minimum abundance estimates for the Mississippi Sound, Lake Borgne, Bay Boudreau Stock of common bottlenose dolphins with Maximum Productivity Rate (R_{max}), Recovery Factor (F_r) and PBR.

N_{est}	N_{est} CV	N_{min}	F_r	R_{max}	PBR

1,265	0.35	947	0.45	0.04	8.5

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 2015–2019 is unknown. Across Mississippi/Alabama BSE stocks (from Mississippi River Delta east to Mobile Bay, Bonsecour Bay), the total annual estimated mortality for the shrimp trawl fishery was 33 (CV=0.70), but the portion of this attributed to the Mississippi Sound, Lake Borgne, Bay Boudreau Stock is unknown (see Shrimp Trawl section). The mean annual fishery-related mortality and serious injury during 2015–2019 for strandings and at-sea observations identified as fishery-related was 2.0. Additional mean annual mortality and serious injury during 2015–2019 due to other human-caused sources (fishery research, gunshot wounds, and DWH oil spill) was 57. The minimum total mean annual human-caused mortality and serious injury for this stock during 2015–2019 was therefore 59 (Table 2). This is considered 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 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 five commercial fisheries that interact, or potentially could interact, with this stock. These include three Category II fisheries (Southeastern U.S. Atlantic, Gulf of Mexico shrimp trawl; Gulf of Mexico gillnet; Gulf of Mexico menhaden purse seine) and two Category III fisheries (Gulf of Mexico blue crab trap/pot; Atlantic Ocean, Gulf of Mexico, Caribbean commercial passenger fishing vessel (hook and line)). Detailed fishery information is presented in Appendix III.

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

Shrimp Trawl

During 2015–2019, based on limited observer coverage in Louisiana BSE waters under the NMFS MARFIN program, there was one observed mortality and no observed serious injuries of common bottlenose dolphins from Gulf of Mexico BSE stocks by commercial shrimp trawls. Between 1997 and 2019, 13 common bottlenose dolphins and nine 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 observed trips of the commercial shrimp trawl fishery in the Gulf of Mexico (Soldevilla *et al.* 2021). 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, 2021) provided mortality estimates calculated from analysis of shrimp fishery effort data and NMFS's Observer Program bycatch data. Limited observer program coverage of Louisiana BSE waters started in 2015, but has not yet reached sufficient levels for estimating BSE bycatch rates; therefore time-area stratified bycatch rates were extrapolated into inshore waters to estimate the most recent five-year unweighted mean mortality estimate for 2015–2019 based on inshore fishing effort (Soldevila *et al*. 2021). Because the spatial resolution at which fishery effort is modeled is aggregated into four state areas (e.g., Nance *et al.* 2008), the mortality estimate covers all inshore waters of Mississippi, Alabama, and eastern Louisiana and thus all their respective BSE stocks, not just the Mississippi Sound, Lake Borgne, Bay Boudreau Stock. The mean annual mortality estimate for Mississippi/Alabama BSE stocks (from Mississippi River Delta east to Mobile Bay, Bonsecour Bay) was 33 (CV=0.70) dolphins per year. If all of the mortality occurred in the Mississippi Sound, Lake Borgne, Bay Boudreau Stock, the mortality estimate would exceed 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 38% of shrimp fishery effort in eastern Louisiana, Mississippi, and Alabama 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, 2021).

Gillnet

No marine mammal mortalities associated with gillnet fisheries have been reported or observed for the Mississippi Sound, Lake Borgne, Bay Boudreau Stock. There is no observer coverage of gillnet fisheries within the estuarine waters of the Mississippi Sound, Lake Borgne, or Bay Boudreau. There is limited observer coverage of gillnet fisheries in federal waters (e.g., Mathers *et al.* 2020), but none currently in state waters, although during 2012–2018 NMFS placed observers on commercial vessels (state permitted gillnet vessels) in the coastal state waters of Alabama, Mississippi, and Louisiana (Mathers *et al.* 2016). No takes were observed in state coastal waters during that time. However, stranding data suggest that gillnet and marine mammal interactions do occur (Read and Murray 2000), causing mortality and serious injury. During 2015–2019, two stranded common bottlenose dolphins were recovered with markings indicative of interaction with gillnet gear, but no gillnet gear was attached to the carcasses and it is unknown whether the interactions with the gear contributed to the death of these animals. One case was ascribed to the Mississippi Sound, Lake Borgne, Bay Boudreau Stock (2015; was entangled in hook and line gear and is also discussed in the Hook and Line section below), and one case was ascribed to both the Mississippi Sound, Lake Borgne, Bay Boudreau and the Northern Coastal stocks (2016). Because there is no observer program within this stock's boundaries, it is not possible to estimate the total number of mortalities or serious injuries associated with gillnet gear.

Menhaden Purse Seine

During 2015–2019, there were four mortalities documented through the Marine Mammal Authorization Program (MMAP) within waters of the MS Sound Region that involved the menhaden purse seine fishery (Table 2). Two incidents involving two dolphins each were reported as entangled within a single purse seine, both occurring during 2018. There is, however, currently no observer program for the Gulf of Mexico menhaden purse seine fishery. 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 2015–2019, there were three mortalities and one serious injury (Maze-Foley and Garrison 2021) of common bottlenose dolphins for which blue crab trap/pot gear entanglement were documented within the stranding data. Two of the cases were confirmed to involve commercial gear, and for the remaining two, it could not be determined whether the gear was commercial or recreational. Three cases were ascribed to the Mississippi Sound, Lake Borgne, Bay Boudreau Stock, and one case was ascribed to both the Mississippi Sound, Lake Borgne, Bay Boudreau and the Northern Coastal stocks. The mortalities occurred during 2016, 2018, and 2019, and the serious injury occurred in 2017. The mortalities and serious injury were all included in the stranding database (NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 25 August 2020) and in the totals presented in Table 3, as well as in the annual human-caused mortality and serious injury total for this stock (Table 2). There is no 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 2015–2019, there were four mortalities of common bottlenose dolphins for which hook and line gear entanglement or ingestion were documented within the stranding data. One mortality occurred in 2015 and three occurred in 2019. For two of these mortalities (2015, 2019), available evidence from the stranding records suggested the hook and line gear interactions contributed to the cause of death (the 2015 mortality also had markings indicative of interaction with gillnet gear and is also discussed in the Gillnet section above). For one mortality (2019), available evidence suggested the hook and line gear interaction was not a contributing factor to cause of death. For one mortality (2019), based on available evidence, it could not be determined if the hook and line gear interaction contributed to the cause of death. Three cases were ascribed to the Mississippi Sound, Lake Borgne, Bay Boudreau Stock, and one case was ascribed to the Mississippi Sound, Lake Borgne, Bay Boudreau and the Mobile Bay, Bonsecour Bay stocks. These mortalities were included in the stranding database (NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 25 August 2020) and in the totals presented in Table 3. The two mortalities

(2015, 2019) for which evidence suggested the gear contributed to the cause of death were included in the annual human-caused mortality and serious injury total for this stock (Table 2).

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 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 2015–2019, the model projected 282 mortalities (Table 2). The observed differences in abundance estimates from 2011–2012 and 2017–2018 are not consistent with this predicted change in population size. 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).

One mortality was documented in 2016 in the MS Sound Region as a result of an entanglement in a fishery research gillnet. This interaction was included in the stranding database (NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 25 August 2020) and in the totals presented in Table 3, and it was also included in the annual human-caused mortality and serious injury total for this stock (Table 2).

During 2019, one stranded common bottlenose dolphin was recovered with markings indicative of twisted twine 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 this animal. The case was ascribed to both the Mississippi Sound, Lake Borgne, Bay Boudreau and Northern Coastal stocks. This interaction was included in the stranding database (NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 25 August 2020) and in the totals presented in Table 3, but it was not included in the annual human-caused mortality and serious injury total for this stock (Table 2).

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 2015–2019, two mortalities were attributed to a shooting in 2018, and in 2019 gunshot pellets were found in a carcass during necropsy. For the 2018 case, a pregnant dolphin was found to have died from the gunshot wound, and her unborn calf died as a result of her death. For the 2019 case, the gunshot was not believed to be the cause of death (included in Table 3; NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 25 August 2020). The two gunshot mortalities from 2018 were included in the annual human-caused mortality and serious injury total for this stock (Table 2). 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 of fishing catch and/or bait 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). Such conditioning increases risks of subsequent injury and mortality (Christiansen *et al.* 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 2.

Table 2. 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, 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 strandings, at-sea counts, and fisherman self-reported takes, the number reported is a minimum because not all strandings, at-sea cases, or gear interactions are detected. See the Annual Human-Caused Mortality and Serious Injury section for biases and limitations of mortality estimates, and the Strandings section for limitations of stranding data. 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	2015–2019	Observer Data	Undetermined for this stock but may be non-zero (see Shrimp Trawl section)	NA
Menhaden Purse Seine	2015–2019	MMAP fisherman self-reported takes	NA	4
Atlantic Blue Crab Trap/Pot	2015–2019	Stranding Data	NA	4
Hook and Line	2015–2019	Stranding Data	NA	2
Mean Annual Mortality due to commercial fisheries (2015–2019)			**2.0**	
Research Takes (5-year Count)			1	
Other Takes (gunshot wounds; 5-year Count)			2	
Mortality due to DWH (5-year Projection)			282	
Mean Annual Mortality due to research takes, other takes, and DWH (2015–2019)			**57**	
Minimum Total Mean Annual Human-Caused Mortality and Serious Injury (2015–2019)			**59**	

Strandings

During 2015–2019, 405 common bottlenose dolphins were reported stranded within the Mississippi Sound, Lake Borgne, Bay Boudreau Stock area (Table 3; NOAA National Marine Mammal Health and Stranding Response Database unpublished data, 25 August 2020). Of those 405, three 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. There was evidence of human interaction (HI) for 25 of the strandings. No evidence of human interaction was detected for

13 strandings, and for the remaining 367 strandings, it could not be determined if there was evidence of human interaction. Human interactions were from numerous sources, including four entanglements with hook and line gear, four entanglements with crab trap/pot gear, one incidental take in a research gillnet, one mortality with markings indicative of interaction with twisted twine net gear, two mortalities with markings indicative of interactions with gillnet gear, two mortalities with evidence of gunshot wounds, and three animals with evidence of a vessel strike (Table 3). It should be noted that evidence of human interaction does not necessarily mean the interaction caused the animal's stranding or death.

The assignment of animals to a single stock is impossible in some regions where stocks overlap, especially in nearshore coastal waters (Maze-Foley *et al.* 2019). Of the 405 strandings ascribed to the Mississippi Sound, Lake Borgne, Bay Boudreau Stock, 356 were ascribed solely to this stock. It is likely, therefore, that the counts in Table 3 include some animals from the Western Coastal Stock and possibly the Mobile Bay, Bonsecour Bay Stock, and thereby overestimate the number of strandings for the stock; those strandings that could not be definitively ascribed to the Mississippi Sound, Lake Borgne, Bay Boudreau Stock were also included in the counts for the Western Coastal Stock or Mobile Bay, Bonsecour Bay Stock 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).

There are a number of other difficulties associated with the interpretation of stranding data. Stranding data underestimate the extent of human and fishery-related mortality and serious injury because not all of the dolphins that die or are seriously injured in human interactions wash ashore, or, if they do, they are not all recovered (Peltier *et al*. 2012; Wells *et al*. 2015; Carretta *et al.* 2016). 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 four common bottlenose dolphin die-offs or Unusual Mortality Events (UMEs). 1) From January through May 1990, a total of 344 common bottlenose dolphins stranded in the northern Gulf of Mexico 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). 2) In 1996 a UME was declared for common bottlenose dolphins in Mississippi when 31 common 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). 3) 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 section 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. 4) During 1 February 2019 to 30 November 2019, a UME was declared for the area from the eastern border of Taylor County, Florida, west through Alabama, Mississippi, and Louisiana (http://www.nmfs.noaa.gov/pr/health/mmume/cetacean_gulfofmexico.htm, accessed 5 November 2020). A total of 337 common bottlenose dolphins stranded during this event, with 166 of them being from the Mississippi Sound, Lake Borgne, Bay Boudreau Stock. The largest number of mortalities occurred in eastern Louisiana and Mississippi. An investigation concluded the event was caused by exposure to low salinity waters as a result of extreme freshwater discharge from rivers. The unprecedented amount of freshwater discharge during 2019 (e.g., Gasparini and Yuill 2020) resulted in low salinity levels across the region.

Table 3. Common bottlenose dolphin strandings occurring in the Mississippi Sound, Lake Borgne, Bay Boudreau Stock area from 2015 to 2019, 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 25 August 2020). Please note HI does not necessarily mean the interaction caused the animal's death.

Stock	Category	2015	2016	2017	2018	2019	Total

Mississippi Sound, Lake Borgne, Bay Boudreau Stock	Total Stranded	39	88	55	50	173[e]	405
	Human Interaction						
	---Yes	4[a]	5[b]	3[c]	5[d]	8[f]	25
	---No	0	1	2	3	7	13
	---CBD	35	82	50	42	158	367

a. Includes 2 mortalities with evidence of a vessel strike and 2 fisheries interactions (FI), 1 of which was an entanglement interaction (mortality) with hook and line fishing gear.

b. Includes 1 entanglement interaction in research gillnet gear (mortality), and 4 FIs, including 1 with markings indicative of interaction with gillnet gear and 1 entanglement interaction with trap/pot gear (mortality).

c. Includes 1 entanglement interaction with trap/pot gear (released alive seriously injured).

d. Includes 1 mortality with a gunshot wound, 1 mortality with evidence of a vessel strike, and 2 fisheries interactions (FI), 1 of which was an entanglement interaction (mortality) with commercial blue crab trap/pot gear.

e. 166 strandings were part of the UME event in the northern Gulf of Mexico.

f. Includes 1 mortality with evidence of a gunshot wound and 5 FIs, including 3 entanglement interactions (mortalities) with hook and line fishing gear, 1 entanglement interaction (mortality) with commercial blue crab trap/pot gear, and 1 animal with markings indicative of interaction with twisted twine net gear.

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). 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).

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 common 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 common 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 common 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; PCBs, chlordanes, mirex, DDTs, HCB and dieldrin) 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. A subsequent study by Balmer *et al.* (2018), using both blubber and blood samples collected during health assessments in 2011, 2013, and 2014, examined POP concentrations, also suggested that POPs were unlikely the cause of the adverse health and high stranding rates in Mississippi Sound.

Other Habitat Issues

Prior to the DWH oil spill, environmental contaminants have been an issue of concern for bottlenose dolphins throughout the southeastern U.S., including Mississippi Sound. Kucklick *et al.* (2011) examined POPs and polybrominated diphenyl ether (PBDE) concentrations from common bottlenose dolphin blubber and found that 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 other locations. However, as noted, Balmer *et al.* (2015) found lower levels of POPs in Mississippi Sound when compared to the results of Kucklick *et al.* (2011). Balmer *et al.* 2018 found that dolphins from Mississippi Sound had higher overall contaminant levels, based on blood samples, than those in Barataria Bay and Sarasota Bay, levels nearly 1.5 times higher than those detected in dolphins from the Sarasota Bay reference site. The authors suggested higher levels of several contaminants in Mississippi Sound dolphins are due to the established ship-building industry operating in the area.

The presence of vessels may impact common bottlenose dolphin behavior in bays, sounds, and estuaries. Miller *et al.* (2008) investigated the immediate responses of common 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 minimum estimate of human-caused mortality and serious injury exceeds PBR, the Mississippi Sound, Lake Borgne, Bay Boudreau Stock is a strategic stock under the MMPA. The documented mean annual human-caused mortality for this stock for 2015–2019 was 59. 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 section), 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 section). It is likely that this stock is below its optimum sustainable population (NMFS 2016) due to mortalities related to the DWH oil spill and two recent UMEs. Total fishery-related mortality and serious injury for this stock is unknown, but at a minimum is greater than 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*): Northern Gulf of Mexico Bay, Sound, and Estuary Stocks

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 northern Gulf of Mexico. To date, eight stocks have individual reports completed or drafted (West Bay, Galveston Bay/East Bay/Trinity Bay, Terrebonne-Timbalier Bay Estuarine System, Barataria Bay Estuarine System, Mississippi Sound/Lake Borgne/Bay Boudreau, Choctawhatchee Bay, St. Andrew Bay, and St. Joseph Bay), and the remaining 23 stocks are assessed in this report.

STOCK DEFINITION AND GEOGRAPHIC RANGE

Common bottlenose dolphins are distributed throughout the bays, sounds and estuaries of the Gulf of Mexico (Mullin 1988). The identification of demographically independent populations of common bottlenose dolphins in these waters is complicated by the high degree of behavioral variability exhibited by this species (Shane *et al.* 1986; Wells and Scott 1999; Wells 2003), and by the lack of requisite information for much of the region.

Distinct stocks are designated in each of 31 areas of contiguous, enclosed or semi-enclosed bodies of water adjacent to the northern Gulf of Mexico (i.e., U.S. Gulf of Mexico; Table 1; Figure 1). The genesis of the delineation of these stocks was work initiated in the 1970s in Sarasota Bay, Florida (Irvine and Wells 1972; Irvine *et al.* 1981), and in bays in Texas (Shane 1977; Gruber 1981). These studies documented year-round residency of individual common bottlenose dolphins in estuarine waters. As a result, the expectation of year-round resident populations was extended to bay, sound and estuary (BSE) waters across the northern Gulf of Mexico when the first stock assessment reports were published in 1995 (Blaylock *et al.* 1995). Since these early studies, long-term (year-round, multi-year) residency has been reported from nearly every site where photographic identification (photo-ID) or tagging studies have been conducted in the Gulf of Mexico. In Texas, long-term resident dolphins have been reported in the Matagorda-Espiritu Santo Bay area (Gruber 1981; Lynn and Würsig 2002), Aransas Pass (Shane 1977; Weller 1998), San Luis Pass (Maze and Würsig 1999; Irwin and Würsig 2004), and Galveston Bay (Bräger 1993; Bräger *et al.* 1994; Fertl 1994; Fazioli and Mintzer 2020). In Louisiana, Miller (2003) concluded the common bottlenose dolphin population in the Barataria Basin was relatively closed, and Wells *et al.* (2017) documented long-term, year-round residency in Barataria Bay based on telemetry data. Hubard *et al.* (2004) reported sightings of dolphins in Mississippi Sound that were known from tagging efforts there 12–15 years prior; long-term residency was further documented by Mullin *et al*. (2017). In Florida, long-term residency has been reported from Tampa Bay (Wells 1986; Wells *et al.* 1996b; Urian *et al.* 2009), Sarasota Bay (Irvine and Wells 1972; Irvine *et al.* 1981; Wells 1986; 1991; 2003; 2014; Wells *et al.* 1987; Scott *et al.* 1990), Lemon Bay (Wells *et al.* 1996a; Bassos-Hull *et al.* 2013), Charlotte Harbor/Pine Island Sound (Shane 1990; Wells *et al.* 1996a, 1997; Shane 2004; Bassos-Hull *et al.* 2013), and Gasparilla Sound (Bassos-Hull *et al.* 2013). In Sarasota Bay, which has the longest research history, up to five concurrent generations of identifiable residents have been identified, including individuals identified through more than four decades (Wells 2014). Maximum immigration and emigration rates of about 2–3% have been estimated (Wells and Scott 1990).

Genetic data also support the concept of relatively discrete BSE stocks. Analyses of mitochondrial DNA haplotype distributions indicate the existence of clinal variations along the Gulf of Mexico coastline (Duffield and Wells 2002). Differences in reproductive seasonality from site to site also suggest genetic-based distinctions between communities (Urian *et al.* 1996). Mitochondrial DNA analyses suggest finer-scale structural levels as well. For example, dolphins in Matagorda Bay, Texas, appear to be a localized population, and differences in haplotype frequencies distinguish among adjacent communities in Tampa Bay, Sarasota Bay, and Charlotte Harbor/Pine Island Sound, along the central west coast of Florida (Duffield and Wells 1991; 2002). Additionally, Sellas *et al.* (2005) examined population subdivision among dolphins sampled in Sarasota Bay, Tampa Bay, Charlotte Harbor, Matagorda Bay, and the coastal Gulf of Mexico (1–12 km offshore) from just outside Tampa Bay to the southern 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. Rosel *et al.* (2017) also identified significant population differentiation between estuarine residents of Barataria Bay and the adjacent coastal stock. The Sellas *et* *al.* (2005) and Rosel *et al.* (2017) findings support the separate identification of BSE populations from those occurring in adjacent Gulf coastal waters.

In many cases, residents occur primarily in BSE waters, with limited movements through passes to the Gulf of Mexico (Shane 1977, 1990; Gruber 1981; Irvine *et al.* 1981; Maze and Würsig 1999; Lynn and Würsig 2002; Fazioli *et al.* 2006). These habitat use patterns are reflected in the ecology of the dolphins in some areas; for example, residents of Sarasota Bay, Florida, lacked squid in their diet, unlike non-resident dolphins stranded on nearby Gulf beaches (Barros and Wells 1998). However, in some areas year-round residents may co-occur with non-resident dolphins. For example, about 14–17% of group sightings involving resident Sarasota Bay dolphins include at least one non-resident as well (Wells *et al.* 1987; Fazioli *et al.* 2006). Mixing of inshore residents and non-residents has been seen at San Luis Pass, Texas (Maze and Würsig 1999), Cedar Keys, Florida (Quintana-Rizzo and Wells 2001), and Pine Island Sound, Florida (Shane 2004). Non-residents exhibit a variety of movement patterns, ranging from apparent nomadism recorded as transience to a given area, to apparent seasonal or non-seasonal migrations. Passes, especially the mouths of the larger estuaries, serve as mixing areas. For example, dolphins from several different areas were documented at the mouth of Tampa Bay, Florida (Wells 1986), and most of the dolphins identified in the mouths of Galveston Bay and Aransas Pass, Texas, were considered transients (Henningsen 1991; Bräger 1993; Weller 1998).

Seasonal movements of dolphins into and out of some of the bays, sounds and estuaries have also been documented. In Sarasota Bay, Florida, and San Luis Pass, Texas, residents have been documented using Gulf coastal waters more frequently in fall/winter, and inshore waters more in spring/summer (Irvine *et al.* 1981; Maze and Würsig 1999). Fall/winter increases in abundance have been noted for Tampa Bay (Scott *et al.* 1989) and are thought to occur in Matagorda Bay (Gruber 1981; Lynn and Würsig 2002) and Aransas Pass (Shane 1977; Weller 1998). Spring/summer increases in abundance occur in Mississippi Sound (Hubard *et al.* 2004) and are thought to occur in Galveston Bay (Henningsen 1991; Bräger 1993; Fertl 1994). However, Mullin *et al.* (2017) found that seasonal fluctuations in Mississippi Sound were less than previously reported.

Spring and fall increases in abundance have been reported for St. Joseph Bay, Florida. Mark-recapture abundance estimates were highest in spring and fall and lowest in summer and winter (Table 1; Balmer *et al.* 2008). Individuals with low site-fidelity indices were sighted more often in spring and fall, whereas individuals sighted during summer and winter displayed higher site-fidelity indices. In conjunction with health assessments, 23 dolphins were radio tagged during April 2005 and July 2006. Dolphins tagged in spring 2005 displayed variable utilization areas and variable site fidelity patterns. In contrast, during summer 2006 the majority of radio-tagged individuals displayed similar utilization areas and moderate to high site-fidelity patterns. The results of the studies suggest that during summer and winter St. Joseph Bay hosts dolphins that spend most of their time within this region, and these may represent a resident community. In spring and fall, St. Joseph Bay is visited by dolphins that range outside of this area (Balmer *et al.* 2008).

The current BSE stocks are designated as described in Table 1. There are some estuarine areas that are not currently part of any stock's range. Many of these are areas that dolphins cannot readily access. For example, the marshlands between Galveston Bay and Sabine Lake and between Sabine Lake and Calcasieu Lake are fronted by long, sandy beaches that prohibit dolphins from entering the marshes. The region between the Calcasieu Lake and Vermilion Bay/Atchafalaya Bay stocks has some access, but these marshes are predominantly freshwater rather than saltwater marshes, making them unsuitable for long-term survival of a viable population of common bottlenose dolphins. In other regions, there is insufficient estuarine habitat to harbor a demographically independent population, for instance between the Matagorda Bay and West Bay Stocks in Texas, and/or sufficient isolation of the estuarine habitat from coastal waters. The regions between the south end of the Estero Bay Stock area to just south of Naples and between Little Sarasota Bay and Lemon Bay are highly developed and contain little appropriate habitat. South of Naples to Marco Island and Gullivan Bay is also not currently covered within a stock boundary. This region contains common bottlenose dolphins, but the relationship of any dolphins in this region to other BSE stocks is unknown. They may be members of the Gullivan Bay to Chokoloskee Bay stock as there is passage behind Marco Island that would allow dolphins to move north. The regions between Apalachee Bay and Cedar Key/Waccasassa Bay, between Crystal Bay and St. Joseph Sound, and between Chokoloskee Bay and Whitewater Bay comprise thin strips of marshland with no barriers to adjacent coastal waters. Further work is necessary to determine whether year-round resident dolphins use these thin marshes or whether dolphins in these areas are members of the coastal stock that use the fringing marshland as well. Finally, the region between the eastern border of the Barataria Bay Estuarine System Stock and the Mississippi River Delta Stock to the east may harbor dolphins, but the area is small and work is necessary to determine whether any dolphins utilizing this habitat come from an adjacent BSE stock.

As more information becomes available, combination or division of these stocks, or alterations to stock boundaries, may be warranted. For example, research based on photo-ID data collected by Bassos-Hull *et al.* (2013) recommended combining Lemon Bay with Gasparilla Sound/Charlotte Harbor/Pine Island Sound. Therefore, these stocks have been combined (see Table 1). However, it should be noted this change was made in the absence of genetic data and could be revised again in the future when genetic data are available. Additionally, a number of geographically and socially distinct subgroupings of dolphins in regions such as Tampa Bay, Charlotte Harbor, Pine Island Sound, Barataria Bay, Aransas Pass, and Matagorda Bay have been identified (Shane 1977; Gruber 1981; Wells *et al.* 1996a, 1996b, 1997, 2017; Lynn and Würsig 2002; Urian 2002; Rosel *et al.* 2017). For Tampa Bay, Urian *et al.* (2009) described five discrete communities (including the adjacent Sarasota Bay community) that differed in their social interactions and ranging patterns. Structure was found despite a lack of physiographic barriers to movement within this large, open embayment. Urian *et al.* (2009) further suggested that fine-scale structure may be a common element among common bottlenose dolphins in the southeastern U.S. and recommended that management should account for fine-scale structure that exists within current stock designations. These results indicate that it is plausible some of these estuarine stocks, particularly those in larger bays and estuaries, comprise multiple demographically-independent populations.

Table 1. Most recent common bottlenose dolphin abundance estimate (N_{est}), coefficient of variation of N_{est} (CV N_{est}), minimum population estimate (N_{min}), Potential Biological Removal (PBR), year of the most recent abundance estimate and associated publication (Year), and minimum counts of annual human-caused mortality and serious injury (HCMSI) in northern Gulf of Mexico bay, sound and estuary stocks. When estimates are based on data collected more than eight years ago, they are considered unknown or undetermined for management purposes. Blocks refer to aerial survey blocks illustrated in Figure 1. UNK – unknown; UND – undetermined. For each stock denoted with a † symbol, please refer to the stand-alone report for this stock.

Blocks	Gulf of Mexico Estuary	N_{est}	CV N_{est}	N_{min}	PBR	Year (Reference)	Minimum Annual HCMSI, 2015–2019
B51	Laguna Madre	80	1.57	UNK	UND	1992 (A)	0.8
B52	Nueces Bay/Corpus Christi Bay	58	0.61	UNK	UND	1992 (A)	0.2
B50	Copano Bay/Aransas Bay/ San Antonio Bay/ Redfish Bay/Espiritu Santo Bay	55	0.82	UNK	UND	1992 (A)	0.6
B54	Matagorda Bay/ Tres Palacios Bay/Lavaca Bay	61	0.45	UNK	UND	1992 (A)	0.4
B55	West Bay†						
B56	Galveston Bay/East Bay/ Trinity Bay†						
B57	Sabine Lake	122[a]	0.19	104	0.9	2017 (B)	0
B58	Calcasieu Lake	0[b]	-	-	UND	1992 (A)	0.2
B59	Vermilion Bay/West Cote Blanche Bay/Atchafalaya Bay	0[b]	-	-	UND	1992 (A)	0
B60	Terrebonne-Timbalier Bay Estuarine System†						
B61	Barataria Bay Estuarine System†						

B30	Mississippi River Delta	1,446[c]	0.19	1,238	11	2017–2018 (C)	9.2
B02–05, 29, 31	Mississippi Sound/ Lake Borgne/Bay Boudreau†						
B06	Mobile Bay/Bonsecour Bay	122	0.34	UNK	UND	1993 (A)	16.0
B07	Perdido Bay	0[b]	-		UND	1993 (A)	0.8
B08	Pensacola Bay/East Bay	33	0.80	UNK	UND	1993 (A)	0.4
B09	Choctawhatchee Bay†						
B10	St. Andrew Bay†						
B11	St. Joseph Bay†						
B12–13	St. Vincent Sound/Apalachicola Bay/St. George Sound	439	0.14	UNK	UND	2007 (D)	0.2
B14–15	Apalachee Bay	491	0.39	UNK	UND	1993 (A)	0
B16	Waccasassa Bay/Withlacoochee Bay/Crystal Bay	UNK	-	UNK	UND	-	0.4
B17	St. Joseph Sound/ Clearwater Harbor	UNK	-	UNK	UND	-	0.8[d]
B32–34	Tampa Bay	UNK	-	UNK	UND	-	3.0
B20, 35	Sarasota Bay/Little Sarasota Bay	158	0.27	126	1.0	2015 (E)	0.2[e]
B21–23	Pine Island Sound/ Charlotte Harbor/ Gasparilla Sound/Lemon Bay	826	0.09	UNK	UND	2006 (F)	1.0[f]
B36	Caloosahatchee River	0[b]	-	-	UND	1985 (G)	0.4[g]
B24	Estero Bay	UNK	-	UNK	UND	-	0.4
B25	Chokoloskee Bay/Ten Thousand Islands/Gullivan Bay	UNK	-	UNK	UND	-	0.2
B27	Whitewater Bay	UNK	-	UNK	UND	-	0
B28	Florida Keys (southwest Marathon Key to Marquesas Keys)	UNK	-	UNK	UND	-	0.2

References: A – Blaylock and Hoggard 1994; B – Ronje *et al.* 2020; C – Garrison *et al.* 2021; D – Tyson *et al.* 2011; E – Tyson and Wells 2016; F – Bassos-Hull *et al.* 2013; G – Scott *et al.* 1989

Notes:

a. Winter seasonal estimate, Selective dataset.

b. During earlier surveys (Scott *et al.* 1989), the range of seasonal abundances was as follows: Calcasieu Lake, 0–6 (0.34); Vermilion Bay/West Cote Blanche Bay/Atchafalaya Bay, 0–0; Perdido Bay, 0–0; Lemon Bay, 0–15 (0.43); and Caloosahatchee River, 0–0.

c. Abundance estimate utilizes density estimate from adjacent waters. See Garrison *et al.* (2021) for details.

d. The minimum count would have been higher (1.2 instead of 0.8) had it not been for mitigation efforts.

e. The minimum count would have been higher (0.4 instead of 0.2) had it not been for mitigation efforts.

f. The minimum count would have been higher (1.4 instead of 1.0) had it not been for mitigation efforts.

g. The minimum count would have been higher (0.6 instead of 0.4) had it not been for mitigation efforts.

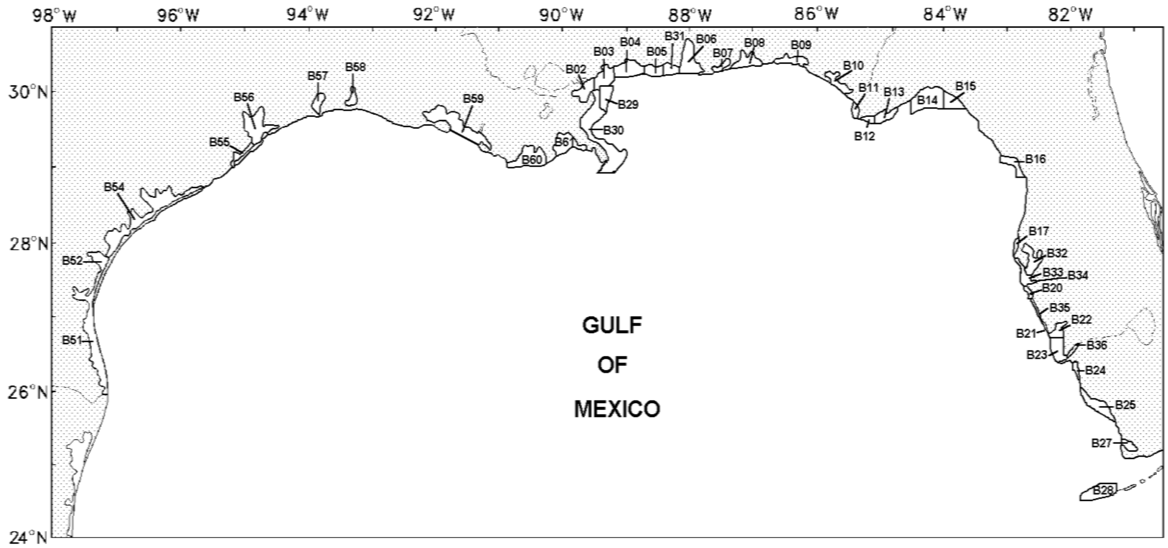

Figure 1. Northern Gulf of Mexico bays, sounds, and estuaries. Each of the alpha-numerically designated blocks corresponds to one of the NMFS Southeast Fisheries Science Center logistical aerial survey areas listed in Table 1. The common bottlenose dolphins inhabiting each bay, sound, or estuary are considered to comprise a unique stock for purposes of this assessment. Eight stocks have their own stock assessment report (see Table 1).

POPULATION SIZE

Population size estimates for most of these stocks are more than eight years old and therefore the current population sizes for all but three are considered unknown (Wade and Angliss 1997). However, a capture-mark-recapture population size estimate is available for Sarasota Bay/Little Sarasota Bay for 2015 (Tyson and Wells 2016) and Sabine Lake for 2017 (Ronje *et al.* 2020). Recent aerial survey line-transect population size estimates are available for Mississippi River Delta for 2017–2018 (Garrison *et al.* 2021; Table 1). Population size estimates for many stocks were generated from preliminary analyses of line-transect data collected during aerial surveys conducted in September–October 1992 in Texas and Louisiana and in September–October 1993 in Louisiana, Mississippi, Alabama, and the Florida Panhandle (Blaylock and Hoggard 1994; Table 1). Standard line-transect perpendicular sighting distance analytical methods (Buckland *et al.* 1993) and the computer program DISTANCE (Laake *et al.* 1993) were used.

Minimum Population Estimate

The population sizes for all but three stocks are currently unknown and the minimum population estimates are given for those three stocks in Table 1. The minimum population estimate is the lower limit of the two-tailed 60% confidence interval of the log-normally distributed best abundance estimate. This is equivalent to the 20th percentile of the log-normal distribution as specified by Wade and Angliss (1997). The minimum population estimate was calculated for each block from the estimated population size and its associated coefficient of variation.

Current Population Trend

The data are insufficient to determine population trends for most of the Gulf of Mexico BSE common bottlenose dolphin stocks.

CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

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

POTENTIAL BIOLOGICAL REMOVAL

Potential Biological Removal (PBR) is the product of minimum population size, one-half the maximum productivity rate and a recovery factor (Wade and Angliss 1997). The recovery factor is 0.45 for Louisiana, Mississippi, and Alabama BSE stocks because the CV of the shrimp trawl mortality estimate for those stocks is greater than 0.6. The recovery factor is 0.4 for Texas and Florida BSE stocks because the CV of the shrimp trawl mortality estimate for those stocks is greater than 0.8 (Wade and Angliss 1997). PBR is undetermined for all but three stocks because the population size estimates are more than eight years old. PBR for those stocks with population size estimates less than eight years old is given in Table 1.

ANNUAL HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

The total annual human-caused mortality and serious injury for these stocks of common bottlenose dolphins during 2015–2019 is unknown. Minimum estimates of human-caused mortality and serious injury for each stock are given in Table 1. These estimates are considered a minimum because: 1) not all fisheries that could interact with these stocks 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 because estimates are not available for individual BSE stocks (see Shrimp Trawl section), and 6) various assumptions were made in the population model used to estimate population decline for northern Gulf of Mexico BSE stocks impacted by the *Deepwater Horizon* (DWH) oil spill.

Fishery Information

There are seven commercial fisheries that interact, or that potentially could interact, with these stocks in the Gulf of Mexico. These include four Category II fisheries (Southeastern U.S. Atlantic, Gulf of Mexico shrimp trawl; Gulf of Mexico menhaden purse seine; Southeastern U.S. Atlantic, Gulf of Mexico stone crab trap/pot; and Gulf of Mexico gillnet fisheries); and three Category III fisheries (Gulf of Mexico blue crab trap/pot; Florida spiny lobster 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.

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

Shrimp Trawl

During 2015–2019, based on limited observer coverage in Louisiana BSE waters under the NMFS MARFIN program, there was one observed mortality and no observed serious injuries of common bottlenose dolphins from Gulf of Mexico BSE stocks by commercial shrimp trawls. Between 1997 and 2019, 13 common bottlenose dolphins and nine 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.* 2015, 2016, 2021). 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, 2021) provided mortality estimates calculated from analysis of shrimp fishery effort data and NMFS's Observer Program bycatch data. Limited observer coverage in Louisiana BSE waters started in 2015, but has not yet reached sufficient levels for estimating BSE bycatch rates; therefore time-area stratified bycatch rates were extrapolated into inshore waters to estimate the most recent five-year unweighted mean mortality estimate for 2015–2019 based on inshore fishing effort (Soldevilla *et al.* 2021). The 4-area (Texas, Louisiana, Mississippi/Alabama, Florida) stratification method was chosen because it best approximates how fisheries operate (Soldevilla *et al*. 2015, 2016, 2021). The BSE stock mortality estimates were aggregated at the state area level as this was the spatial resolution at which fishery effort is modeled (e.g., Nance *et al.* 2008). The mean annual mortality estimates for the BSE stocks were as follows: Texas BSE (from Galveston Bay/East Bay/Trinity Bay south to Laguna Madre): 0.4 (CV=1.62); Louisiana BSE (from Sabine Lake east to Barataria Bay): 45 (CV=0.65);

Mississippi/Alabama BSE (from Mississippi River Delta east to Mobile Bay/Bonsecour Bay): 33 (CV=0.70); and Florida BSE (from Perdido Bay east and south to the Florida Keys): 0.7 (CV=1.58). These estimates do not include skimmer trawl effort, which accounts for 61% of shrimp fishery effort in western Louisiana, and 38% of shrimp fishery effort in eastern Louisiana, Mississippi, and Alabama 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, 2021). It should be noted that because bycatch for individual BSE stocks cannot be quantified at this time, shrimp trawl bycatch is not being included in the annual human-caused mortality and serious injury total for any BSE stock.

During 2015–2019, stranding data documented two mortalities of common bottlenose dolphins associated with entanglement in shrimp trawl gear. Both mortalities occurred in 2016—one in Pensacola Bay and one in Perdido Bay.

During 2016 the Marine Mammal Authorization Program (MMAP) documented a self-reported incidental take (mortality) of a common bottlenose dolphin by a commercial fisherman trawling in Mobile Bay. The dolphin was entangled in the lazy line of the gear.

Menhaden Purse Seine

During 2015–2019 there was one mortality documented within waters of the Mississippi River Delta Stock that involved the menhaden purse seine fishery (Table 2). Through the Marine Mammal Authorization Program (MMAP), one animal was reported as entangled within a purse seine during 2018. There is currently no observer program for the Gulf of Mexico menhaden purse seine fishery.

Without an ongoing observer program, it is not possible to obtain statistically reliable incidental mortality and serious injury rates for this fishery, and the stocks from which common bottlenose dolphins are being taken. The documented mortality in this commercial fishery represents a minimum known count of mortalities and serious injuries in the last five years.

Blue Crab, Stone Crab and Florida Spiny Lobster Trap/Pot

During 2015–2019, there were nine documented interactions between trap/pot fisheries and BSE stocks. During 2019, two serious injuries occurred, one due to entanglement in commercial spiny lobster trap/pot gear, ascribed to the Florida Keys Stock, and the second due to entanglement in unidentified trap/pot gear, ascribed to the Waccassasa Bay Stock. Also during 2019, an animal was disentangled from commercial blue crab trap/pot gear and released alive. It could not be determined if the animal was seriously injured following mitigation efforts (the initial determination was seriously injured). This animal was ascribed to the Caloosahatchee River Stock. During 2017, one mortality and one serious injury occurred, both due to entanglement in commercial blue crab trap/pot gear. The mortality was ascribed to the Caloosahatchee River Stock, and the serious injury to the Waccasassa Bay Stock. During 2016, one animal was partially disentangled from unidentified trap/pot gear and released alive seriously injured. This animal was ascribed to the Pine Island Sound/Charlotte Harbor/Gasparilla Sound/Lemon Bay Stock. Also in 2016, an animal was disentangled from commercial stone crab trap/pot gear and released alive not seriously injured following disentanglement efforts (the initial determination was seriously injured). This animal was ascribed to the Sarasota Bay/Little Sarasota Bay Stock. During 2015, one mortality occurred due to entanglement in commercial blue crab trap/pot gear. This animal was ascribed to the Mobile Bay/Bonsecour Bay Stock. Also in 2015, one animal was disentangled and released alive from unidentified crab trap/pot gear but it could not be determined if the animal was seriously injured following mitigation efforts (the initial determination was seriously injured). This freeze-branded animal was known to belong to the Sarasota Bay/Little Sarasota Bay Stock. The specific fishery could not be identified for the trap/pot gear involved in several of the live releases. The mortality and the animals released alive were all included in the stranding database (NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 25 August 2020) and are included in the stranding totals in Table 4. The details for serious determinations for the live animals are provided in Maze-Foley and Garrison (2021).

Because there is no observer program for these fisheries, it is not possible to estimate the total number of interactions or mortalities associated with trap/pot gear. The documented interactions in this gear represent a minimum known count of interactions in the last five years.

Gillnet

During 2015–2019, there was one documented interaction with gillnet gear and a BSE stock. During 2019, a stranded carcass was recovered with gillnet gear wrapped around its rostrum, and it was ascribed to the St. Vincent

Sound, Apalachicola Bay, St. George Sound Stock. There has been limited observer coverage of this fishery in state waters. During 2012–2018, NMFS placed observers on commercial vessels (state permitted gillnet vessels) in the coastal waters of Alabama, Mississippi, and Louisiana (Mathers *et al.* 2016). No takes were observed in state waters during this time. However, stranding data indicate that gillnet interactions do occur, causing mortality and serious injury. During 2015–2019, three stranded common bottlenose dolphins were recovered with markings indicative of interaction with gillnet gear (Read and Murray 2000), but no gear was attached to the carcasses and it is unknown whether the interactions with the gear contributed to the death of these animals. Two of the strandings were ascribed to the Mobile Bay Stock and one to the Perdido Bay Stock. Because there is no observer program within BSE waters, it is not possible to estimate total mortalities and serious injuries incidental to gillnet fisheries.

In 1995, a Florida state constitutional amendment banned gillnets and large nets from bays, sounds, estuaries, and other inshore waters. Commercial and recreational gillnet fishing is also prohibited in Texas state waters.

For details on research-related entanglements in gillnet gear, see the Other Mortality section and Table 3 below.

Hook and Line (Rod and Reel)

During 2015–2019 there were 20 documented interactions (entanglements or ingestions) between hook and line gear and BSE stocks—14 mortalities and six live animals (disentanglement efforts were made for four of the six). The stranding data indicate that for six of the 14 mortalities, the hook and line gear interaction contributed to the cause of death. For five mortalities, evidence suggested the hook and line gear interaction was incidental and was not a contributing factor to cause of death. For three mortalities, it could not be determined if the hook and line gear interaction contributed to cause of death. Two live animals were considered seriously injured and no disentanglement efforts were made. Attempts were made to disentangle the remaining four live animals from hook and line gear. All four were considered seriously injured by the gear prior to mitigation efforts, but based on observations during mitigations, three animals were considered not seriously injured post-mitigation. For the remaining live animal, following mitigation it could not be determined if the animal was seriously injured. In summary, the evidence available from stranding data suggested that at least six mortalities and two serious injuries to animals from BSE stocks resulted from interactions with rod and reel hook and line gear. This number would have been higher without mitigation efforts to disentangle four live animals.

Interactions by year with hook and line gear were as follows: During 2015, there was one mortality. During 2016, there were three mortalities, two live animals considered seriously injured, and one live animal for which it could not be determined if it was seriously injured following disentanglement efforts (the initial determination was seriously injured). During 2017, there were four mortalities. During 2018, there were three mortalities and two live animals considered not seriously injured following disentanglement efforts (the initial determinations were seriously injured; one animal was initially sighted in 2018 and later disentangled in 2019). During 2019, there were three mortalities and one live animal considered not seriously injured following disentanglement efforts (the initial determination was seriously injured).

The mortalities and serious injuries likely involved animals from the following BSE stocks: Laguna Madre, Mobile Bay/Bonsecour Bay, Perdido Bay, Waccasassa Bay/Withlacoochee Bay/Crystal Bay, St. Joseph Sound/Clearwater Harbor, Tampa Bay, Sarasota Bay/Little Sarasota Bay, Pine Island Sound/Charlotte Harbor/Gasparilla Sound/Lemon Bay, and Estero Bay.

All mortalities and live entanglements were included in the stranding database (NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 25 August 2020) and are included in the stranding totals presented in Table 4. The details for serious determinations for the live animals are provided in Maze-Foley and Garrison (2021).

It should be noted that, in general, it cannot be determined if rod and reel hook and line gear originated from a commercial (i.e., charter boat 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

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 Mississippi River Delta Stock, the model predicted the stock experienced a 71% (95%CI: 40–97) maximum reduction in population size due to the oil spill (DWH MMIQT 2015; Schwacke *et al.* 2017), and for the years 2015–2019, the model projected 45 mortalities. For the Mobile Bay/Bonsecour Bay Stock, the model predicted a 31% (95% CI: 20–51) maximum reduction in population size due to the oil spill (DWH MMIQT 2015; Schwacke *et al*. 2017), and for the years 2015–2019, the model projected 73 mortalities. This population model has a number of sources of uncertainty. Because no current abundance estimates existed at the time of the spill, the baseline population sizes were estimated from studies initiated after initial exposure to DWH oil occurred. Therefore, it is possible that the pre-spill population sizes were 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).

There were two live dolphins during 2015–2019 that were entangled in unidentified fishing gear or unidentified gear, and one occurred in the Pine Island Sound/Charlotte Harbor/Gasparilla Sound/Lemon Bay Stock area in 2017 and the other occurred in Perdido Bay in 2018. The animal from 2018 was considered not seriously injured, and the 2017 animal was initially considered seriously injured, but following mitigation efforts, was released alive without serious injury (Maze-Foley and Garrison 2018). During 2015, an animal in the St. Joseph Sound/Clearwater Harbor Stock area (Florida) was released alive without serious injury following entrapment behind an oil boom (Maze-Foley and Garrison 2018). During 2016, there was a dead dolphin in the Copano Bay/Aransas Bay/San Antonio Bay/Redfish Bay/Espiritu Santo Bay Stock area found entangled in electrical cord. It is unknown whether the entanglement contributed to the death of the animal. All of these cases were included in the stranding database (NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 25 August 2020) and are included in the stranding totals presented in Table 4.

In addition to animals included in the stranding database, during 2015–2019, there were 42 at-sea observations in BSE stock areas of common bottlenose dolphins entangled in fishing gear or unidentified gear (hook and line, crab trap/pot and unidentified gear/line/rope) or displaying vessel-strike injuries. In 27 of these cases, the animals were seriously injured; in six cases the animals were not seriously injured, and for the remaining nine cases, it could not be determined (CBD) if the animals were seriously injured (Maze-Foley and Garrison 2021; see Table 2).

Table 2. At-sea observations of common bottlenose dolphins entangled in fishing gear or unidentified gear during 2015–2019, including the serious injury determination (mortality, serious injury, not a serious injury [Not serious], or could not be determined [CBD] if seriously injured) and stock to which each animal likely belonged based on sighting location. Further details can be found in Maze-Foley and Garrison (2021).

Year (Identifier from Maze-Foley and Garrison [2021])	Determination	Stock
2015 (row 92)	Serious injury	Calcasieu Lake
2015 (row 93)	Not serious	Tampa Bay
2015 (row 98)	Serious injury	Tampa Bay
2015 (row 99)	Serious injury	Laguna Madre
2015 (row 101)	CBD	Sarasota Bay/Little Sarasota Bay
2015 (row 102)	Serious injury	St. Joseph Sound/Clearwater Harbor
2015 (row 104)	CBD	Mobile Bay/Bonsecour Bay (or Northern Coastal)
2015 (row 106)	Not serious	Sarasota Bay/Little Sarasota Bay
2015 (row 109)	CBD	Apalachee Bay

2016 (row 120)	Serious injury	Laguna Madre
2016 (row 126)	CBD	St. Joseph Sound/Clearwater Harbor
2016 (row 127)	Serious injury	Mobile Bay/Bonsecour Bay
2017 (row 129)	CBD	Sarasota Bay/Little Sarasota Bay
2017 (row 130)	CBD	Sarasota Bay/Little Sarasota Bay
2017 (row 131)	Serious injury	undefined stock area (Miller's Bayou, Florida; in between the Waccasassa Bay/Withlacoochee Bay/Crystal Bay Stock and the St. Joseph Sound/Clearwater Harbor Stock)
2017 (row 135)	Serious injury	Sarasota Bay/Little Sarasota Bay
2017 (row 137)	Serious injury	Copano Bay/Aransas Bay/San Antonio Bay/ Redfish Bay/Espiritu Santo Bay
2017 (row 139)	Serious injury	Tampa Bay
2017 (row 140)	Serious injury	Tampa Bay
2017 (row 148)	Serious injury	Tampa Bay
2017 (row 150)	Serious injury	St. Joseph Sound/Clearwater Harbor
2018 (row 153)	Serious injury	Tampa Bay (or Eastern Coastal)
2018 (row 155)	CBD	Tampa Bay
2018 (row 156)	CBD	St. Joseph Sound/Clearwater Harbor (or Eastern Coastal)
2018 (row 158)	Not serious	Chokoloskee Bay/Ten Thousand Islands/ Gullivan Bay
2018 (row 160)	Serious injury	Estero Bay
2018 (row 162)	Serious injury	Laguna Madre
2018 (row 166)	Serious injury	Perdido Bay
2018 (row 168)	CBD	Perdido Bay
2018 (row 171)	Serious injury	Tampa Bay
2018 (row 25, vessel strike tab)	Serious injury	Perdido Bay
2019 (row 172)	Not serious	Chokoloskee Bay/Ten Thousand Islands/ Gullivan Bay
2019 (row 173)	Serious injury	Chokoloskee Bay/Ten Thousand Islands/ Gullivan Bay

2019 (row 175)	Serious injury	Pine Island Sound/Charlotte Harbor/ Gasparilla Sound/Lemon Bay
2019 (row 176)	Serious injury	Tampa Bay
2019 (row 179)	Not serious	St. Joseph Sound/Clearwater Harbor (or Eastern Coastal)
2019 (row 182/183)	Serious injury	Tampa Bay
2019 (row 189)	Serious injury	Tampa Bay
2019 (row 190)	Serious injury	Tampa Bay
2019 (row 192)	Not serious	Tampa Bay or St. Joseph Sound/Clearwater Harbor
2019 (row 194)	Serious injury	Laguna Madre
2019 (row 27, vessel strike tab)	Serious injury	Pine Island Sound/Charlotte Harbor/ Gasparilla Sound/Lemon Bay

Interactions between common bottlenose dolphins and research-fishery gear are also known to occur. During 2015–2019, nine dolphins were entangled in research-related gillnets—in Texas (7), Alabama (1) and Florida (1). One of the nine entanglements resulted in a mortality; five entanglements resulted in serious injuries; and three entanglements were released alive without serious injury (Maze-Foley and Garrison 2021; see Table 3). All of the interactions with research gear were included in the stranding database (NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 25 August 2020).

Table 3. Research-related takes of common bottlenose dolphins during 2015–2019, including the serious injury determination for each animal (mortality, serious injury, not a serious injury [Not serious], or could not be determined [CBD] if seriously injured) and stock to which each animal likely belonged based on location of the interaction. All of these interactions were included in the stranding database (NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 25 August 2020). Further details on injury determinations can be found in Maze-Foley and Garrison (2021).

Year	Gear Type	Determination	Stock
2015	Gillnet	Serious injury	Matagorda Bay/Tres Palacios Bay/Lavaca Bay
2016	Gillnet	Serious injury	Matagorda Bay/Tres Palacios Bay/Lavaca Bay
2016	Gillnet	Not serious	Laguna Madre
2017	Gillnet	Serious injury	Copano Bay/Aransas Bay/San Antonio Bay/ Redfish Bay/Espiritu Santo Bay
2018	Gillnet	Not serious	Sarasota Bay/Little Sarasota Bay
2019	Gillnet	Not serious	Perdido Bay
2019	Gillnet	Mortality	Nueces Bay/Corpus Christi Bay
2019	Gillnet	Serious injury	Laguna Madre
2019	Gillnet	Serious injury	Copano Bay/Aransas Bay/San Antonio Bay/ Redfish Bay/Espiritu Santo Bay

NOAA's Office of Law Enforcement has been investigating increasing numbers of reports from the northern Gulf of Mexico coast of violence against common bottlenose dolphins, including shootings using guns and bows and arrows, throwing pipe bombs and cherry bombs, and stabbings (Vail 2016). There have been several documented stabbings of BSE common bottlenose dolphins in recent years. In 2018, an animal was impaled by a metal rod resulting in mortality, and this mortality was ascribed to the Pensacola Bay/East Bay Stock. Also in 2018, an animal ascribed to the Tampa Bay Stock was documented with a puncture wound associated with fractured vertebrae and a necrotic tissue track, likely resulting in mortality. In 2019, an animal was stabbed/impaled in its head with a spear-like object while the animal was still alive, resulting in mortality. This animal was ascribed to the Pine Island Sound/Charlotte Harbor/Gasparilla Sound/Lemon Bay Stock. All three of these cases were included in the stranding database (NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 25 August 2020) and in Table 4.

Depredation of fishing catch and/or bait 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 of wild common bottlenose dolphins may encourage depredation because provisioning conditions dolphins to approach humans and vessels, where they then may prey on bait and catches (Vail 2016). Christiansen *et al.* (2016) found that via direct and indirect food provisioning, an increasing percentage of the long-term Sarasota Bay residents were becoming conditioned to human interactions. In addition, when comparing conditioned to unconditioned dolphins, Christiansen *et al.* (2016) reported it was more likely for a conditioned dolphin to be injured by human interactions.

Illegal feeding or provisioning of wild common bottlenose dolphins has been documented in Florida, particularly near St. Andrew Bay (Panama City Beach) in the Panhandle (Samuels and Bejder 2004; Powell *et al.* 2018) and in and near Sarasota Bay (Cunningham-Smith *et al.* 2006; Powell and Wells 2011), and also in Texas near Corpus Christi (Bryant 1994). Feeding wild dolphins is defined under the MMPA as a form of 'take' because it can alter their natural behavior and increase their risk of injury or death. Nevertheless, a high rate of provisioning was observed near Panama City Beach in 1998 (Samuels and Bejder 2004) and in 2014 (Powell *et al.* 2018), and provisioning was observed frequently and predictably south of Sarasota Bay during 1990–2007 (Cunningham-Smith *et al*. 2006; Powell and Wells 2011). Provisioning of four dolphins was documented within the Tampa Bay Stock area during 2019 while the dolphins were swimming in a local canal (NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 25 August 2020). There are emerging questions regarding potential linkages between provisioning and depredation of recreational fishing gear and associated entanglement and ingestion of gear, which is increasing through much of Florida. During 2006, at least 2% of the long-term resident dolphins of Sarasota Bay died from ingestion of recreational fishing gear (Powell and Wells 2011).

Swimming with wild common bottlenose dolphins has also been documented in Florida in Key West (Samuels and Engleby 2007) and near Panama City Beach (Samuels and Bejder 2004). Near Panama City Beach, Samuels and Bejder (2004) concluded that dolphins were amenable to swimmers due to illegal provisioning. Swimming with wild dolphins may cause harassment, and harassment is illegal under the MMPA.

As noted previously, common bottlenose dolphins are known to be struck by vessels (Wells and Scott 1997; Wells *et al.* 2008). During 2015–2019, 16 stranded bottlenose dolphins (of 523 total strandings) showed signs of a boat collision (NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 25 August 2020). It is possible some of the instances were post-mortem collisions. In addition to vessel collisions, the presence of vessels may also impact common bottlenose dolphin behavior in bays, sounds and estuaries. Nowacek *et al.* (2001) reported that boats pass within 100 m of each bottlenose dolphin in Sarasota Bay once every six minutes on average, leading to changes in dive patterns and group cohesion. Buckstaff (2004) noted changes in communication patterns of Sarasota Bay dolphins when boats approached. Miller *et al*. (2008) investigated the immediate responses of common bottlenose dolphins to "high-speed personal watercraft" (i.e., recreational 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 that dolphins attempted to avoid high-speed personal watercraft. It is likely that repeated short-term effects will result in long-term consequences like reduced health and viability or habitat displacement of dolphins (Bejder *et al*. 2006). Further studies are needed to determine the impacts throughout the Gulf of Mexico.

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. Historically, there have been interactions, including mortalities, documented for common bottlenose dolphins likely belonging to BSE stocks. However, no interactions with common bottlenose dolphins have been documented during the most recent five years, 2015–2019.

Historically, there have been two documented mortalities of common bottlenose dolphins during health-assessment research projects in the Gulf of Mexico, but none have occurred during the most recent five years, 2015–2019.

Some of the BSE communities were the focus of a live-capture fishery for common bottlenose dolphins which supplied dolphins to the U.S. Navy and to oceanaria and laboratories for research and public display for more than two decades (Reeves and Leatherwood 1984; Scott 1990). Between 1973 and 1988, 533 common bottlenose dolphins were removed from Southeastern U.S. waters (Scott 1990). The impact of these removals on the stocks is unknown. In 1989, the Alliance of Marine Mammal Parks and Aquariums declared a self-imposed moratorium on the capture of common bottlenose dolphins in the Gulf of Mexico (Corkeron 2009).

Strandings

During 2015–2019, 527 common bottlenose dolphins were found stranded within bays, sounds and estuaries of the northern Gulf of Mexico (Table 4; NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 25 August 2020). There was evidence of human interaction (HI) for 102 of the strandings. No evidence of human interaction was detected for 25 strandings, and for the remaining 400 strandings, it could not be determined if there was evidence of human interaction. Human interactions were from numerous sources, including entanglements with hook and line gear, trap/pot gear, commercial shrimp trawl gear, research gillnet gear, stabbings/impalements, an entrapment between oil booms, and animals with evidence of a vessel strike (see Table 4). Strandings with evidence of fishery-related interactions are reported above in the respective gear sections. It should be noted that evidence of human interaction does not necessarily mean the interaction caused the animal's stranding or death.

There are a number of difficulties associated with the interpretation of stranding data. Except in rare cases, such as Sarasota Bay, Florida, where residency can be determined, it is possible that some or all of the stranded dolphins may have been from a nearby coastal stock. However, the proportion of stranded dolphins belonging to another stock cannot be determined because of the difficulty of determining from where the stranded carcasses originated. Stranding data underestimate the extent of human and fishery-related mortality and serious injury because not all of the dolphins that die or are seriously injured in human interactions wash ashore, or, if they do, they are not all recovered (Peltier *et al.* 2012; Wells *et al.* 2015; Carretta *et al.* 2016). Additionally, not all carcasses will show evidence of human interaction, entanglement, or other fishery-related interaction due to decomposition, scavenger damage, etc. (Byrd *et al.* 2014). Finally, the level of technical expertise among stranding network personnel varies widely as does the ability to recognize signs of human interaction.

Since 1990, there have been 15 common bottlenose dolphin die-offs or Unusual Mortality Events (UMEs) in the northern Gulf of Mexico (Litz *et al.* 2014; http://www.nmfs.noaa.gov/pr/health/mmume/events.html, accessed 5 November 2020).

1) From January through May 1990, 344 common bottlenose dolphins stranded in the northern Gulf of Mexico. Overall this represented a two-fold increase in the prior maximum recorded number of strandings for the same period, 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).

2) A UME was declared for Sarasota Bay, Florida, in 1991 involving 31 common bottlenose dolphins. The cause was not determined, but it is believed biotoxins may have contributed to this event (Litz *et al.* 2014).

3) In March and April 1992, 119 common bottlenose dolphins stranded in Texas - about nine times the average number. The cause of this event was not determined, but low salinity due to record rainfall combined with pesticide runoff and exposure to morbillivirus were suggested as potential contributing factors (Duignan *et al.* 1996; Colbert *et al.* 1999; Litz *et al.* 2014).

4) In 1993–1994 a UME of common bottlenose dolphins caused by morbillivirus started in the Florida Panhandle and spread west with most of the mortalities occurring in Texas (Lipscomb *et al.* 1994; Litz *et al.* 2014). From February through April 1994, 236 common bottlenose dolphins were found dead on Texas beaches, of which 67 occurred in a single 10-day period.

5) In 1996 a UME was declared for common bottlenose dolphins in Mississippi when 31 common bottlenose dolphins stranded during November and December. The cause was not determined, but a *Karenia brevis* (red tide) harmful algal bloom was suspected to be responsible (Litz *et al.* 2014).

6) Between August 1999 and May 2000, 150 common bottlenose dolphins died coincident with *K. brevis* blooms and fish kills in the Florida Panhandle (additional strandings included three Atlantic spotted dolphins, *Stenella frontalis*, one Risso's dolphin, *Grampus griseus*, two Blainville's beaked whales, *Mesoplodon densirostris*, and four unidentified dolphins. Brevetoxin was determined to be the cause of this event (Twiner *et al.* 2012; Litz *et al.* 2014).

7) In March and April 2004, in another Florida Panhandle UME attributed to *K. brevis* blooms, 105 common bottlenose dolphins and two unidentified dolphins stranded dead (Litz *et al.* 2014). Although there was no indication of a *K. brevis* bloom at the time, high levels of brevetoxin were found in the stomach contents of the stranded dolphins (Flewelling *et al.* 2005; Twiner *et al.* 2012).

8) In 2005, a particularly destructive red tide (*K. brevis*) bloom occurred off central west Florida. Manatee, sea turtle, bird and fish mortalities were reported in the area in early 2005 and a manatee UME had been declared. Dolphin mortalities began to rise above the historical averages by late July 2005, continued to increase through October 2005, and were then declared to be part of a multi-species UME. The multi-species UME extended into 2006, and ended in November 2006. In total, 190 dolphins were involved, primarily common bottlenose dolphins (plus strandings of one Atlantic spotted dolphin and 23 unidentified dolphins). The evidence suggests a red tide bloom contributed to the cause of this event (Litz *et al.* 2014).

9) A separate UME was declared in the Florida Panhandle after elevated numbers of dolphin strandings occurred in association with a *K. brevis* bloom in September 2005. Dolphin strandings remained elevated through the spring of 2006 and brevetoxin was again detected in the tissues of most of the stranded dolphins and determined to be the cause of the event (Twiner *et al.* 2012; Litz *et al.* 2014). Between September 2005 and April 2006 when the event was officially declared over, a total of 88 common bottlenose dolphin strandings occurred (plus strandings of five unidentified dolphins).

10) During February and March of 2007 an event was declared for northeast Texas and western Louisiana involving 64 common bottlenose dolphins and two unidentified dolphins. Decomposition prevented conclusive analyses on most carcasses (Litz *et al.* 2014).

11) During February and March of 2008 an additional event was declared in Texas involving 111 common bottlenose dolphin strandings (plus strandings of one unidentified dolphin and one melon-headed whale, *Peponocephala electra*). Most of the animals recovered were in a decomposed state. A direct cause could not be identified. However, there were numerous, co-occurring harmful algal bloom toxins detected during the time period of this UME which may have contributed to the mortalities (Fire *et al.* 2011).

12) A UME was declared for cetaceans in the northern Gulf of Mexico beginning 1 February 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 UME began a few months prior to the DWH oil spill, however most of the strandings prior to May 2010 were in Lake Pontchartrain, Louisiana, and western Mississippi and were likely a result of low salinity and cold temperatures (Venn-Watson *et al.* 2015a). The largest increase in strandings (compared to historical data) occurred after May 2010 following the DWH spill, and strandings were focused in areas exposed to DWH oil. 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.* 2015b; Colegrove *et al.* 2016; DWH NRDAT 2016; see Habitat Issues section).

13) A UME occurred from November 2011 to March 2012 across five Texas counties and included 126 common bottlenose dolphin strandings. The strandings were coincident with a harmful algal bloom of *K. brevis*, but researchers have not determined that was the cause of the event. During 2011, six animals from BSE stocks were considered to be part of the UME; during 2012, 24 animals.

14) A bottlenose dolphin UME occurred in southwest Florida from 1 July 2018 through 30 June 2019, with peak strandings occurring between 1 July 2018 and 30 April 2019. A total of 183 dolphins were reported (note the dates and numbers are subject to change as the closure package has not yet been approved by the UME Working Group). All age classes of dolphins were represented and the majority of the animals recovered were in moderate to advanced stages of decomposition. The cause of the bottlenose dolphin UME was determined to be due to biotoxin exposure

from the *K. brevis* harmful algal bloom off the coast of southwest Florida. The additional supporting evidence of fish kills and other species die-offs linked to brevetoxin during the same time and space support that the impacts of the harmful algal bloom caused the dolphin mortalities (Rycyk *et al.* 2020).

15) During 1 February 2019 to 30 November 2019, a UME was declared for the area from the eastern border of Taylor County, Florida, west through Alabama, Mississippi, and Louisiana (http://www.nmfs.noaa.gov/pr/health/mmume/cetacean_gulfofmexico.htm, accessed 5 November 2020). A total of 337 common bottlenose dolphins stranded during this event. The largest number of mortalities occurred in eastern Louisiana and Mississippi. An investigation concluded the event was caused by exposure to low salinity waters as a result of extreme freshwater discharge from rivers. The unprecedented amount of freshwater discharge during 2019 (e.g., Gasparini and Yuill 2020) resulted in low salinity levels across the region.

Table 4. Common bottlenose dolphin strandings occurring in bays, sounds, and estuaries in the northern Gulf of Mexico from 2015 to 2019, 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. Data are from the NOAA National Marine Mammal Health and Stranding Response Database (unpublished data, accessed 25 August 2020). Please note HI does not necessarily mean the interaction caused the animal's death. Please also note that this table does not include strandings from West Bay, Galveston Bay/East Bay/Trinity Bay, Terrebonne-Timbalier Bay Estuarine System, Barataria Bay Estuarine System, Mississippi Sound/Lake Borgne/Bay Boudreau, Choctawhatchee Bay, St. Andrew Bay, and St. Joseph Bay.

Category	2015	2016	2017	2018	2019	Total
Total Stranded	68	87	91	115	166	527
HI--Yes	12[a]	23[b]	18[c]	16[d]	33[e]	102
HI--No	1	3	7	8	6	25
HI--CBD	55	61	66	91	127	400

a. Includes 1 entanglement interaction with hook and line gear (mortality); 1 entanglement interaction in commercial blue crab trap/pot gear (mortality); 1 entanglement interaction with unidentified trap/pot gear (released alive, could not be determined if seriously injured or not); 1 entanglement interaction with research gillnet gear (released alive, seriously injured); 1 live release without serious injury following entrapment between oil booms (animal was initially seriously injured, but due to mitigation efforts, was released without serious injury); and 3 animals with evidence of a vessel strike (2 mortalities, 1 live animal without serious injury).

b. Includes 6 entanglement interactions with hook and line gear (3 mortalities [1 also had evidence of a vessel strike and 1 had evidence of entanglement with shrimp trawl gear] and 3 released alive seriously injured); 6 mortalities with evidence of a vessel strike (1 was also an entanglement interaction with hook and line gear); 1 entanglement interaction with trap/pot gear (released alive, seriously injured); 1 entanglement interaction with commercial stone crab trap/pot gear (live animal without serious injury); 1 entanglement interaction with research gillnet gear (released alive, seriously injured); and 1 entanglement interaction with shrimp trawl gear (mortality, also an interaction with hook and line gear); and 1 animal with markings indicative of interaction with gillnet gear (mortality).

c. Includes 3 entanglement interactions with hook and line gear (mortalities), 1 entanglement interaction with commercial blue crab trap/pot gear (mortality); 1 entanglement interaction with trap/pot gear (released alive, seriously injured); 1 entanglement interaction with research gillnet gear (released alive, seriously injured); and 4 animals with evidence of a vessel strike (mortalities).

d. Includes 5 entanglement interactions with hook and line gear (3 mortalities and 2 animals initially seriously injured, but due to mitigation efforts, were released alive without serious injury); 2 stabbings (mortalities); 1 animal with markings indicative of interaction with gillnet gear (mortality); and 1 entanglement in possible gillnet gear (live animal without serious injury)

e. Includes 4 entanglement interactions with hook and line gear (3 mortalities and 1 animal initially seriously injured, but due to mitigation efforts, was released alive without serious injury); 1 stabbing (mortality); 3 animals with evidence of a vessel strike (mortalities); 1 entanglement interaction with commercial blue crab trap/pot gear (animal was initially seriously injured, but due to mitigation efforts, was released without serious injury); 1 entanglement interaction with crab trap/pot gear (mortality); 1 entanglement interaction with commercial spiny lobster trap/pot gear (seriously injured); 1 animal with markings indicative of interaction with gillnet gear (mortality); 4 entanglement interactions with research gillnet gear (1 mortality and 3 live animals, 2 of which were seriously injured and 1 without serious injury); and 1 interaction with unidentified gillnet gear (mortality).

HABITAT ISSUES

Issues Related to the *Deepwater Horizon* (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 west of the Mississippi River on the Mississippi Delta and in Barataria and Terrebonne Bays, and to the east of the river on the Chandeleur Islands. Some heavy to moderate oiling occurred on Alabama and Florida beaches, with the heaviest stretch occurring from Dauphin Island, Alabama, to Gulf Breeze, Florida. Light to trace oil was reported along the majority of Mississippi's mainland coast, from Gulf Breeze to Panama City, Florida, and outside of Atchafalaya and Vermilion Bays in western Louisiana. 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. These studies estimated that for the Mississippi River Delta Stock of common bottlenose dolphins, 46% (95%CI: 21–65) of females suffered from reproductive failure, and 37% (95%CI: 14–57) suffered adverse health effects (DWH MMIQT 2015). A population model estimated that the stock experienced a 71% maximum reduction in population size (see Other Mortality section above). For the Mobile Bay Stock of common bottlenose dolphins, 46% (95%CI: 21–65) of females suffered from reproductive failure, and 24% (95%CI: 0–48) suffered adverse health effects (DWH MMIQT 2015). The population model estimated that the stock experienced a 31% maximum reduction in population size (see Other Mortality section above).

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). 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.

The DWH NRDA Trustees quantified injuries to four BSE stocks of common bottlenose dolphins, including two stocks included in this report, the Mississippi River Delta Stock and the Mobile Bay/Bonsecour Bay Stock, as well two stocks that have their own SARs (Barataria Bay Estuarine System Stock and Mississippi Sound/Lake Borgne/Bay Bourdreau Stock). A suite of research efforts indicated the DWH oil spill negatively affected these stocks 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). Colegrove *et al*. (2016) 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 *in utero* infections, including pneumonia and brucellosis.

In the absence of any additional non-natural mortality or restoration efforts, the DWH damage assessment estimated the Mississippi River Delta Stock will take 52 years to recover to pre-spill population size, and the Mobile Bay/Bonsecour Bay Stock, 31 years (DWH MMIQT 2015).

Other Habitat Issues

The nearshore habitat occupied by many of these stocks is adjacent to areas of high human population, and in some bays, such as Mobile Bay in Alabama and Galveston Bay in Texas, is highly industrialized. Many of the enclosed bays in Texas are surrounded by agricultural lands that receive periodic pesticide applications.

Concentrations of chlorinated hydrocarbons and metals were examined in conjunction with an anomalous mortality event of common bottlenose dolphins in Texas bays in 1990 and found to be relatively low in most; however, some had concentrations at levels of possible toxicological concern (Varanasi *et al*. 1992). No studies to date have determined the amount, if any, of indirect human-induced mortality resulting from pollution or habitat degradation.

Analyses of organochlorine concentrations in the tissues of common bottlenose dolphins in Sarasota Bay, Florida, have found that the concentrations in male dolphins exceeded toxic threshold values that may result in adverse effects on health or reproductive rates (Schwacke *et al*. 2002). 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). While there are no direct measurements of adverse effects of pollutants on estuary dolphins, the exposure to environmental pollutants and subsequent effects on population health are areas of concern and active research.

STATUS OF STOCKS

The status of these stocks relative to optimum sustainable population is unknown and this species is not listed as threatened or endangered under the Endangered Species Act. The occurrence of 15 Unusual Mortality Events (UMEs) among common bottlenose dolphins along the northern Gulf of Mexico coast since 1990 (Litz *et al*. 2014; http://www.nmfs.noaa.gov/pr/health/mmume/events.html, accessed 5 November 2020) is cause for concern. Notably, stock areas in Louisiana, Mississippi, Alabama, and the western Florida panhandle have recently been impacted by several UMEs. However, the effects of the mortality events on stock abundance have not yet been determined, in large part because it has not been possible to assign mortalities to specific stocks and a lack of current abundance estimates for some stocks.

Human-caused mortality and serious injury for each of these stocks is unknown. Considering the evidence from stranding data (Table 4) and the low PBRs for stocks with recent abundance estimates, the total fishery-related mortality and serious injury likely exceeds 10% of the total known PBR or previous PBR, and therefore, it is probably not insignificant and not approaching the zero mortality and serious injury rate. NMFS considers each of these stocks, except for the Sabine Lake, Mississippi River Delta, and Sarasota Bay/Little Sarasota Bay stocks, to be strategic because most of the stock sizes are currently unknown, but are likely small such that relatively few mortalities and serious injuries would exceed PBR.

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ATLANTIC SPOTTED DOLPHIN (*Stenella frontalis*): Northern Gulf of Mexico Stock

STOCK DEFINITION AND GEOGRAPHIC RANGE

There are two species of spotted dolphins in the Atlantic Ocean, the Atlantic spotted dolphin (*Stenella frontalis*) and the pantropical spotted dolphin (*S. attenuate*; Perrin *et al.* 1987). The Atlantic spotted dolphin occurs in two forms which may be distinct subspecies (Perrin *et al*. 1987, 1994; Rice 1998; Viricel and Rosel 2014): the large, heavily spotted form which inhabits the continental shelf and is usually found inside or near the 200-m isobath; and the smaller, less spotted island and offshore form which occurs in the Atlantic Ocean but is not known to occur in the Gulf of Mexico (Fulling *et al.* 2003; Mullin and Fulling 2003; Mullin and Fulling 2004; Viricel and Rosel 2014). Where they co-occur, the offshore form of the Atlantic spotted dolphin and the pantropical spotted dolphin can be difficult to differentiate at sea.

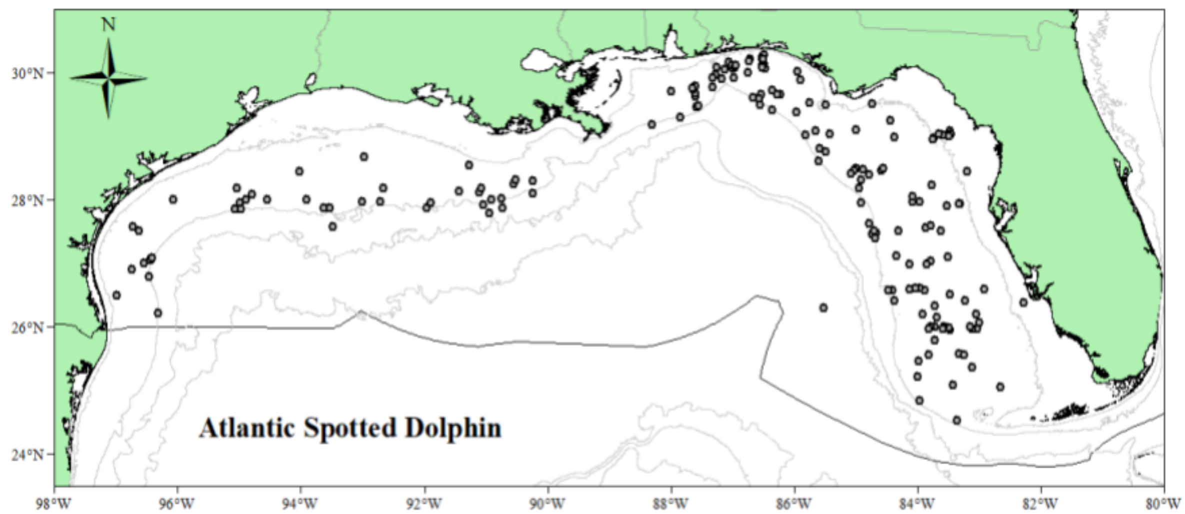

Figure 1. Distribution of Atlantic spotted dolphin on-effort sightings from SEFSC spring and fall vessel surveys during 1996–2001, vessel surveys during summer 2003, spring 2004, summer 2009, summer 2017, and summer/fall 2018, and aerial surveys during spring 2011, summer 2011, fall 2011, winter 2012, summer 2017, winter 2018, and fall 2018. Isobaths are the 20-m, 200-m, 1,000-m, and 2,000-m depth contours. The darker line indicates the U.S. EEZ.

The Atlantic spotted dolphin is endemic to the Atlantic Ocean in temperate to tropical waters (Perrin *et al.* 1987, 1994). In the Gulf of Mexico, Atlantic spotted dolphins occur primarily from continental shelf waters 10–200 m deep to slope waters <500 m deep (Figure 1; Fulling *et al.* 2003; Mullin and Fulling 2004; Maze-Foley and Mullin 2006; Garrison and Aichinger Dias 2020). Atlantic spotted dolphins were seen in all seasons during aerial and vessel surveys of the northern Gulf of Mexico (i.e., U.S. Gulf of Mexico; Hansen *et al.* 1996; Mullin and Hoggard 2000; Fulling *et al.* 2003; Mullin and Fulling 2004; Maze-Foley and Mullin 2006; Garrison and Aichinger Dias 2020). It has been suggested that this species may move inshore seasonally during spring, but data supporting this hypothesis are limited (Caldwell and Caldwell 1966; Fritts *et al.* 1983).

All the cetacean species found in the northern Gulf of Mexico almost certainly occur in similar habitat beyond U.S. boundaries in the southern Gulf. There are fewer cetacean sighting and stranding records in the southern Gulf due to more limited effort. Because there are confirmed records from the southern Gulf of Mexico beyond U.S.

boundaries (e.g., Jefferson and Schiro 1997; Ortega Ortiz 2002), this is likely a transboundary stock with Mexico.

Genetic analysis of Atlantic spotted dolphins in the Gulf of Mexico and western North Atlantic revealed significant differentiation for both nuclear and mitochondrial DNA markers (Adams and Rosel 2005; Viricel and Rosel 2014). Estimates of immigration rates between the western North Atlantic shelf population and the Gulf of Mexico were less than 1% per year (Viricel and Rosel 2014), which is well below the 10% per year threshold for demographic independence (Hastings 1993), thereby supporting separate stocks for Gulf of Mexico and western North Atlantic shelf populations. Viricel and Rosel (2014) also found support for two demographically independent populations within the northern Gulf of Mexico. One population primarily occupied shelf waters from the Texas-Mexico border eastward to Cape San Blas, Florida, while the second population was concentrated over the Florida shelf in the eastern Gulf of Mexico and stretched westward to the Florida panhandle. Thus, the two populations appear to overlap slightly in shelf waters between approximately Mobile Bay and Cape San Blas. Additional work is necessary to identify a boundary between them.

POPULATION SIZE

The current population size for the Atlantic spotted dolphin in the northern Gulf of Mexico is 21,506 (CV=0.26; Table 1). This estimate combines an estimate from an aerial survey during summer 2017 covering waters over the continental shelf (Garrison *et al.* 2021) and an estimate from summer 2017/2018 that covers oceanic waters (Garrison *et al.* 2020).

Earlier Abundance Estimates

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

Recent Surveys and Abundance Estimates

An abundance estimate for Atlantic spotted dolphins was generated from vessel surveys conducted in the northern Gulf of Mexico from the continental shelf edge (~200-m isobath) to the seaward extent of the U.S. EEZ (Table 1; Garrison *et al.* 2020). One survey was conducted from 2 July to 25 August 2017 and consisted of 7,302 km of on-effort trackline, and the second survey was conducted from 11 August to 6 October 2018 and consisted of 6,473 km of on-effort trackline within the surveyed strata. Both surveys used a double-platform data-collection procedure, which allowed estimation of the detection probability on the trackline using the independent observer approach assuming point independence (Laake and Borchers 2004). Abundance was calculated using mark-recapture distance sampling implemented in package mrds (version 2.21; Laake *et al.* 2020) in the R statistical programming language. This approach accounted for the effects of covariates (e.g., sea state, glare) on detection probability within the surveyed strip. The surveys were conducted in "passing mode" (e.g., Schwarz *et al.* 2010) while all prior surveys in the Gulf of Mexico have been conducted in "closing mode." Passing mode eliminates the problems of fragmented tracklines associated with using closing mode in areas with high densities of animals. When using the closing mode with the two-team method, both teams must be allowed the opportunity to see a mammal group and allow it to pass behind the ship before turning to close on it, making it difficult to reacquire the group and resulting in long periods spent chasing the group, with the increased potential for off-effort sightings. For passive acoustics, in closing mode the vessel often turns before the acoustic team is able to achieve a good localization. This is especially important for deep-diving species where visual surveys are less optimal for abundance estimates. However, passing mode can result in increased numbers of unidentified sightings and may have affected group size estimation for distant groups of dolphins and small whales. The abundance estimate for this stock included sightings of unidentified dolphins that were apportioned among identified species based on their relative density within the survey strata (Garrison *et al.* 2020). The inverse variance weighted mean abundance estimate for Atlantic spotted dolphins in oceanic waters during 2017 and 2018 was 5,577 (CV=0.41; Garrison *et al.* 2020). Unlike previous abundance estimates, this estimate was corrected for the probability of detection on the trackline.

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 summer (June–August) 2017 and fall (October–November) 2018 (Garrison *et al.* 2021). The stock was only partially surveyed during a winter 2018 aerial survey, and therefore this survey was not included in the current abundance estimates (Garrison *et al.* 2021). The surveys were conducted along tracklines oriented perpendicular to the shoreline and spaced 20 km apart. The total survey effort varied during each survey due to weather conditions, and was 10,781 km (fall) and 14,590 km (summer). 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). Abundance was calculated using mark-recapture distance sampling implemented in package mrds (version 2.21; Laake *et al.* 2020) in the R statistical programming language. This approach estimates both the probability of detection on the trackline and within the surveyed strip accounting for the effects of sighting conditions (e.g., sea state, glare, turbidity, and cloud cover). A different detection probability model was used for each seasonal survey (Garrison *et al.* 2021). The survey data were post-stratified into spatial boundaries corresponding to the defined boundaries of common bottlenose dolphin stocks within the surveyed area. The abundance estimates for the Continental Shelf Stock of common bottlenose dolphins were based upon tracklines and sightings in waters from the 20-m to the 200-m isobaths and between the Texas-Mexico border and the Florida Keys. The seasonal abundance estimates for this stock were: summer – 15,929 (CV=0.32) and fall – 2,529 (CV=0.71). Because the aerial survey estimate needs to be combined with vessel based estimates from surveys conducted during summer months, the summer 2017 aerial survey was used.

The best abundance estimate for Atlantic spotted dolphins is the sum of the estimates from continental shelf and oceanic waters during summer 2017–2018 surveys, and is 21,506 (CV=0.26; Table 1).

Table 1. Most recent abundance estimate (N_{est}) and coefficient of variation (CV) of northern Gulf of Mexico Atlantic spotted dolphins in continental shelf waters (coastline to 200-m isobath) and oceanic waters (200 m to the offshore extent of the EEZ) based on 2017 and 2018 aerial and vessel surveys.

Season/Year	Area	N_{est}	CV N_{est}
Summers 2017 and 2018	Oceanic	5,577	0.41
Summer 2017	Continental Shelf	15,929	0.32
Summers 2017 and 2018	Oceanic and Continental Shelf	21,506	0.26

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 Atlantic spotted dolphins is 21,506 (CV=0.26). The minimum population estimate for Atlantic spotted dolphins is 17,339 (Table 2).

Current Population Trend

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). Estimates of the portion of this stock over the continental shelf are available from aerial surveys conducted in 2011–2012 that can be compared to estimates from the 2017–2018 surveys. However, there is no corresponding vessel survey for the summer of 2011 that would allow an assessment of potential trend in the abundance of this stock. Therefore, no trend analysis can be conducted for the entire stock of Atlantic spotted dolphins.

CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

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

POTENTIAL BIOLOGICAL REMOVAL

Potential Biological Removal (PBR) is the product of the minimum population size, one half the maximum net productivity rate and a recovery factor (MMPA Sec. 3.16 U.S.C. 1362; Wade and Angliss 1997). The minimum population size of this stock is 17,339. The maximum productivity rate is 0.04, the default value for cetaceans. The recovery factor is 0.48 because the CV of the shrimp trawl mortality estimate is greater than 0.3 (Wade and Angliss 1997). PBR for the northern Gulf of Mexico Atlantic spotted dolphin is 166 (Table 2).

Table 2. Best and minimum abundance estimates for northern Gulf of Mexico Atlantic spotted dolphins with Maximum Productivity Rate (R_{max}), Recovery Factor (F_r) and PBR.

N_{est}	N_{est} CV	N_{min}	F_r	R_{max}	PBR
21,506	0.26	17,339	0.48	0.04	166

ANNUAL HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

Total annual estimated fishery-related mortality and serious injury for this stock during 2015–2019 was 36 (CV=0.47) Atlantic spotted dolphins based on observer data for the commercial shrimp trawl fishery (Table 3; see Fisheries Information section below). The mean annual mortality and serious injury during 2015–2019 due to the *Deepwater Horizon* (DWH) oil spill was projected to be 231 continental shelf dolphins, which includes both Atlantic spotted dolphins and the Continental Shelf Stock of common bottlenose dolphins. Therefore, the mean annual mortality and serious injury during 2015–2019 due to other human-caused actions (DWH oil spill) is unknown for this stock. The minimum total mean annual human-caused mortality and serious injury for this stock during 2015–2019 was, therefore, 36. This is considered a minimum because 1) not all fisheries that could interact with this stock are observed and/or observer coverage is very low, and 2) the population model used to estimate population decline for the northern Gulf of Mexico stocks impacted by the DWH oil spill includes both Atlantic spotted dolphins and common bottlenose dolphins inhabiting the continental shelf and does not estimate mortality and serious injury to the Atlantic spotted dolphin stock alone. Therefore no estimate for injury has been included for the Atlantic spotted dolphin stock due to the DWH oil spill.

Table 3. Total annual estimated fishery-related mortality and serious injury for northern Gulf of Mexico Atlantic spotted dolphins.

Years	Source	Annual Avg.	CV
2015–2019	U.S. fisheries using observer data	36	0.47

Fisheries Information

There are two commercial fisheries that interact, or that potentially could interact, with this stock in the Gulf of Mexico. These are the Category I Atlantic Ocean, Caribbean, Gulf of Mexico large pelagics longline fishery and the Category II Southeastern U.S. Atlantic, Gulf of Mexico shrimp trawl fishery. Detailed fishery information is presented in Appendix III.

Longline Fishery

Pelagic swordfish, tunas and billfish are the targets of the large pelagics longline fishery operating in the northern Gulf of Mexico. Percent observer coverage (percentage of sets observed) for this fishery for each year during 2015–2019 was 19, 23, 13, 20 and 13, respectively. There were no observed mortalities or serious injuries to Atlantic spotted dolphins by this fishery during 2015–2019 (Garrison and Stokes 2017, 2019, 2020a, 2020b, 2021).

Shrimp Trawl

Between 1997 and 2019, 13 common bottlenose dolphins and nine unidentified dolphins, which could have been either common bottlenose dolphins or Atlantic spotted dolphins, became entangled in the lazy line, turtle excluder device or tickler chain gear in observed trips of the commercial shrimp trawl fishery in the Gulf of Mexico (Soldevilla *et al.* 2021). All dolphin bycatch interactions resulted in mortalities except for one unidentified dolphin that was released alive in 2009 (Maze-Foley and Garrison 2016). Soldevilla *et al.* (2015, 2016, 2021) provided mortality estimates calculated from analysis of shrimp fishery effort data and NMFS's Observer Program bycatch data. Annual mortality estimates were calculated for the years 2015–2019 from stratified annual fishery effort and bycatch rates, and the five-year unweighted mean mortality estimate was calculated for Gulf of Mexico dolphin stocks (Soldevilla *et al.* 2021). The 4-area (TX, LA, MS/AL, FL) stratification method was chosen because it best approximates how fisheries operate (Soldevilla *et al.* 2021). The mean annual mortality estimate for the Atlantic spotted dolphin stock is 36 (CV=0.47). Limitations and biases of annual bycatch mortality estimates are described in detail in Soldevilla *et al.* (2021).

Other Mortality

A population model was developed to estimate the injury and time to recovery for stocks affected by the DWH oil spill, taking into account long-term effects resulting from mortality, reproductive failure, reduced survival rates, and the proportion of the stock exposed to DWH oil (DWH MMIQT 2015). Overall, the model estimated that continental shelf dolphins, including Atlantic spotted dolphins and the continental shelf stock of common bottlenose dolphins, experienced a 3% maximum reduction in population size due to the oil spill (DWH MMIQT 2015). The mortality projected for the years 2010–2014 due to the spill has not been reported previously. Based on the population model, it was projected that 3,384 continental shelf dolphins died during 2010–2014 (five year annual average of 677) due to elevated mortality associated with oil exposure (see Appendix VI). For the 2015–2019 reporting period of this SAR, the population model estimated 1,153 continental shelf dolphins died due to elevated mortality associated with oil exposure. The population model used to predict shelf dolphin mortality due to the DWH event has a number of sources of uncertainty. Model parameters (e.g., survival rates, reproductive rates, and life-history parameters) were derived from literature sources for common bottlenose dolphins occupying waters outside of the Gulf of Mexico. In addition, proxy values for the effects of DWH oil exposure on both survival rates and reproductive success were applied based upon estimated values for common bottlenose dolphins in Barataria Bay. Finally, there was no estimation of uncertainty in model parameters or outputs.

Although outside the time period of this report, it should be noted that there was an entanglement in seismic survey nodal line during 2014 that resulted in one mortality of an Atlantic spotted dolphin.

Strandings

Three Atlantic spotted dolphins were reported stranded in the Gulf of Mexico during 2015–2019 (NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 25 August 2020). One animal stranded in Alabama in 2017, one in Alabama in 2018, and one in Florida in 2019. For all three strandings, it could not be determined if there was evidence of human interaction.

There are a number of difficulties associated with the interpretation of stranding data. Stranding data underestimate the extent of human and fishery-related mortality and serious injury because not all of the dolphins that die or are seriously injured in human interactions wash ashore, or, if they do, they are not all recovered (Peltier *et al*. 2012; Wells *et al*. 2015; Carretta *et al.* 2016). Additionally, not all carcasses will show evidence of human interaction, entanglement or other fishery-related interaction due to decomposition, scavenger damage, etc. (Byrd *et al*. 2014). Finally, the level of technical expertise among stranding network personnel varies widely as does the ability to recognize signs of human interaction.

Since 1990, there have been 15 common bottlenose dolphin die-offs or Unusual Mortality Events (UMEs) in the northern Gulf of Mexico, and three of these included Atlantic spotted dolphins. 1) Between August 1999 and May 2000, 150 common bottlenose dolphins died coincident with *Karenia brevis* blooms and fish kills in the Florida Panhandle. Additional strandings included three Atlantic spotted dolphins, one Risso's dolphin, *Grampus griseus*, two Blainville's beaked whales, *Mesoplodon densirostris*, and four unidentified dolphins. Brevetoxin was determined to be the cause of this event (Twiner *et al.* 2012; Litz *et al.* 2014). 2) In 2005, a particularly destructive red tide (*K. brevis*) bloom occurred off of central west Florida. Manatee, sea turtle, bird and fish mortalities were reported in the area in early 2005 and a manatee UME had been declared. Common bottlenose dolphin mortalities began to rise above the historical averages by late July 2005, continued to increase through October 2005, and were then declared to be part of a multi-species UME. The multi-species UME extended into 2006, and ended in November 2006. A total of 190 dolphins were involved, primarily common bottlenose dolphins plus strandings of one Atlantic spotted dolphin and 23 unidentified dolphins. The evidence suggests the effects of a red tide bloom contributed to the cause of this event (Litz *et al.* 2014). 3) An Unusual Mortality Event (UME) was declared for cetaceans in the northern Gulf of Mexico beginning 1 March 2010 and ending 31 July 2014 (Litz *et al.* 2014; https://www.fisheries.noaa.gov/national/marine-life-distress/2010-2014-cetacean-unusual-mortality-event-northern-gulf-mexico). It included cetaceans that stranded prior to the DWH oil spill (see Habitat Issues section below), during the spill, and after. Exposure to the DWH oil spill was determined to be the primary underlying cause of the elevated stranding numbers in the northern Gulf of Mexico after the spill (e.g., Schwacke *et al.* 2014; Venn-Watson *et al.* 2015; Colegrove *et al.* 2016; DWH NRDAT 2016; see Habitat Issues section). Fourteen strandings of Atlantic spotted dolphins during 2010–2014 were considered to be part of this UME.

HABITAT ISSUES

The *Deepwater Horizon* (DWH) MC252 drilling platform, located approximately 80 km southeast of the

Mississippi River Delta in waters about 1,500 m deep, exploded on 20 April 2010. The rig sank, and over 87 days 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 Natural Resource Damage Assessment (NRDA) process was initiated under the Oil Pollution Act of 1990. A variety of NRDA research studies were conducted to determine potential impacts of the spill on marine mammals. These studies estimated that 13% (95%CI: 9–19) of continental shelf dolphins, including Atlantic spotted dolphins and the continental shelf stock of common bottlenose dolphins, in the Gulf were exposed to oil, that 6% (95%CI: 3–8) of females suffered from reproductive failure, and 5% (95%CI: 2–7) of continental shelf dolphins suffered adverse health effects (DWH MMIQT 2015). A population model estimated that the stock experienced a 3% maximum reduction in population size (see Other Mortality section above).

STATUS OF STOCK

Atlantic spotted dolphins are not listed as threatened or endangered under the Endangered Species Act, and the northern Gulf of Mexico stock is not considered strategic under the MMPA. 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. The status of Atlantic spotted dolphins in the northern Gulf of Mexico, relative to optimum sustainable population, is unknown. There are insufficient data to determine the population trends for this stock.

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Appendix I: Estimated mortality and serious injury (M/SI) of Western North Atlantic marine mammals listed by U.S. observed fisheries. Marine mammal species with zero (0) observed M/SI are not shown in this table. (unk = unknown)

Category, Fishery, Species	Years Observed	Observer Coverage	Est. SI by Year (CV)	Est. Mortality by Year (CV)	Mean Annual Mortality (CV)	PBR
CATEGORY I						
Gillnet Fisheries: Northeast Gillnet						
Harbor Porpoise	2015-2019	.14, .10, .12, .11, .12	0, 0, 7, 0, 0	177 (.28), 125 (.34), 129 (.28), 92 (.52), 195(.23)	145 (.14)	851
Common Dolphin	2015-2019	.14, .10, .12, .11, .12	0, 0, 0, 0, 0	55 (.54), 80 (.38), 133 (.28), 93 (.45), 5 (.68)	73 (.19)	1,452
Risso’s Dolphin	2015-2019	.14, .10, .12, .11, .12	0, 0, 0, 0, 0	0, 0, 0, 0, 5 (.7)	1 (3.5)	303
Bottlenose Dolphin, Offshore	2015-2019	.14, .10, .12, .11, .12	0, 0, 0, 0, 0	0, 0, 8 (.92), 0, 0	2.0 (.46)	561
Harbor Seal	2015-2019	.14, .10, .12, .11, .12	0, 0, 0, 0, 0	474 (.17), 245 (.29), 298 (.18), 188 (.36), 316 (.15)	304 (.10)	2,006
Gray Seal	2015-2019	.14, .10, .12, .11, .12	0, 0, 0, 0, 0	1021 (.25), 498 (.33), 930 (.16), 1113 (.32), 2019 (.17)	1116 (.11)	1,389
Harp Seal	2015-2019	.14, .10, .12, .11, .12	0, 0, 0, 0, 0	119 (.34), 85 (.50), 44 (.37), 14 (.8), 163 (.19)	85 (.16)	unk
Gillnet Fisheries: US Mid-Atlantic Gillnet						
Harbor Porpoise	2015-2019	.06, .08, .09, .09, .13	0, 0, 0, 0, 0	33 (1.16), 23 (.64), 9 (.95), 0, 13 (.51)	16 (.68)	851
Common Dolphin	2015-2019	.06, .08, .09, .09, .13	0, 0, 11, 0, 0	30 (.55), 7 (.97), 11 (.71), 8 (.91), 20 (.56)	17 (.31)	1,452
Harp Seal	2015-2019	.06, .08, .09, .09, .13	0, 0, 0, 0, 0	0, 0, 0, 0, 29 (.84)	6 (4.2)	unk
Harbor Seal	2015-2019	.06, .08, .09, .09, .13	0, 0, 0, 0, 0	48 (.52), 18 (.95), 3 (.18), 26 (.52), 17 (.35)	22 (.30)	2,006
Gray Seal	2015-2019	.06, .08, .09, .09, .13	0, 0, 0, 0, 0	15 (1.04), 7 (.93), 0, 0, 18 (.40)	8.0 (76)	1,389
Minke Whale	2015-2019	.06, .08, .09, .09, .13	0, 0, 0, 0, 0	0, 1, 0, 0, 0	0.2	14
Longline Fisheries: Pelagic Longline (Excluding NED-E)						
Risso's Dolphin	2015-2019	.12, .15, .12, .10, .10	8.4 (.71), 10.5 (.69), 0.2 (1), 0.2 (.94), 0	0, 5.6 (1), 0, 0, 0	5.0 (.44)	303

Category, Fishery, Species	Years Observed	Observer Coverage	Est. SI by Year (CV)	Est. Mortality by Year (CV)	Mean Annual Mortality (CV)	PBR
Short-finned Pilot Whale	2015-2019	.12, .15, .12, .10, .10	200 (.24), 106 (.31), 133 (.29), 102 (.39), 131 (.37)	0, 5.1 (1.9), 0, 0, 0	136 (.14)	236
Long-finned Pilot Whale	2015-2019	.12, .15, .12, .10, .10	2.2 (.49), 1.1 (1), 3.3 (.98), 0.4 (.93), 0.4 (1)	0, 0, 0, 0, 0	1.5 (.49)	306
Common Dolphin	2015-2019	.12, .15, .12, .10. .10	9.1 (1), 0, 4.9 (1), 1.4 (1), 0	0, 0, 0, 0, 0	3.1 (.67)	1,452
CATEGORY II						
Trawl Fisheries: Northeast Bottom Trawl						
Harp Seal	2015-2019	.19, .12, .16, .12, 16	0, 0, 0, 0, 0	0, 0, 0, 0, 5.4 (.89)	1.1 (.89)	unk
Harbor Seal	2015-2019	.19, .12, .16, .12, 16	0, 0, 0, 0, 0	0, 0, 8.3 (.96), 0, 5.4 (.88)	2.7 (.68)	2,006
Gray Seal	2015-2019	.19, .12, .16, .12, 16	0, 0, 0, 0, 0	23 (.46), 0, 16 (.24), 32 (.42), 30 (.37)	20 (.23)	1,389
Risso's Dolphin	2015-2019	.19, .12, .16, .12, 16	0, 0, 0, 0, 0	0, 17 (.88), 0, 0, 0	3.4 (.88)	303
Bottlenose Dolphin, Offshore	2015-2019	.19, .12, .16, .12, 16	0, 0, 0, 0, 0	19 (.65), 34 (.89), 0, 0, 5.6 (.92)	11.5 (.56)	519
Long-finned Pilot Whale	2015-2019	.19, .12, .16, .12, 16	0, 6, 0, 0, 0	0, 29 (.58), 0, 0, 5.4 (.88)	6.9 (.51)	306
Common Dolphin	2015-2019	.19, .12, .16, .12, 16	0, 0, 0, 0, 0	22(.45), 16(.46), 0, 28(.54), 10 (.62)	15 (.27)	1,452
Atlantic White-sided Dolphin	2015-2019	.19, .12, .16, .12, 16	0, 0, 0, 0, 7.4	15 (.52), 28 (.46), 15(.64), 0, 79 (.28)	27 (.21)	544
Harbor Porpoise	2015-2019	.19, .12, .16, .12, 16	0, 0, 0, 0, 0	0, 0, 0, 0, 11 (.63)	2.2 (.63)	851
Mid-Atlantic Bottom Trawl						
Common Dolphin	2015-2019	.09, .10, .10, .12, .12	0, 0, 0, 5, 15	250 (.32), 177 (.33), 380 (.23), 200 (.54), 395 (.23)	281 (.12)	1,452
Risso's Dolphin	2015-2019	.09, .10, .10, .12, .12	0, 0, 27, 0, 12	40(.63), 39 (.56), 43 (.51), 0, 0	24 (.33)	303
Bottlenose Dolphin, Offshore	2013-2017	.06, .08, .09, .10, .10	0, 0, 0, 0, 0	0, 7.3 (.93), 22 (.66), 6.3 (.91), 0	7.2 (.48)	561
Harbor Seal	2015-2019	.09, .10, .10, .12, .12	0, 0, 0, 0, 0	7, 0, 0, 6 (.94), 7.3 (.93)	4.1 (0.56)	2,006

Category, Fishery, Species	Years Observed	Observer Coverage	Est. SI by Year (CV)	Est. Mortality by Year (CV)	Mean Annual Mortality (CV)	PBR
Gray Seal	2015-2019	.09, .10, .10, .12, .12	0, 0, 0, 0, 0	0, 26 (.57), 26 (.40), 56 (.58), 22 (.53)	26 (.30)	1,389
Northeast Mid-water Trawl (Including Pair Trawl)						
Long-finned Pilot Whale	2015-2019	.08, .27, .16, .14, .28	0, 0, 0, 0, 0	0, .6 (na), 0, 0, 0	0.6 (na)	306
Harbor Seal	2015-2019	.08, .27, .16, .14, .28	0, 0, 0, 0, 0	.4 (na), .2 (na), 0, 0, 0	0.6 (na)	2,006
Gray Seal	2015-2019	.08, .27, .16, .14, .28	0, 0, 0, 0, 0	0, 0, 0, .2 (na), 0	0.2 (na)	1,389

Appendix II: Summary of the confirmed anecdotal human-caused mortality and serious injury (M/SI) events involving baleen whale stocks along the Gulf of Mexico Coast, U.S. East Coast, and adjacent Canadian Maritimes, 2015–2019, with number of events attributed to entanglements or vessel collisions by year.

Stock	Mean Annual M/SI rate (PBR[1] for reference)	Entanglements Annual Rate (U.S. waters, Canadian waters, unknown first sighted in U.S., unknown first sighted in Canada)	Entanglements Confirmed Mortalities (2015, 2016, 2017, 2018, 2019)	Entanglements Injury Value Against PBR (2015, 2016, 2017, 2018, 2019)	Vessel Collisions Annual Rate (U.S. waters, Canadian waters, unknown first sighted in U.S., unknown first sighted in Canada)	Vessel Collisions Confirmed Mortalities (2015, 2016, 2017, 2018, 2019)	Vessel Collisions Injury Value Against PBR (2015, 2016, 2017, 2018, 2019)
Western North Atlantic Right Whale ***(Eubalaena glacialis)***	7.65 (0.8)	5.65 (0.00/ 1.95/ 2.65/ 1.05)	(0, 2, 4, 3, 1)	(3.5, 7.5, 2, 4.25, 1)	2.00 (0.40/ 1.60/ 0.00/ 0.00)	(0, 1, 5, 0, 4)	(0)
Gulf of Maine Humpback Whale ***(Megaptera novaeangliae)***[2]	16.25 (22)	9.35 (2.50/ 0.60/ 5.75/ 0.50)	(1, 3, 2, 3, 1)	(7.5, 8, 6, 9.25, 6)	6.90 (6.10/ 0.00/ 0.80/ 0.00)	(4, 5, 8, 7, 5)	(0, 2, 1, 2, 0.52)
Western North Atlantic Fin Whale ***(Balaenoptera physalus)***	1.85 (11)	1.45 (0.00/ 0.80/ 0.65/ 0.00)	(0, 0, 1, 1, 2)	(1, 2.25, 0, 0, 0)	0.40 (0.40/ 0.00/ 0.00/ 0.00)	(0, 0, 1, 1, 0)	0
Nova Scotian Sei Whale ***(B. borealis)***	0.6 (6.2)	0.4 (0, 0, 0.4, 0)	(0, 0, 0, 1, 0)	(0, 0, 1, 0, 0)	0.20 (0.20/ 0.00/ 0.00/ 0.00)	(0, 1, 0, 0, 0)	0
Canadian East Coast Minke Whale ***(B. acutorostrata)***	10.35 (170)	9.55 (2.95/ 3.20/ 2.35/ 1.05)	(7, 3, 12, 11, 3)	(2.5, 1.75, 1.5, 2.25, 3.75)	0.80 (0.60/ 0.20/ 0.00/ 0.00)	(1, 0, 2, 1, 0)	0

[1] Potential Biological Removal (PBR)
[2] Humpback SAR not updated in 2021– values reported here are published in Henry *et al* in press

Appendix III: Fishery Descriptions

This appendix is broken into two parts: Part A describes commercial fisheries that have documented interactions with marine mammals in the Atlantic Ocean; and Part B describes commercial fisheries that have documented interactions with marine mammals in the Gulf of Mexico. A complete list of all known fisheries for both oceanic regions, the List of Fisheries, is published in the *Federal Register* annually. Each part of this appendix contains three sections: (I) data sources used to document marine mammal mortality/entanglements and commercial fishing effort trip locations, (II) links to fishery descriptions for Category I, II and some category III fisheries that have documented interactions with marine mammals and their historical level of observer coverage, and (III) historical fishery descriptions.

Part A. Description of U.S. Atlantic Commercial Fisheries

I. Data Sources

Items 1–5 describe sources of marine mammal mortality, serious injury or entanglement data; items 6–9 describe the sources of commercial fishing effort data used to summarize different components of each fishery (i.e. active number of permit holders, total effort, temporal and spatial distribution) and generate maps depicting the location and amount of fishing effort.

1. Northeast Region Fisheries Observer Program (NEFOP)

In 1989, a Fisheries Observer Program was implemented in the Northeast Region (Maine–Rhode Island) to document incidental bycatch of marine mammals in the Northeast Region Multi-species Gillnet Fishery. In 1993, sampling was expanded to observe bycatch of marine mammals in Gillnet Fisheries in the Mid-Atlantic Region (New York–North Carolina). The Northeast Fisheries Observer Program (NEFOP) has since been expanded to sample multiple gear types in both the Northeast and Mid-Atlantic Regions for documenting and monitoring interactions of marine mammals, sea turtles and finfish bycatch attributed to commercial fishing operations. At-sea observers placed onboard commercial fishing vessels collect data on fishing operations, gear and vessel characteristics, kept and discarded catch composition, bycatch of protected species, animal biology, and habitat (NMFS-NEFSC 2020).

2. Southeast Region Fishery Observer Programs

Three Fishery Observer Programs are managed by the Southeast Fisheries Science Center (SEFSC) that observe commercial fishery activity in U.S. Atlantic waters. The Pelagic Longline Observer Program (POP) administers a mandatory observer program for the U.S. Atlantic Large Pelagics Longline Fishery. The program has been in place since 1992 and randomly allocates observer effort by eleven geographic fishing areas proportional to total reported effort in each area and quarter. Observer coverage levels are mandated under the Highly Migratory Species Fisheries Management Plan (HMS FMP, 50 CFR Part 635). The second program is the Shark Gillnet Observer Program that observes the Southeastern U.S. Atlantic Shark Gillnet Fishery. The Observer Program is mandated under the HMS FMP, the Atlantic Large Whale Take Reduction Plan (ALWTRP; 50 CFR Part 229.32), and the Biological Opinion under Section 7 of the Endangered Species Act. Observers are deployed on any active fishing vessel reporting shark drift gillnet effort. In 2005, this program also began to observe sink gillnet fishing for sharks along the southeastern U.S. coast. The observed fleet includes vessels with an active directed shark permit and fish with sink gillnet gear (Carlson and Bethea 2007). The third program is the Southeastern Shrimp Otter Trawl Fishery Observer Program. Prior to 2007, this was a voluntary program administered by SEFSC in cooperation with the Gulf and South Atlantic Fisheries Foundation. The program was funding and project dependent, therefore observer coverage is not necessarily randomly allocated across the fishery. In 2007, the observer program was expanded, and it became mandatory for fishing vessels to take an observer, if selected. The program now includes more systematic sampling of the fleet based upon reported landings and effort patterns. The total level of observer coverage for this program is approximately 1% of the total fishery effort. In each Observer Program, the observers record information on the total target species catch, the number and type of interactions with protected species (including both marine mammals and sea turtles), and biological information on species caught.

3. Regional Marine Mammal Stranding Networks

The Northeast and Southeast Region Stranding Networks are components of the Marine Mammal Health and Stranding Response Program (MMHSRP). The goals of the MMHSRP are to facilitate collection and dissemination of data, assess health trends in marine mammals, correlate health with other biological and environmental parameters, and coordinate effective responses to unusual mortality events (Becker *et al.* 1994). Since 1997, the Northeast Region Marine Mammal Stranding Network has been collecting and storing data on marine mammal strandings and entanglements that occur from Maine through Virginia. The Southeast Region Strandings Program is responsible for data collection and stranding response coordination along the Atlantic coast from North Carolina to Florida, along the U.S. Gulf of Mexico coast from Florida through Texas, and in the U.S. Virgin Islands and Puerto Rico. Prior to 1997, stranding and entanglement data were maintained by the New England Aquarium and the National Museum of Natural History, Washington, D.C. Volunteer participants, acting under a letter of agreement, collect data on stranded animals that include: species; event date and location; details of the event (i.e., signs of human interaction) and determination on cause of death; animal disposition; morphology; and biological samples. Collected data are reported to the appropriate Regional Stranding Network Coordinator and are maintained in regional and national databases.

4. Marine Mammal Authorization Program

Commercial fishing vessels engaging in Category I or II fisheries are automatically registered under the Marine Mammal Authorization Program (MMAP) in order to lawfully take a non-endangered/threatened marine mammal incidental to fishing operations. These fishermen are required to carry an Authorization Certificate onboard while participating in the listed fishery, must be prepared to carry a fisheries observer if selected, and must comply with all applicable take reduction plan regulations. All vessel owners, regardless of the category of fishery they are operating in, are required to report, within 48 hours of the incident and even if an observer has recorded the take, all incidental injuries and mortalities of marine mammals that have occurred as a result of fishing operations (NMFS-OPR 2019). Events are reported by fishermen on the Marine Mammal Mortality/Injury forms then submitted to and maintained by the NMFS Office of Protected Resources. The data reported include: captain and vessel demographics; gear type and target species; date, time and location of event; type of interaction; animal species; mortality or injury code; and number of interactions. Reporting can be done online at:
https://docs.google.com/a/noaa.gov/forms/d/e/1FAIpQLSfKe0moEVK24x1Jbly33A0MRAa2ljZgmAcCVO1hEXghtB3SYA/viewform

5. Other Data Sources for Protected Species Interactions/Entanglements/Ship Strikes

In addition to the above, data on fishery interactions/entanglements and vessel collisions with large cetaceans are reported from a variety of other sources including the New England Aquarium (Boston, Massachusetts); Provincetown Center for Coastal Studies (Provincetown, Massachusetts); U.S. Coast Guard; whale watch vessels; Canadian Department of Fisheries and Oceans (DFO); and members of the Atlantic Large Whale Disentanglement Network. These data, photographs, etc. are maintained by the Protected Species Division at the Greater Atlantic Regional Fisheries Office (GARFO), the Protected Species Branch at the Northeast Fisheries Science Center (NEFSC) and the Southeast Fisheries Science Center (SEFSC).

6. Northeast Region Vessel Trip Reports

The Northeast Region Vessel Trip Report Data Collection System is a mandatory, but self-reported, commercial fishing effort database (Wigley *et al.* 1998). The data collected include: species kept and discarded, gear types used, trip location, trip departure and landing dates, port, and vessel and gear characteristics. The reporting of these data is mandatory only for vessels fishing under a federal permit. Vessels fishing under a federal permit are required to report in the Vessel Trip Report even when they are fishing within state waters.

7. Southeast Region Fisheries Logbook System

The Fisheries Logbook System (FLS) is maintained at the SEFSC and manages data submitted from mandatory Fishing Vessel Logbook Programs under several FMPs. In 1986, a comprehensive logbook program was initiated for the Large Pelagics Longline Fishery and this reporting became mandatory in 1992. Logbook reporting has also been initiated since the 1990s for a number of other fisheries including: Reef Fish Fisheries, Snapper-Grouper Complex Fisheries, federally managed Shark Fisheries, and King and Spanish Mackerel Fisheries. In each case, vessel captains are required to submit information on the fishing location, the amount and type of fishing gear used, the total amount of fishing effort (e.g., gear sets) during a given trip, the total weight and composition of the catch, and the disposition of the catch during each unit of effort (e.g., kept, released alive, released dead). FLS data are used to estimate the total amount of fishing effort in the fishery and thus expand bycatch rate estimates from observer data to estimates of the total incidental take of marine mammal species in a given fishery. More information is available at: https://www.fisheries.noaa.gov/southeast/resources-fishing/southeast-fisheries-permits

8. Northeast Region Dealer Reported Data

The Northeast Region Dealer Database houses trip level fishery statistics on fish species landed by market category, vessel ID, permit number, port location and date of landing, and gear type utilized. The data are collected by both federally permitted seafood dealers and NMFS port agents. Data are considered to represent a census of both vessels actively fishing with a federal permit and total fish landings. It also includes vessels that fish with a state permit (excluding the state of North Carolina) that land a federally managed species. Some states submit the same trip level data to the Northeast Region, but contrary to the data submitted by federally permitted seafood dealers, the trip level data reported by individual states does not include unique vessel and permit information. Therefore, the estimated number of active permit holders reported within this appendix should be considered a minimum estimate. It is important to note that dealers were previously required to report weekly in a dealer call-in system. However, in recent years the NER regional dealer reporting system has instituted a daily electronic reporting system. Although the initial reports generated from this new system did experience some initial reporting problems, these problems have been addressed and the new daily electronic reporting system is providing better real time information to managers.

9. Northeast At-Sea Monitoring Program

At-sea monitors collect scientific, management, compliance, and other fisheries data onboard commercial fishing vessels through interviews of vessel captains and crew, observations of fishing operations, photographing catch, and measurements of selected portions of the catch and fishing gear. At-sea monitoring requirements are detailed under Amendment 16 to the NE Multispecies Fishery Management Plan with a planned implementation date of May 1st, 2010. At-sea monitoring coverage is an integral part of catch monitoring to ensure that Annual Catch Limits are not exceeded. At-sea monitors collect accurate information on catch composition and the data are used to estimate total discards by sectors (and common pool), gear type, and stock area. Coverage levels are expected around

30%.

II. Marine Mammal Protection Act's List of Fisheries

The List of Fisheries (LOF) classifies U.S. commercial fisheries into one of three Categories according to the level of incidental mortality or serious injury of marine mammals:

Category I: Frequent incidental mortality or serious injury of marine mammals

Category II: Occasional incidental mortality or serious injury of marine mammals

Category III: Remote likelihood of/no known incidental mortality or serious injury of marine mammals

The Marine Mammal Protection Act (MMPA) mandates that each fishery be classified by the level of mortality or serious injury and mortality of marine mammals that occurs incidental to each fishery as reported in the annual Marine Mammal Stock Assessment Reports for each stock. A fishery may qualify as one Category for one marine mammal stock and another Category for a different marine mammal stock. A fishery is typically categorized on the LOF according to its highest level of classification (e.g., a fishery that qualifies for Category III for one marine mammal stock and Category II for another marine mammal stock will be listed under Category II). The fisheries listed below are linked to classification based on the most current LOF published in the *Federal Register*.

III. U.S Atlantic Commercial Fisheries

Please see the List of Fisheries for more information on the following fisheries: Northeast Sink Gillnet, Northeast Anchored Float Gillnet Fishery, Northeast Drift Gillnet Fishery, Mid-Atlantic Gillnet, Mid-Atlantic Bottom Trawl, Northeast Bottom Trawl, Northeast Mid-Water Trawl Fishery (includes pair trawls), Mid-Atlantic Mid-Water Trawl Fishery (includes pair trawls), Bay of Fundy Herring Weir, Gulf of Maine Atlantic Herring Purse Seine Fishery, Northeast/Mid-Atlantic American Lobster Trap/Pot, Atlantic Mixed Species Trap/Pot Fishery, Atlantic Ocean/Caribbean/Gulf of Mexico Large Pelagics Longline, Southeast Atlantic Gillnet, Southeastern U.S. Atlantic Shark Gillnet Fishery, Atlantic Blue Crab Trap/Pot, Mid-Atlantic Haul/Beach Seine, North Carolina Inshore Gillnet Fishery, North Carolina Long Haul Seine, North Carolina Roe Mullet Stop Net, Virginia Pound Net, Mid-Atlantic Menhaden Purse Seine, Southeastern U.S. Atlantic/Gulf of Mexico Shrimp Trawl, and Southeastern U.S. Atlantic/Gulf of Mexico Stone Crab Trap/Pot Fishery.

IV. Historical Fishery Descriptions

Atlantic Foreign Mackerel

Prior to 1977, there was no documentation of marine mammal bycatch in Distant-Water Fishing (DWF) activities off the Northeast coast of the U.S. In 1977, with implementation of the Magnuson Fisheries Conservation and Management Act (MFCMA), an Observer Program was established which recorded fishery data and information on incidental bycatch of marine mammals. DWF effort in the U.S. Atlantic Exclusive Economic Zone (EEZ) under MFCMA had been directed primarily towards Atlantic mackerel and squid. From 1977 through 1982, an average mean of 120 different foreign vessels per year (range 102–161) operated within the U.S. Atlantic EEZ. In 1982, there were 112 different foreign vessels; 16%, or 18 vessels, were Japanese tuna longline vessels operating along the U.S. east coast. This was the first year that the Northeast Regional Observer Program assumed responsibility for observer coverage of the longline vessels. Between 1983 and 1991, the numbers of foreign vessels operating within the U.S. Atlantic EEZ each year were 67, 52, 62, 33, 27, 26, 14, 13, and 9, respectively. Between 1983 and 1988, the numbers of DWF Japanese longline vessels included 3, 5, 7, 6, 8, and 8, respectively. Observer coverage on DWF vessels was 25-35% during 1977-1982, and increased to 58%, 86%, 95% and 98%, respectively, in 1983–1986. One hundred percent observer coverage was maintained during 1987–1991. Foreign fishing operations for squid ceased at the end of the 1986 fishing season and for mackerel at the end of the 1991 season. Documented interactions with white-sided dolphins were reported in this fishery.

Pelagic Drift Gillnet

In 1996 and 1997, NMFS issued management regulations which prohibited the operation of this fishery in 1997. The fishery operated during 1998. Then, in January 1999 NMFS issued a Final Rule to prohibit the use of drift net gear in the North Atlantic Swordfish Fishery (50 CFR Part 630). In 1986, NMFS established a mandatory self-reported fisheries information system for Large Pelagic Fisheries. Data files are maintained at the SEFSC. The estimated total number of hauls in the Atlantic Pelagic Drift Gillnet Fishery increased from 714 in 1989 to 1,144 in 1990; thereafter, with the introduction of quotas, effort was severely reduced. The estimated number of hauls from 1991 to 1996 was 233, 243, 232, 197, 164, and 149, respectively. Fifty-nine different vessels participated in this fishery at one time or another between 1989 and 1993. In 1994 to 1998 there were 11, 12, 10, 0, and 11 vessels, respectively, in the fishery. Observer coverage, expressed as percent of sets observed, was 8% in 1989, 6% in 1990, 20% in 1991, 40% in 1992, 42% in 1993, 87% in 1994, 99% in 1995, 64% in 1996, no fishery in 1997, and 99% coverage during 1998. Observer coverage dropped during 1996 because some vessels were deemed too small or unsafe by the contractor that provided observer coverage to NMFS. Fishing effort was concentrated along the southern edge of Georges Bank and off Cape Hatteras, North Carolina. Examination of the species composition of the catch and locations of the fishery throughout the year suggest that the Drift Gillnet Fishery was stratified into two strata: (1) a southern, or winter, stratum and (2) a northern, or summer, stratum. Documented interactions with North Atlantic right whales, humpback whales, sperm whales, pilot whale spp., *Mesoplodon* spp., Risso's dolphins, common dolphins, striped dolphins and white-sided dolphins were reported in this fishery.

Atlantic Tuna Purse Seine

The Tuna Purse Seine Fishery occurring between the Gulf of Maine and Cape Hatteras, North Carolina is directed at large medium and giant bluefin tuna (BFT). Spotter aircraft are typically used to locate fish schools. The official start date, set by regulation, is 15 July of each year. Individual Vessel Quotas (IVQs) and a limited access system prevent a derby fishery situation. Catch rates for large medium, and giant tuna can be high and consequently, the season can last only a few weeks, however, over the last number of years, effort expended by this sector of the BFT fishery has diminished dramatically due to the unavailability of BFT on the fishing grounds.

The regulations allocate approximately 18.6% of the U.S. BFT quota to this sector of the fishery (five IVQs) with a tolerance limit established for large medium BFT (15% by weight of the total amount of giant BFT landed).

Limited observer data is available for the Atlantic Tuna Purse Seine Fishery. Out of 45 total trips made in 1996, 43 trips (95.6%) were observed. Forty-four sets were made on the 43 observed trips and all sets were observed. A total of 136 days were covered. No trips were observed during 1997 through 1999. Two trips (seven hauls) were observed in October 2000 in the Great South Channel Region. Four trips were observed in September 2001. No marine mammals were observed taken during these trips. Documented interactions with pilot whale spp. were reported in this fishery.

Atlantic Tuna Pelagic Pair Trawl

The Pelagic Pair Trawl Fishery operated as an experimental fishery from 1991 to 1995, with an estimated 171 hauls in 1991, 536 in 1992, 586 in 1993, 407 in 1994, and 440 in 1995. This fishery ceased operations in 1996 when NMFS rejected a petition to consider pair trawl gear as an authorized gear type in the Atlantic Tuna Fishery. The fishery operated from August to November in 1991, from June to November in 1992, from June to October in 1993 (Northridge 1996), and from mid-summer to December in 1994 and 1995. Sea sampling began in October of 1992 (Gerrior *et al.* 1994) where 48 sets (9% of the total) were sampled. In 1993, 102 hauls (17% of the total) were sampled. In 1994 and 1995, 52% (212) and 55% (238), respectively, of the sets were observed. Nineteen vessels have operated in this fishery. The fishery operated in the area between 35°N to 41°N and 69°W to 72°W. Approximately 50% of the total effort was within a one degree square at 39°N, 72°W, around Hudson Canyon, from 1991 to 1993. Examination of the 1991–1993 locations and species composition of the bycatch, showed little seasonal change for the six months of operation and did not warrant any seasonal or areal stratification of this fishery (Northridge 1996). During the 1994 and 1995 Experimental Pelagic Pair Trawl Fishing Seasons, fishing gear experiments were conducted to collect data on environmental parameters, gear behavior, and gear handling practices to evaluate factors affecting catch and bycatch (Goudey 1995, 1996), but the results were inconclusive. Documented interactions with pilot whale spp., Risso's dolphin and common dolphins were reported in this fishery.

Part B. Description of U.S. Gulf of Mexico Fisheries

I. Data Sources

Items 1 and 2 describe sources of marine mammal mortality, serious injury or entanglement data, and item 3 describes the source of commercial fishing effort data used to generate maps depicting the location and amount of fishing effort and the numbers of active permit holders. In general, commercial fisheries in the Gulf of Mexico have had little directed observer coverage and the level of fishing effort for most fisheries that may interact with marine mammals is either not reported or highly uncertain.

1. Southeast Region Fishery Observer Programs

Two fishery observer programs are managed by the SEFSC that observe commercial fishery activity in the U.S. Gulf of Mexico. The Pelagic Longline Observer Program (POP) administers a mandatory observer program for the U.S. Atlantic Large Pelagics Longline Fishery. The program has been in place since 1992, and randomly allocates observer effort by eleven geographic fishing areas proportional to total reported effort in each area and quarter. Observer coverage levels are mandated under the Highly Migratory Species FMP (HMS FMP, 50 CFR Part 635). The second is the Southeastern Shrimp Otter Trawl Fishery Observer Program. Prior to 2007, this was a voluntary program administered by SEFSC in cooperation with the Gulf and South Atlantic Fisheries Foundation. The program was funding and project dependent, therefore observer coverage is not necessarily randomly allocated across the fishery. In 2007, the observer program was expanded, and it became mandatory for fishing vessels to take an observer if selected. The program now includes more systematic sampling of the fleet based upon reported landings and effort patterns. The total level of observer coverage for this program is ~1% of the total fishery effort. In each Observer Program, the observers record information on the total target species catch, the number and type of interactions with protected species (including both marine mammals and sea turtles), and biological information on species caught.

2. Regional Marine Mammal Stranding Networks

The Southeast Regional Stranding Network is a component of the Marine Mammal Health and Stranding Response Program (MMHSRP). The goals of the MMHSRP are to facilitate collection and dissemination of data, assess health trends in marine mammals, correlate health with other biological and environmental parameters, and coordinate effective responses to unusual mortality events (Becker *et al.* 1994). The Southeast Region Strandings Program is responsible for data collection and stranding response coordination along the U.S. Gulf of Mexico coast from Florida through Texas. Prior to 1997, stranding and entanglement data were maintained by the New England Aquarium and the National Museum of Natural History. Volunteer participants, acting under a letter of agreement with NOAA Fisheries, collect data on stranded animals that include: species, event date and location, details of the event including evidence of human interactions, determinations of the cause of death, animal disposition, morphology, and biological samples. Collected data are reported to the appropriate Regional Stranding Network Coordinator and are maintained in regional and national databases.

3. Southeast Region Fisheries Logbook System (FLS)

The FLS is maintained at the SEFSC and manages data submitted from mandatory fishing vessel logbook programs under several FMPs. In 1986, a comprehensive logbook program was initiated for the Large Pelagics Longline Fisheries, and this reporting became mandatory in 1992. Logbook reporting has also been initiated since the early 1990s for a number of other fisheries including: Reef Fish Fisheries, Snapper-Grouper Complex Fisheries, federally managed Shark Fisheries, and King and Spanish Mackerel Fisheries. In each case, vessel captains are required to submit information on the fishing location, the amount and type of fishing gear used, the total amount of fishing effort (e.g., gear sets) during a given trip, the total weight and composition of the catch, and the disposition of the catch during each unit of effort (e.g., kept, released alive, released dead). FLS data are used to estimate the total amount of fishing effort in the fishery and thus expand bycatch rate estimates from observer data to estimate the total incidental take of marine mammal species in a given fishery.

4. Marine Mammal Authorization Program

Commercial fishing vessels engaging in Category I or II fisheries are automatically registered under the Marine Mammal Authorization Program (MMAP) in order to lawfully take a non-endangered/threatened marine mammal incidental to fishing operations. These fishermen are required to carry an Authorization Certificate onboard while participating in the listed fishery, must be prepared to carry a fisheries observer if selected, and must comply with all applicable take reduction plan regulations. All vessel owners, regardless of the category of fishery they are operating in, are required to report within 48 hours of the incident, even if an observer has recorded the take, all incidental injuries and mortalities of marine mammals that have occurred as a result of fishing operations (NMFS-OPR 2019). Events are reported by fishermen on the Marine Mammal Mortality/Injury forms then submitted to and maintained by the NMFS Office of Protected Resources. The data reported include: captain and vessel demographics; gear type and target species; date, time and location of event; type of interaction; animal species; mortality or injury code; and number of interactions. Reporting can be done online at: https://docs.google.com/a/noaa.gov/forms/d/e/1FAIpQLSfKe0moEVK24x1Jbly33A0MRAa2ljZgmAcCVO1hEXghtB3SYA/viewform

II. Gulf of Mexico Commercial Fisheries

Please see the List of Fisheries for more information on the following fisheries: Spiny Lobster Trap/Pot Fishery, Southeastern U.S. Atlantic/Gulf of Mexico Stone Crab Trap/Pot Fishery, Gulf of Mexico Menhaden Purse Seine Fishery, Gulf of Mexico Gillnet Fishery.

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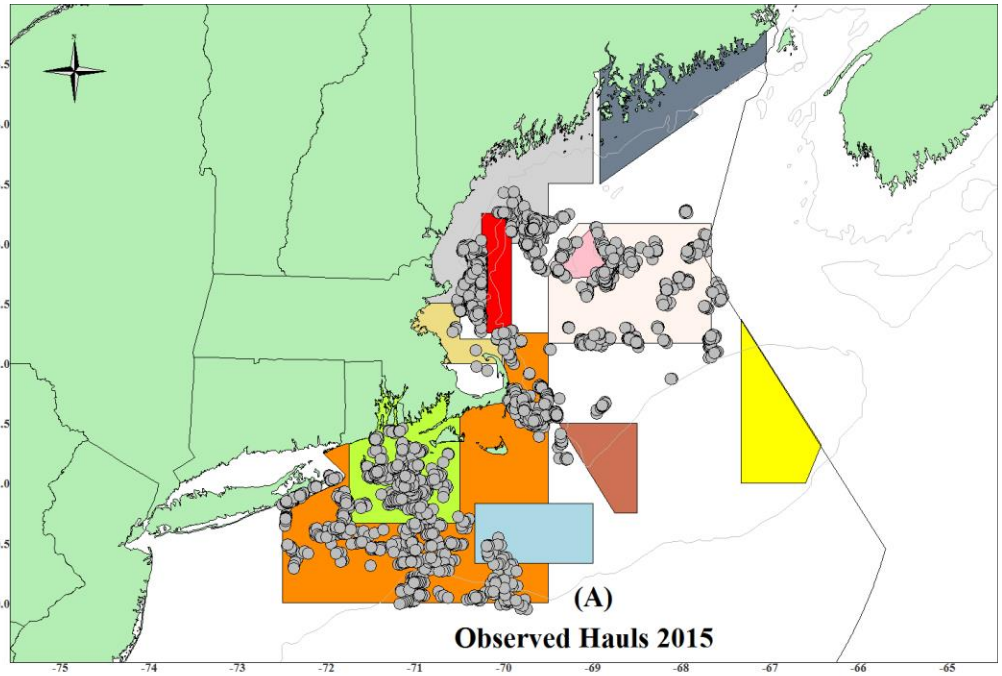
Appendix III: Fishery Descriptions - List of Figures

Figure 1. 2015 Northeast sink gillnet observed hauls (A) and observed takes (B).



Multispecies Fisheries Management Plan year-round closures:

Closed Area 1 | Closed Area 2 | Western Gulf of Maine Closed Area | Nantucket Lightship Closed Area | Cashes Ledge Closed Area

Harbor porpoise Take Reduction Plan management areas:

Offshore Closure | Northeast Closure | MidCoast Closure | Mass Bay Closure | Cod South Closure | Cashes Ledge Closed Area

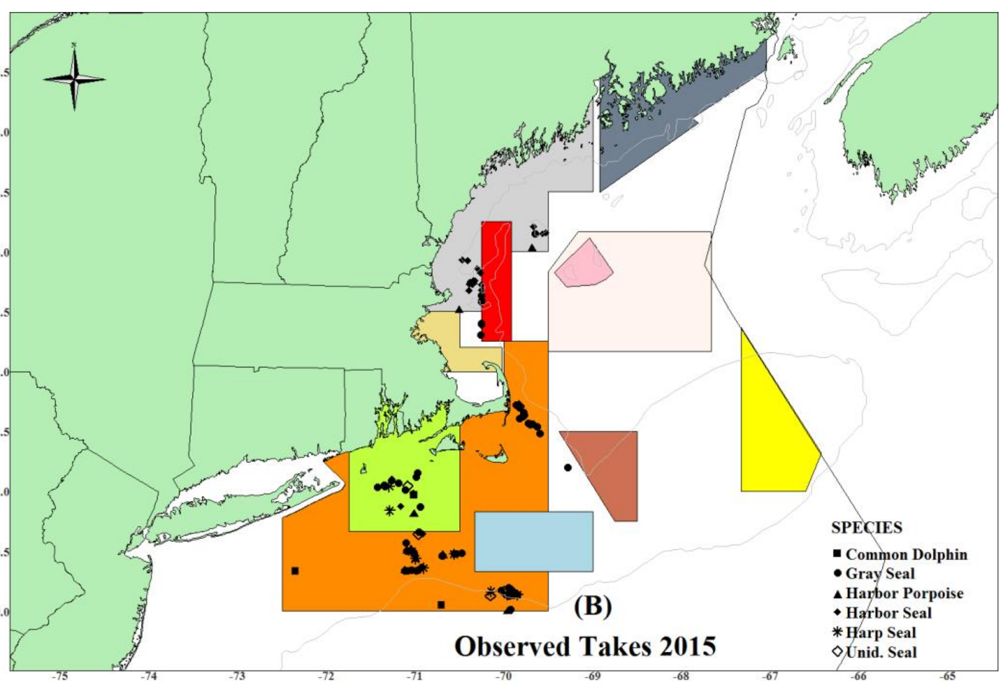
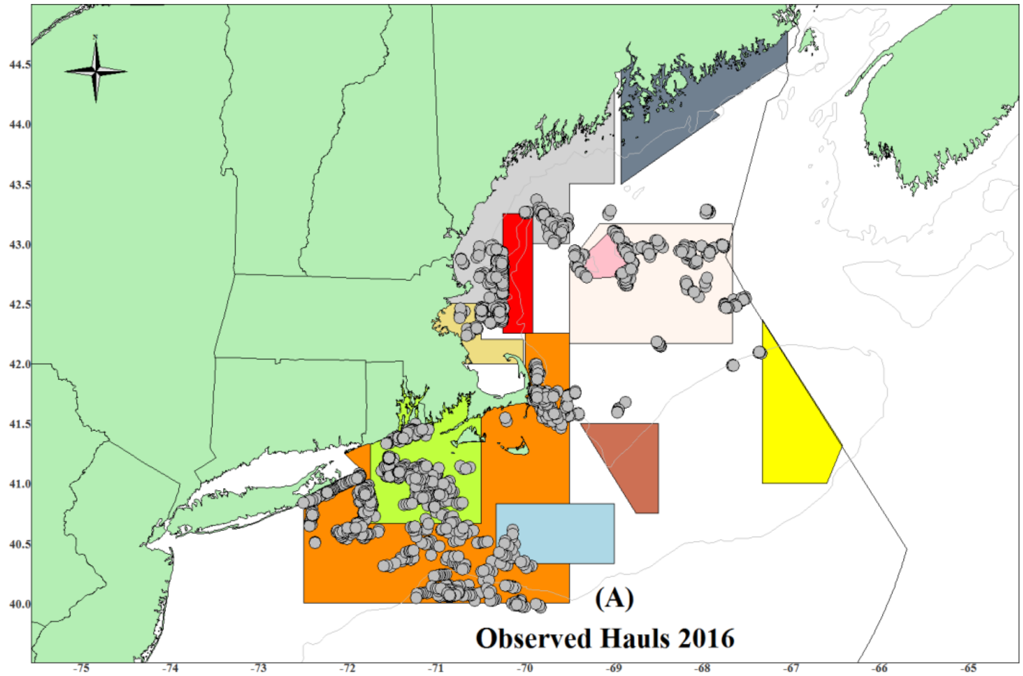


Figure 2. 2016 Northeast sink gillnet observed hauls (A) and observed takes (B).



Multispecies Fisheries Management Plan year-round closures:
Closed Area 1 | Closed Area 2 | Western Gulf of Maine Closed Area | Nantucket Lightship Closed Area | Cashes Ledge Closed Area

Harbor porpoise Take Reduction Plan management areas:
Offshore Closure | Northeast Closure | MidCoast Closure | Mass Bay Closure | Cod South Closure | Cashes Ledge Closed Area

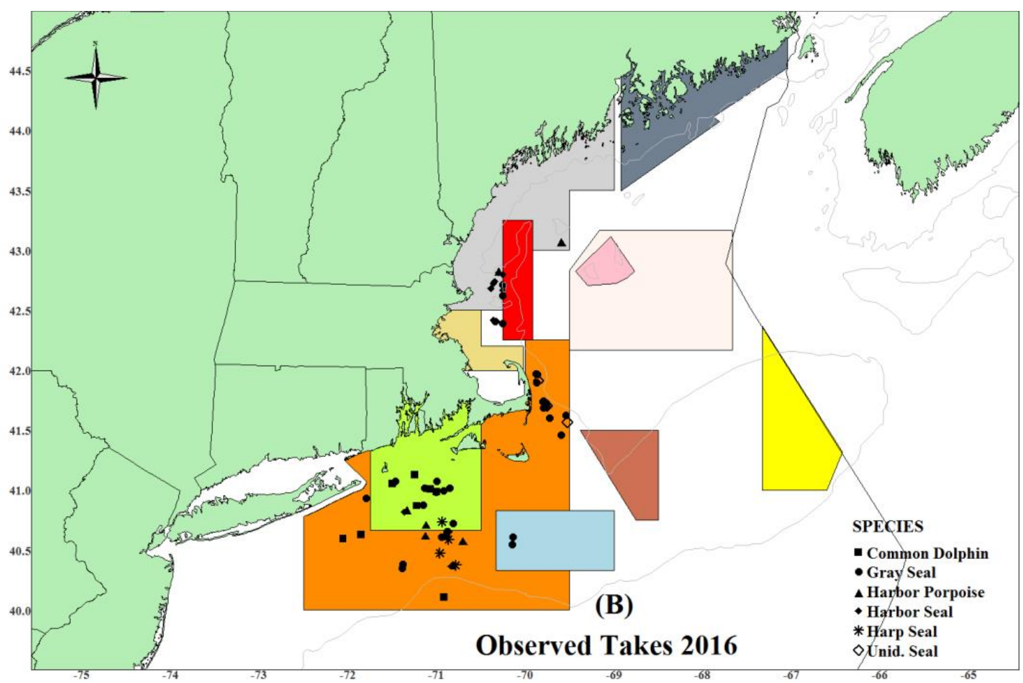
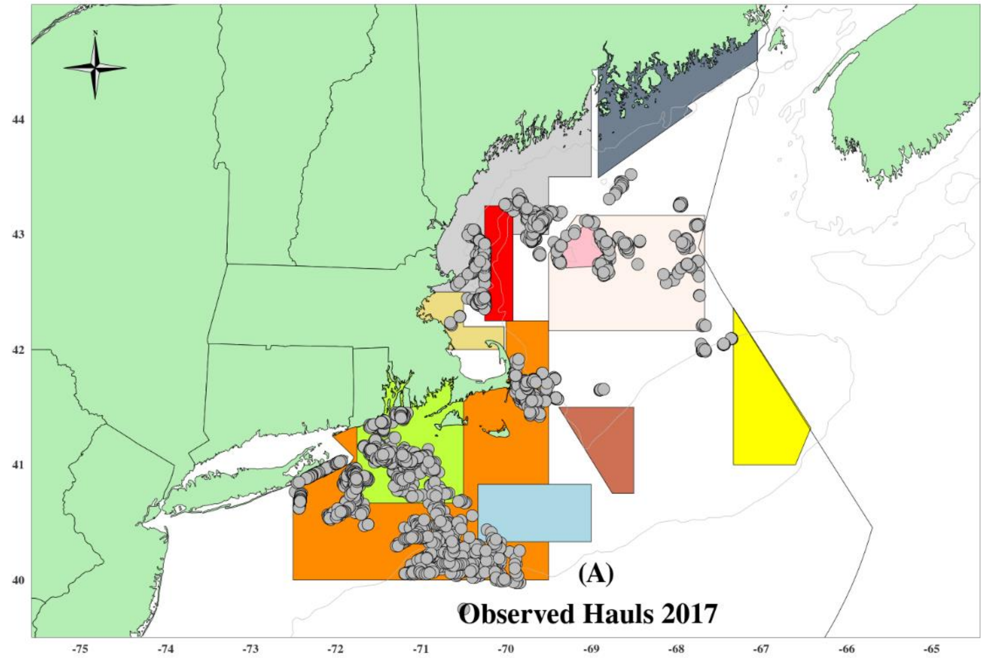


Figure 3. 2017 Northeast sink gillnet observed hauls (A) and observed takes (B).



Multispecies Fisheries Management Plan year-round closures:

Closed Area 1 | Closed Area 2 | Western Gulf of Maine Closed Area | Nantucket Lightship Closed Area | Cashes Ledge Closed Area

Harbor porpoise Take Reduction Plan management areas:

Offshore Closure | Northeast Closure | MidCoast Closure | Mass Bay Closure | Cod South Closure | Cashes Ledge Closed Area

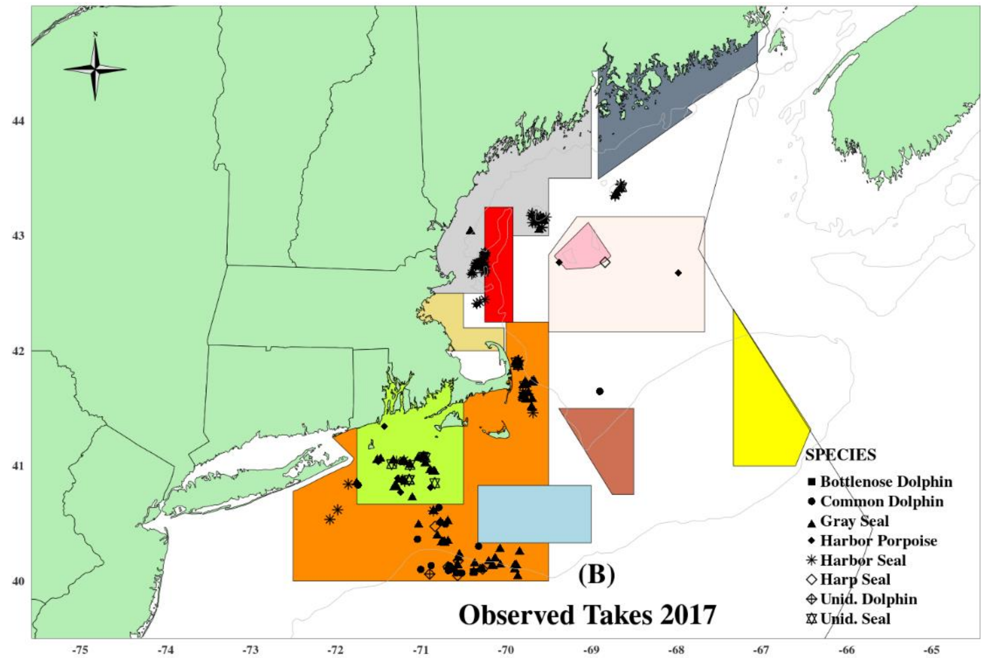
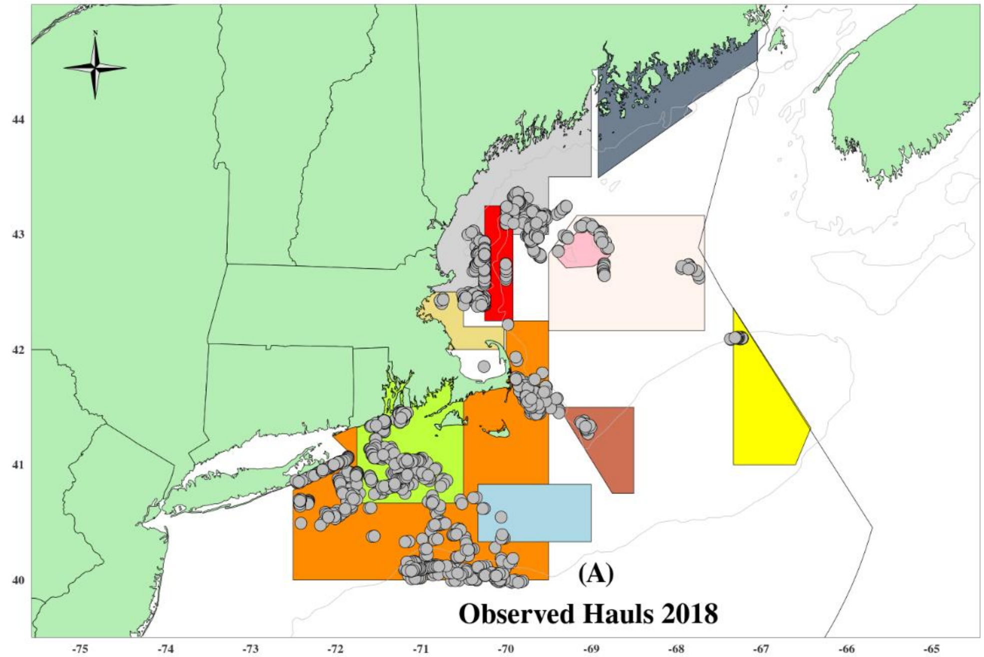


Figure 4. 2018 Northeast sink gillnet observed hauls (A) and observed takes (B).



Multispecies Fisheries Management Plan year-round closures:

Closed Area 1 | Closed Area 2 | Western Gulf of Maine Closed Area | Nantucket Lightship Closed Area | Cashes Ledge Closed Area

Harbor porpoise Take Reduction Plan management areas:

Offshore Closure | Northeast Closure | MidCoast Closure | Mass Bay Closure | Cod South Closure | Cashes Ledge Closed Area

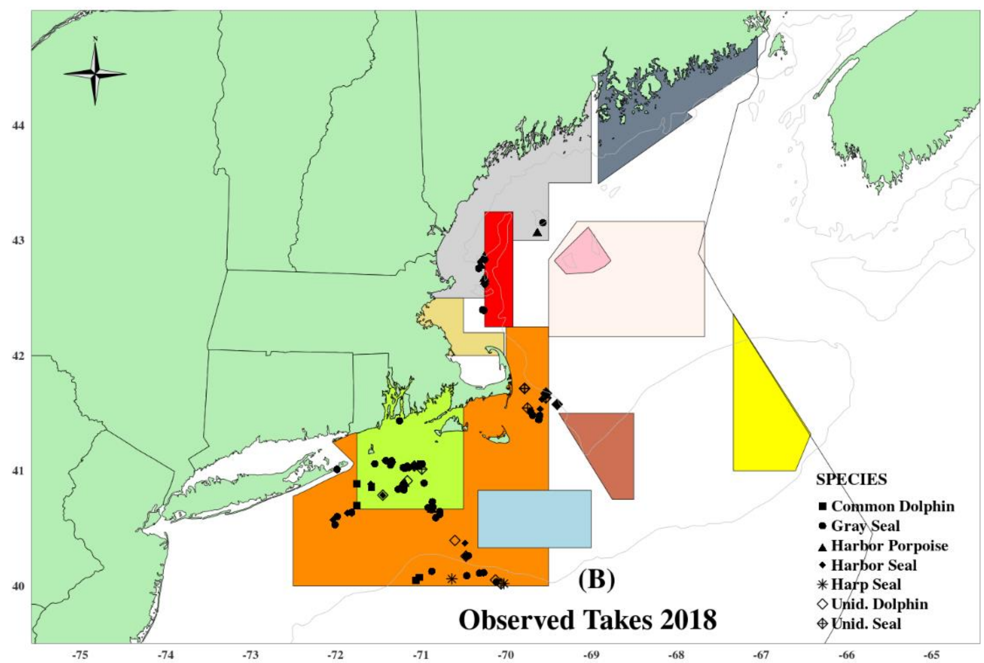
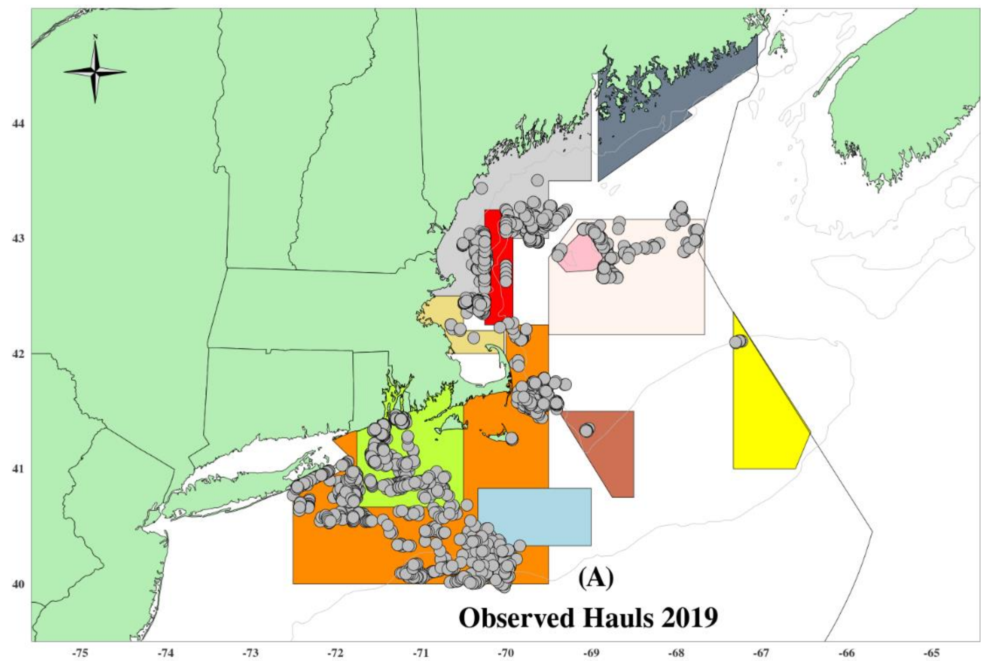


Figure 5. 2019 Northeast sink gillnet observed hauls (A) and observed takes (B).



Multispecies Fisheries Management Plan year-round closures:

Closed Area 1 | Closed Area 2 | Western Gulf of Maine Closed Area | Nantucket Lightship Closed Area | Cashes Ledge Closed Area

Harbor porpoise Take Reduction Plan management areas:

Offshore Closure | Northeast Closure | MidCoast Closure | Mass Bay Closure | Cod South Closure | Cashes Ledge Closed Area

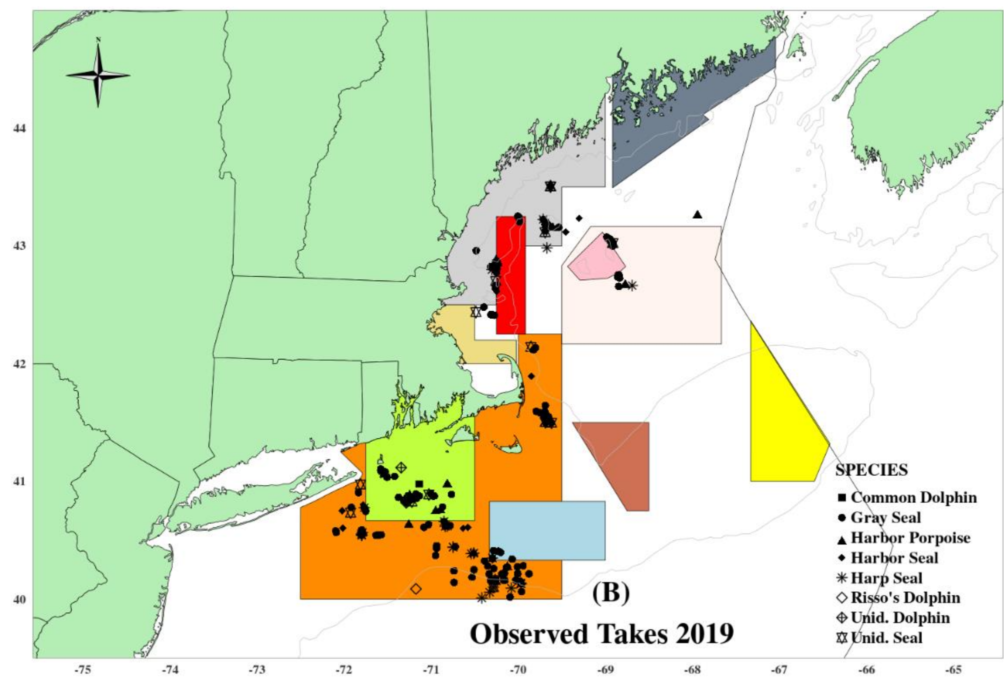
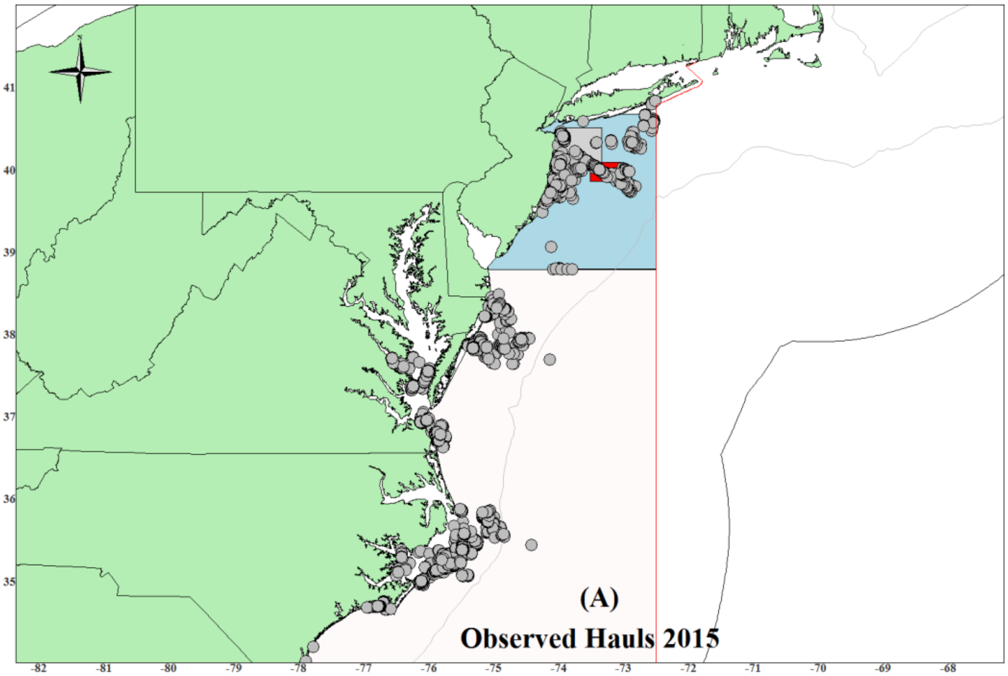


Figure 6. 2015 Mid-Atlantic gillnet observed hauls (A) and observed takes (B).



Harbor porpoise Take Reduction Plan management areas:

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

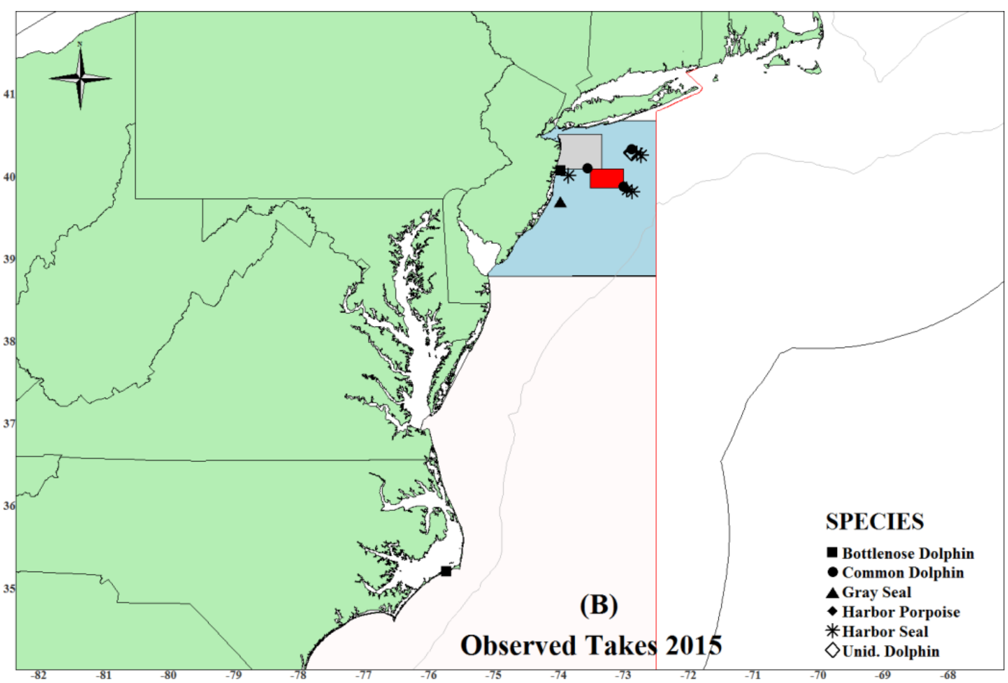
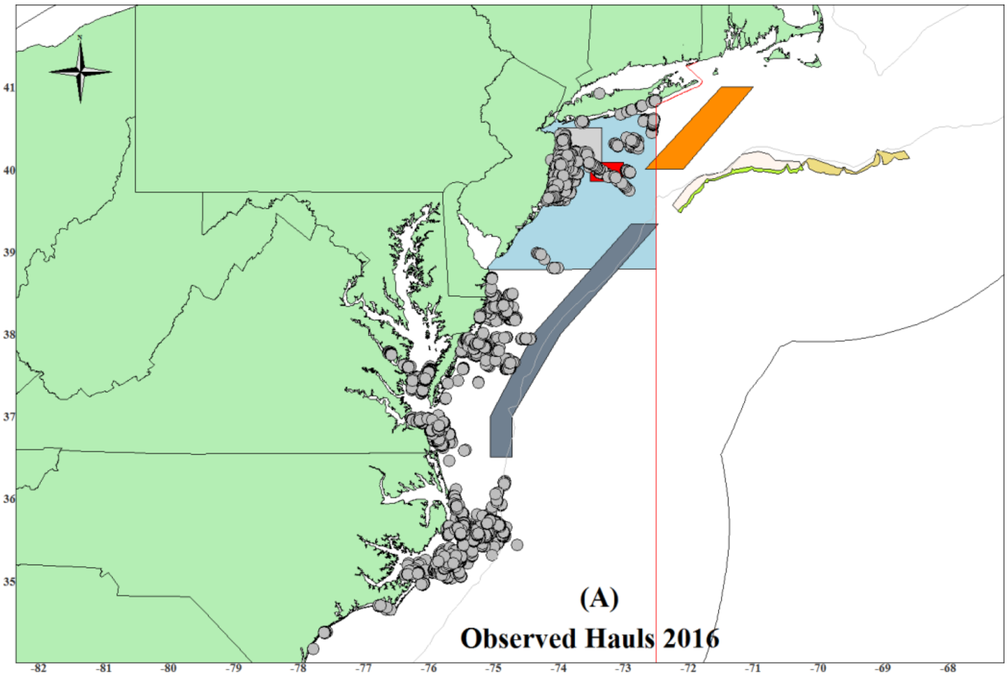


Figure 7. 2016 Mid-Atlantic gillnet observed hauls (A) and observed takes (B).



Harbor porpoise Take Reduction Plan management areas:
Southern mid-Atlantic waters | New Jersey Mudhole | Mudhole South | waters off New Jersey

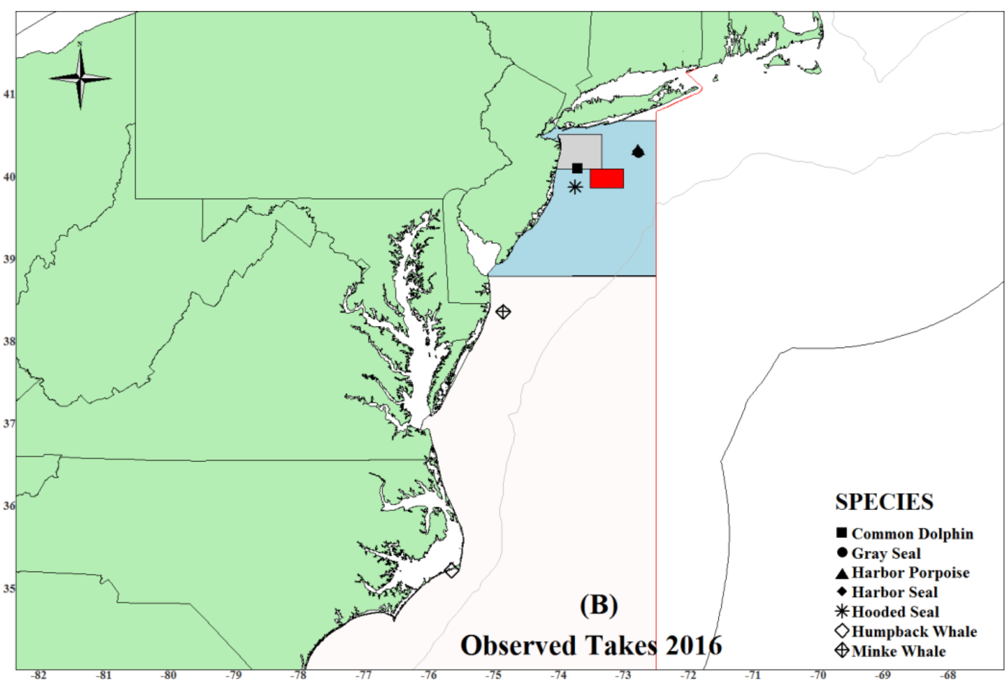
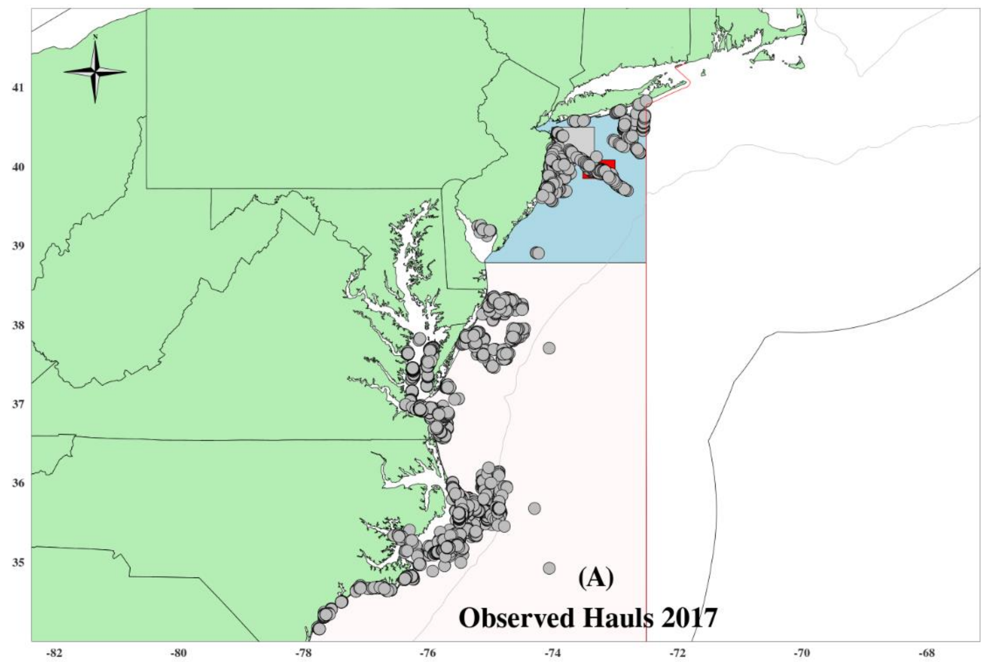


Figure 8. 2017 Mid-Atlantic gillnet observed hauls (A) and observed takes (B).



Harbor porpoise Take Reduction Plan management areas:
Southern mid-Atlantic waters | New Jersey Mudhole | Mudhole South | waters off New Jersey

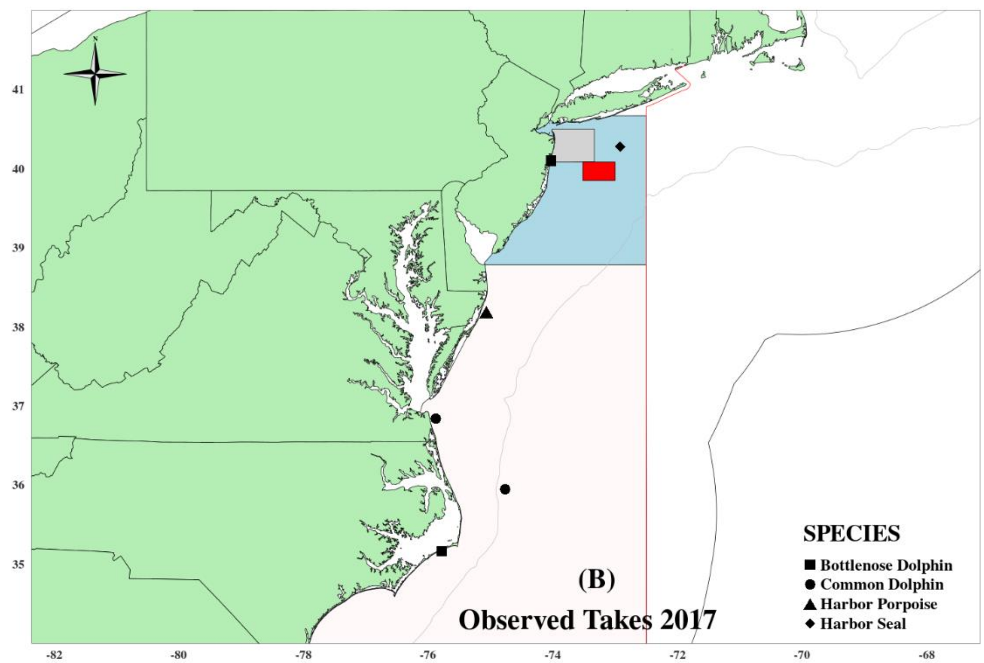
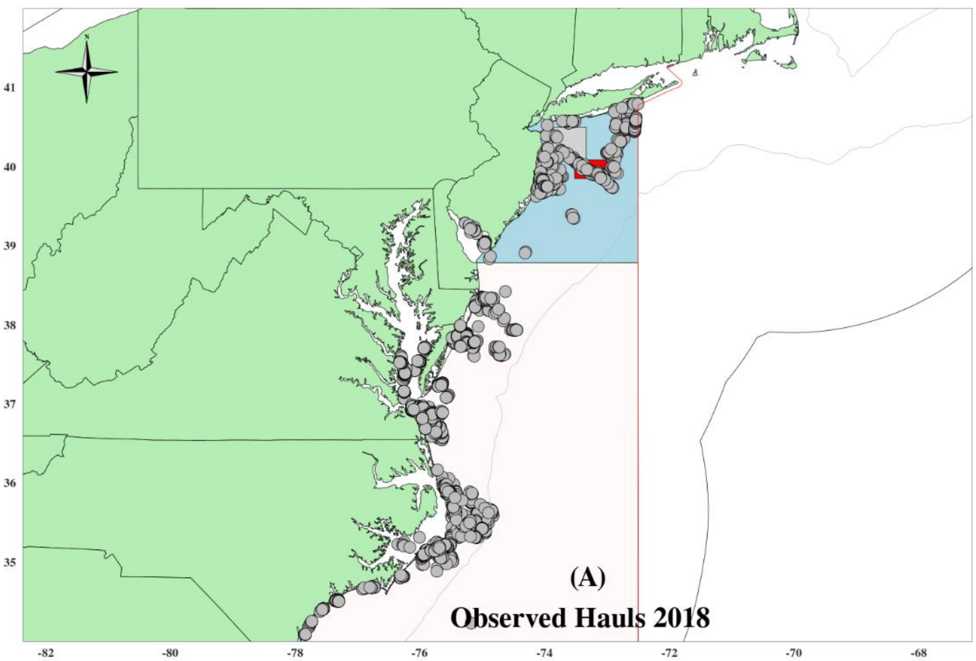


Figure 9. 2018 Mid-Atlantic gillnet observed hauls (A) and observed takes (B).



Harbor porpoise Take Reduction Plan management areas:

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

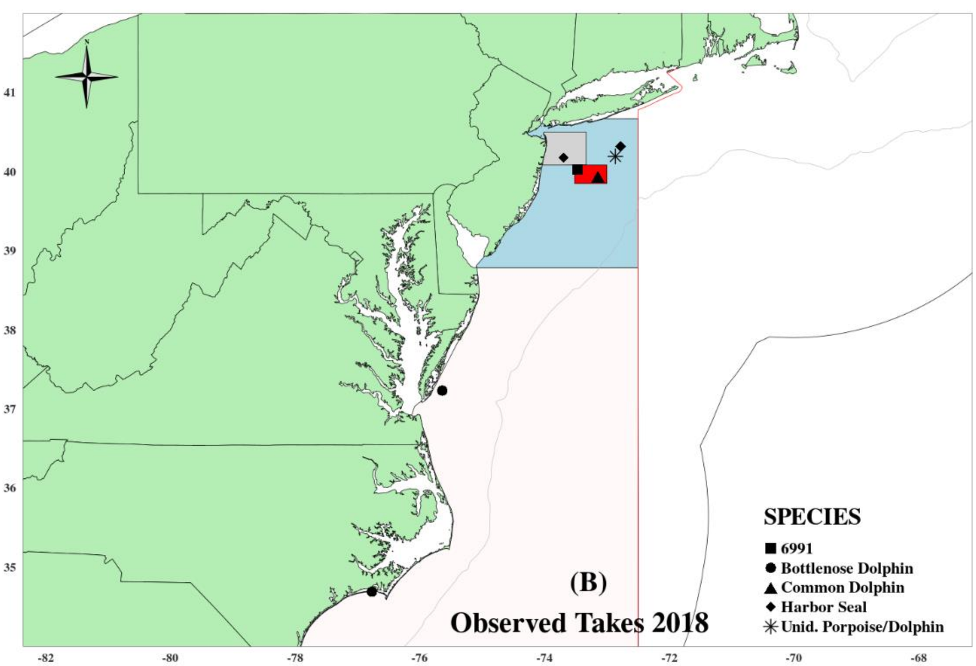
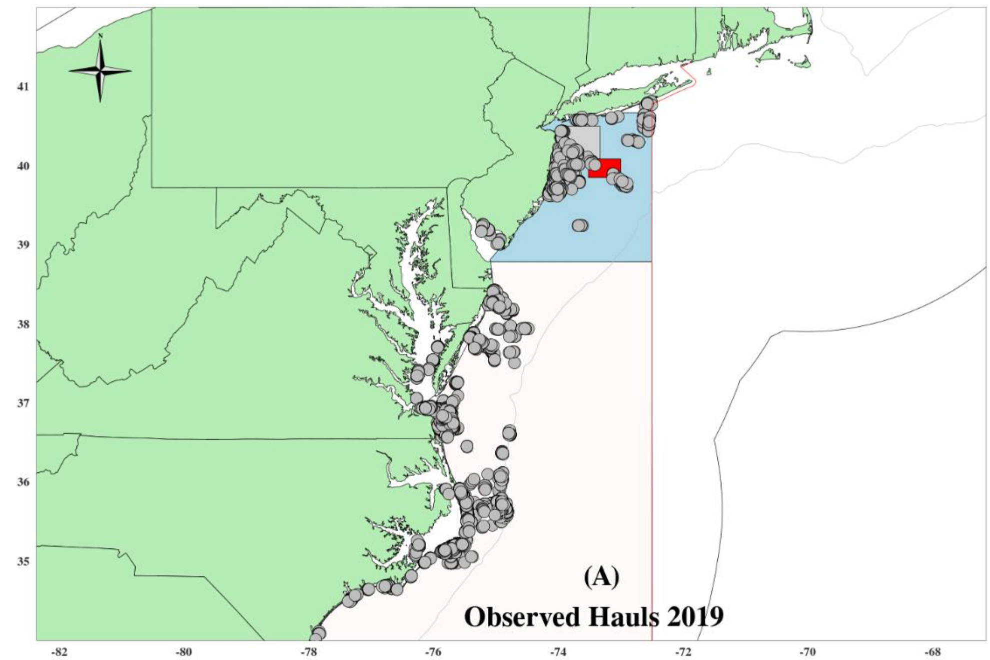


Figure 10. 2019 Mid-Atlantic gillnet observed hauls (A) and observed takes (B).



Harbor porpoise Take Reduction Plan management areas:
Southern mid-Atlantic waters | New Jersey Mudhole | Mudhole South | waters off New Jersey

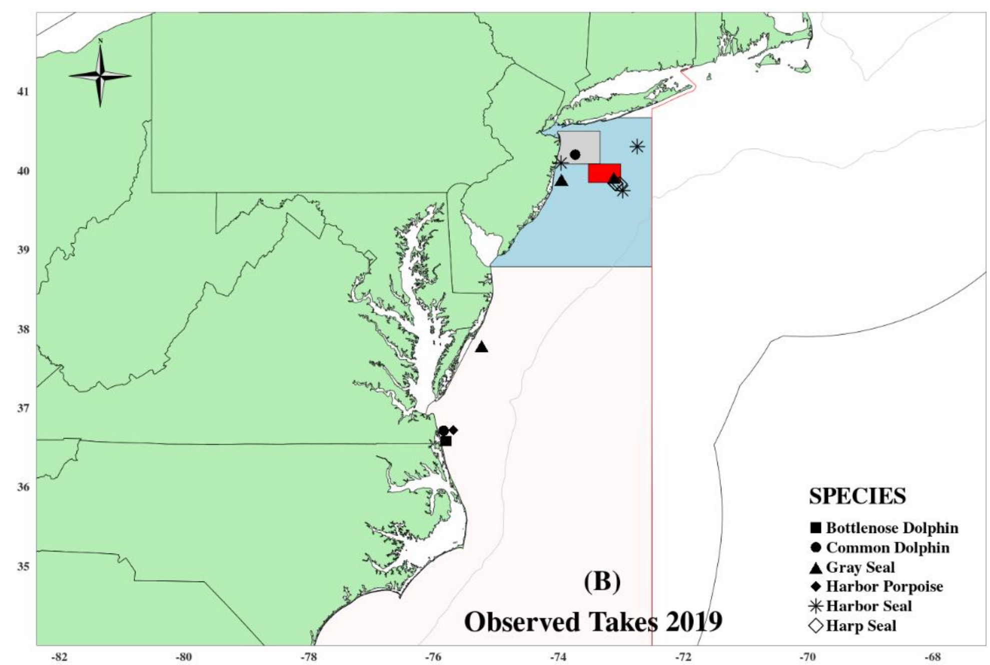


Figure 11. 2015 Mid-Atlantic bottom trawl observed tows (A) and observed takes (B).

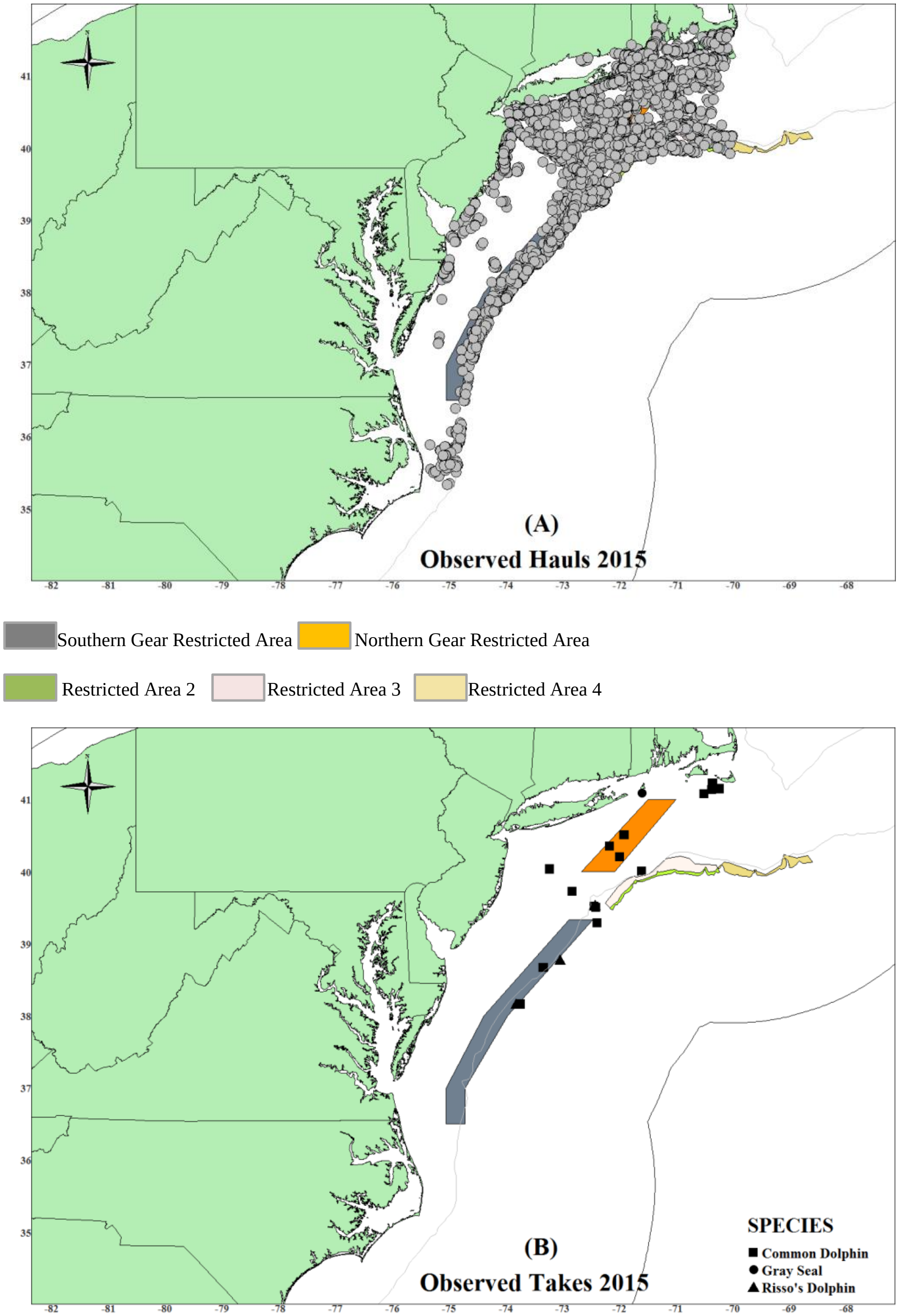


Figure 12. 2016 Mid-Atlantic bottom trawl observed tows (A) and observed takes (B).

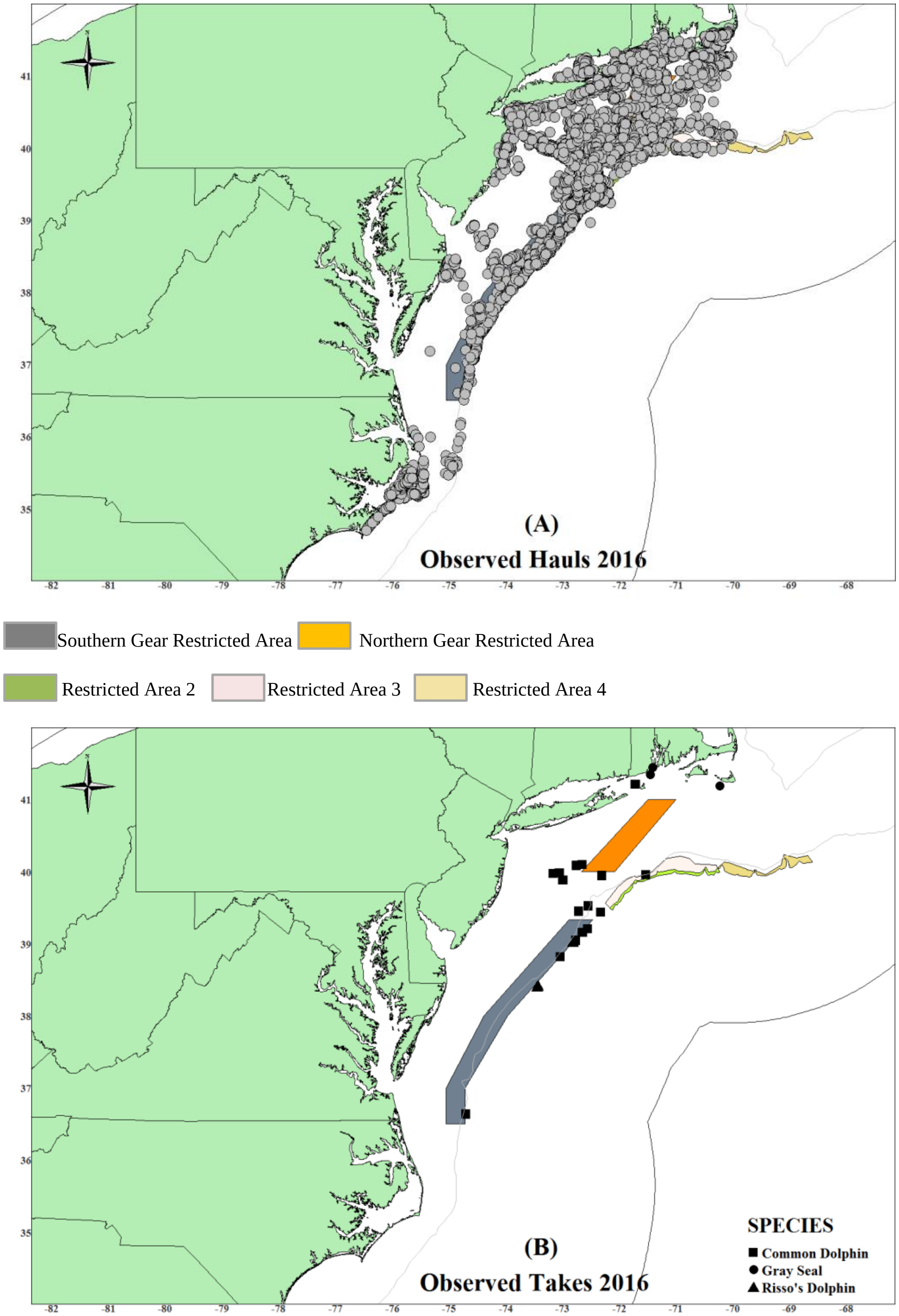
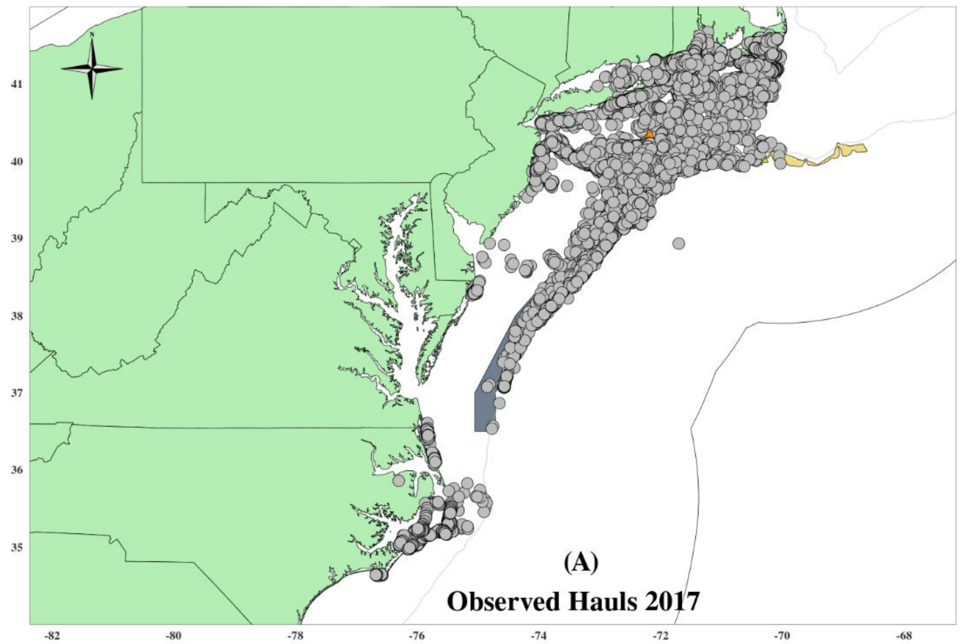


Figure 13. 2017 Mid-Atlantic bottom trawl observed tows (A) and observed takes (B).



Southern Gear Restricted Area Northern Gear Restricted Area

Restricted Area 2 Restricted Area 3 Restricted Area 4

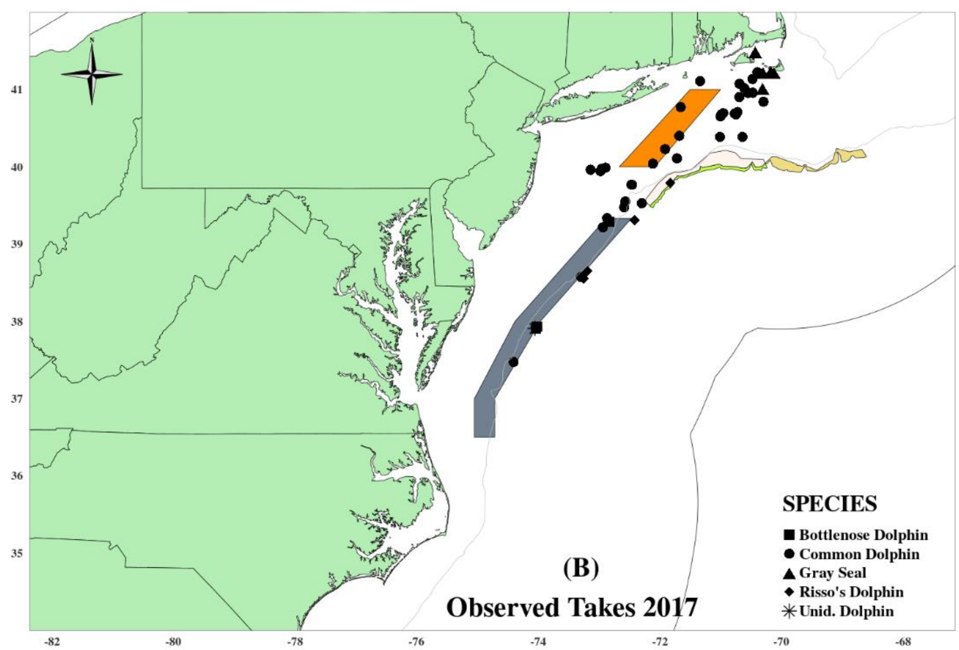
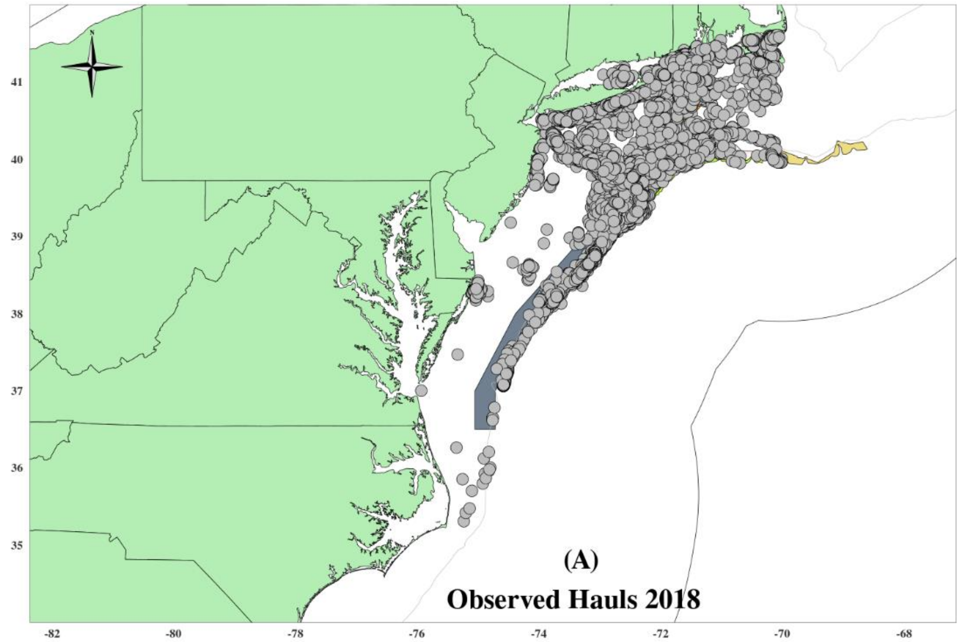


Figure 14. 2018 Mid-Atlantic bottom trawl observed tows (A) and observed takes (B).



Southern Gear Restricted Area Northern Gear Restricted Area

Restricted Area 2 Restricted Area 3 Restricted Area 4

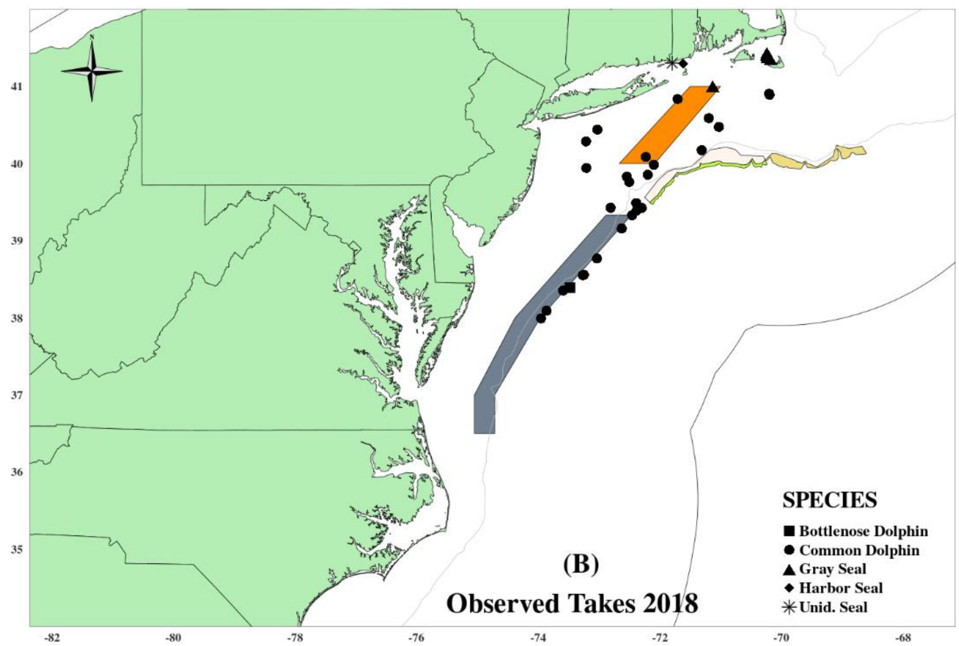


Figure 15. 2019 Mid-Atlantic bottom trawl observed tows (A) and observed takes (B).

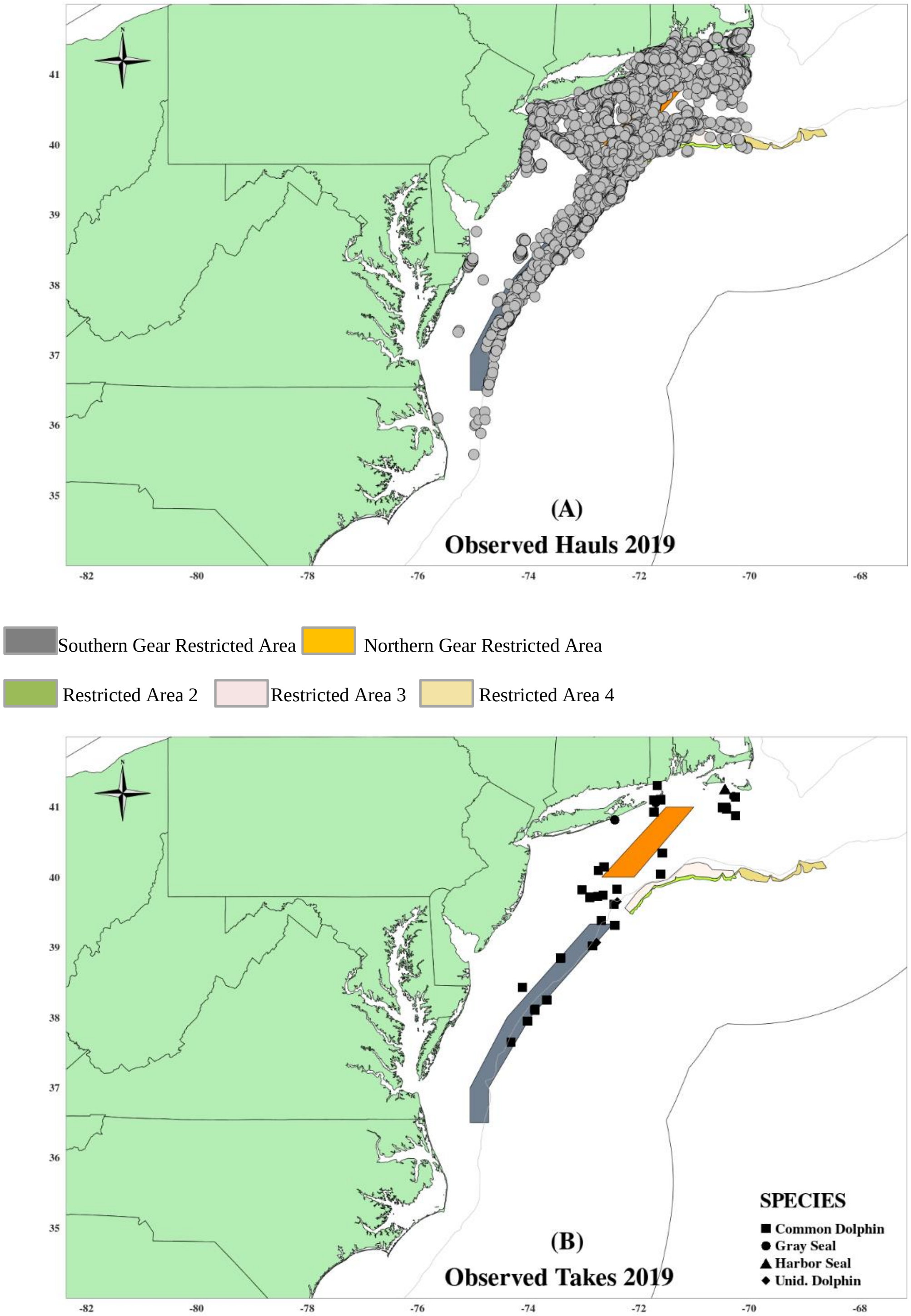
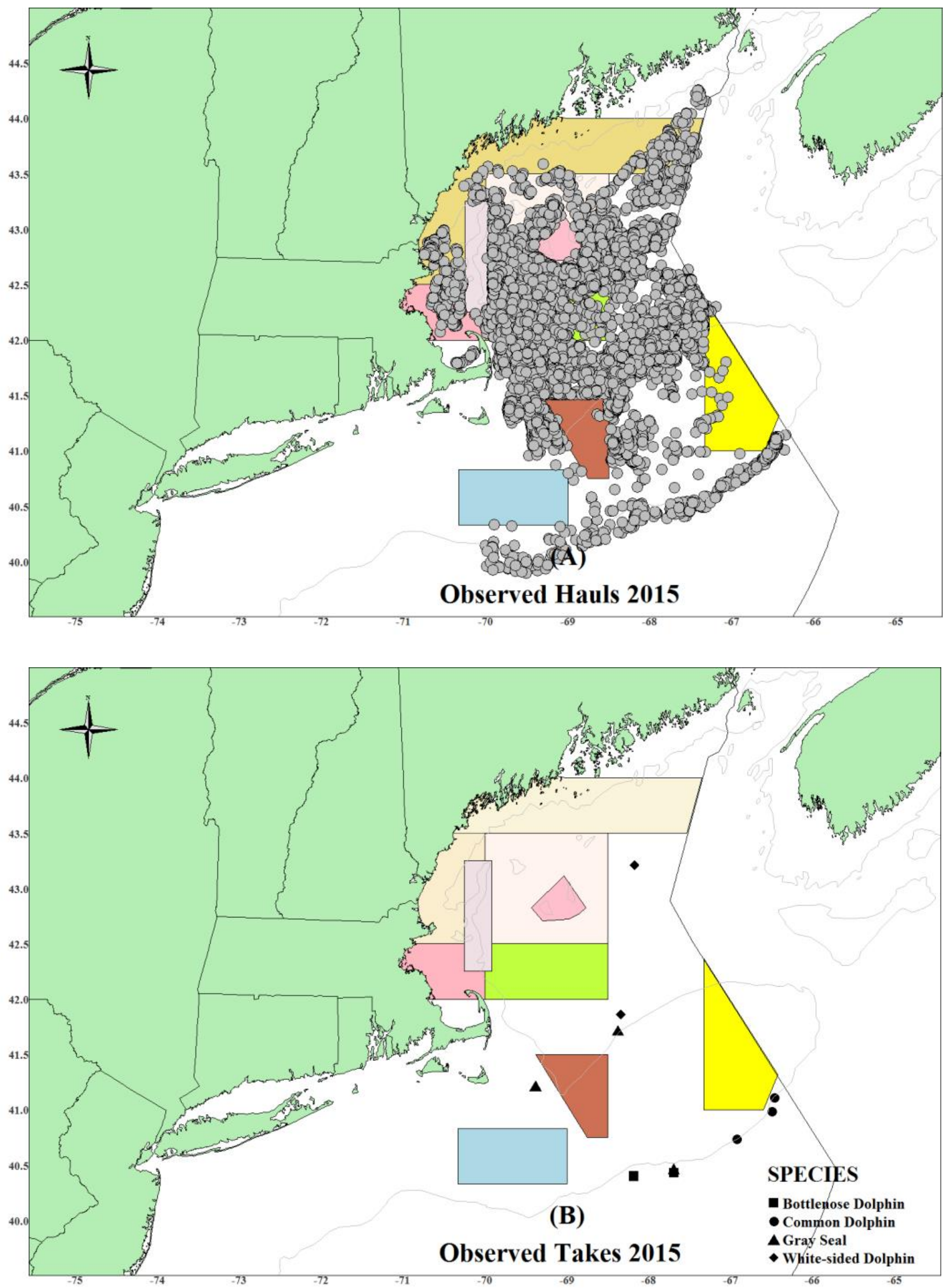


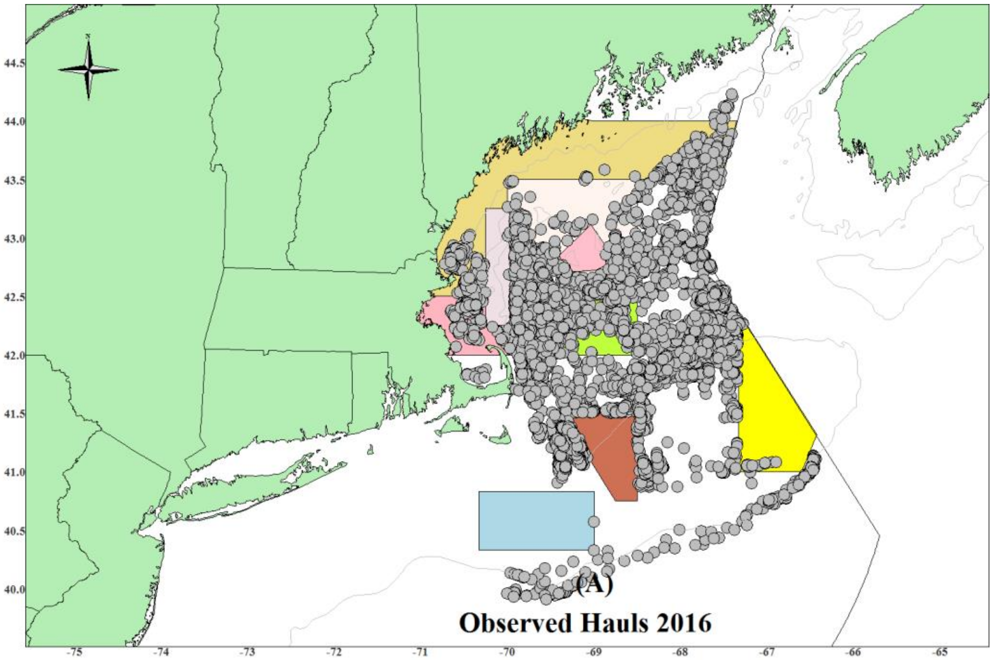
Figure 16. 2015 Northeast bottom trawl observed tows (A) and observed takes (B).

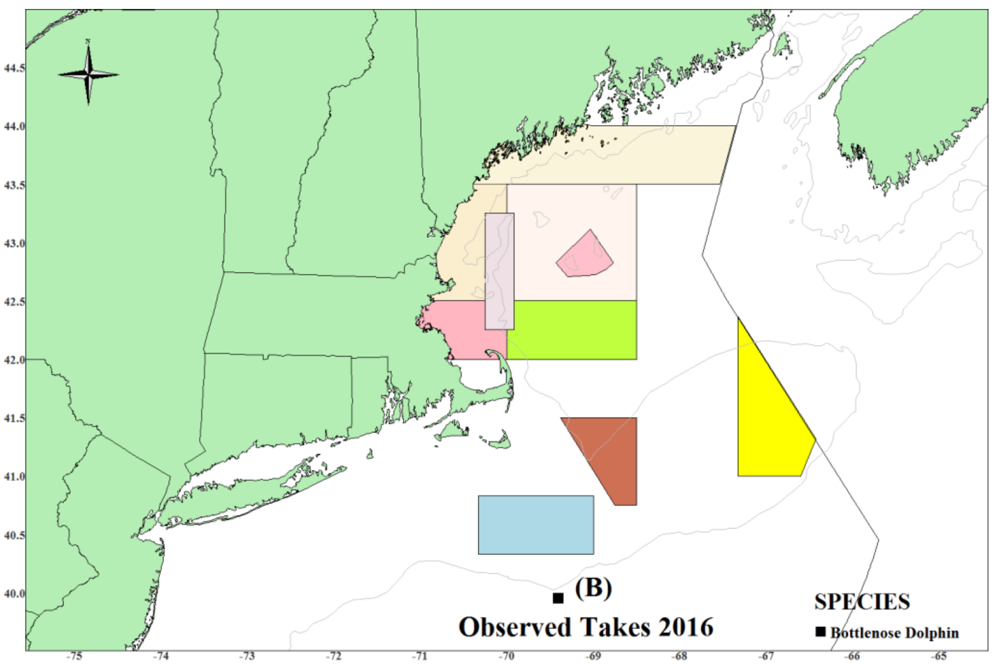


Closed Area 1 | Closed Area 2 | Western Gulf of Maine Closed Area | Nantucket Lightship Closed Area | Cashes Ledge Closed Area

Rolling Closure Area 1 | Rolling Closure Area 2 | Rolling Closure Area 3 | Rolling Closure Area 4 | Rolling Closure Area 5

Figure 17. 2016 Northeast bottom trawl observed tows (A) and observed takes (B).





Closed Area 1 Closed Area 2 Western Gulf of Maine Closed Area Nantucket Lightship Closed Area Cashes Ledge Closed Area

Rolling Closure Area 1 Rolling Closure Area 2 Rolling Closure Area 3 Rolling Closure Area 4 Rolling Closure Area 5

Figure 18. 2017 Northeast bottom trawl observed tows (A) and observed takes (B).

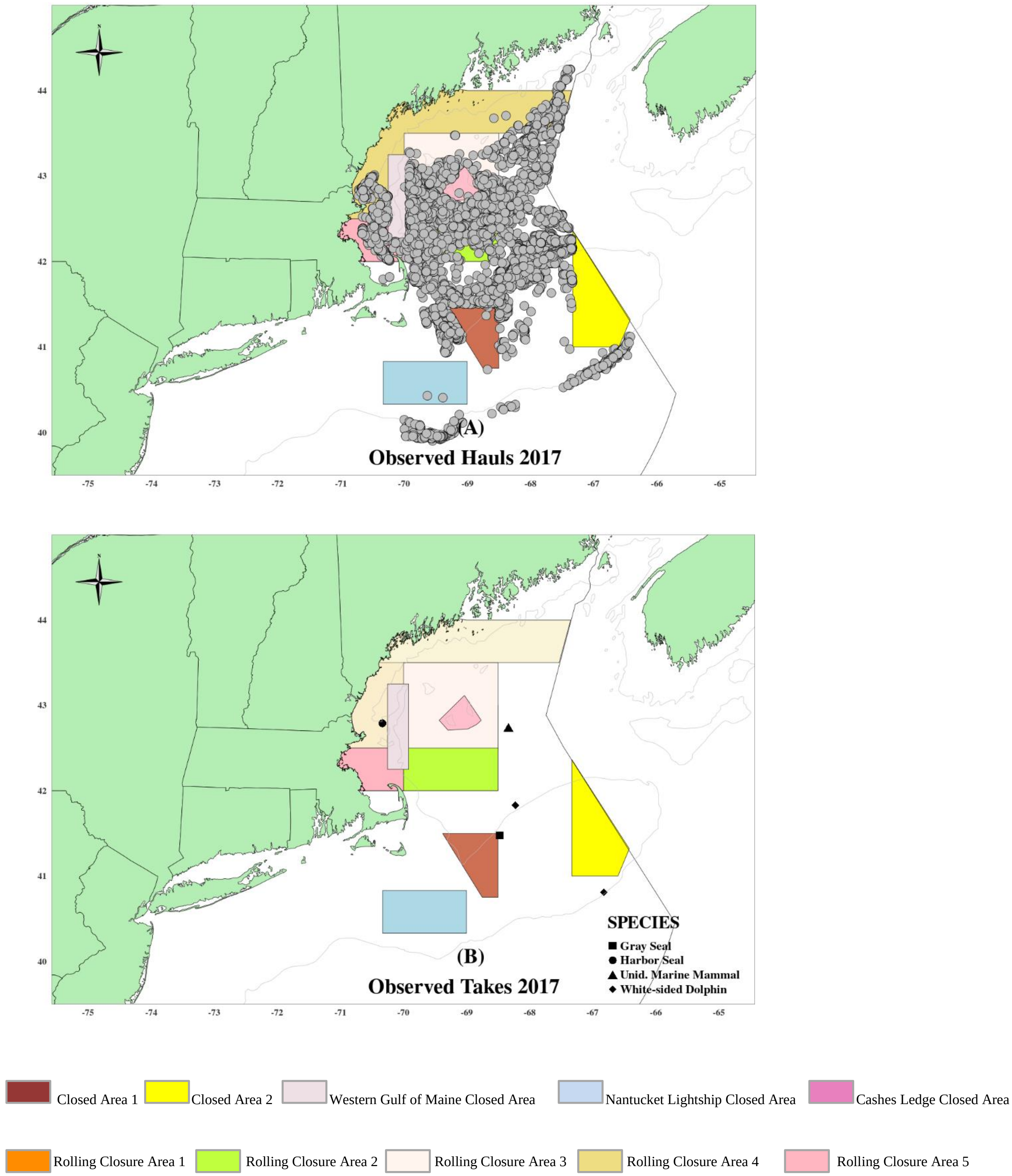


Figure 19. 2018 Northeast bottom trawl observed tows (A) and observed takes (B).

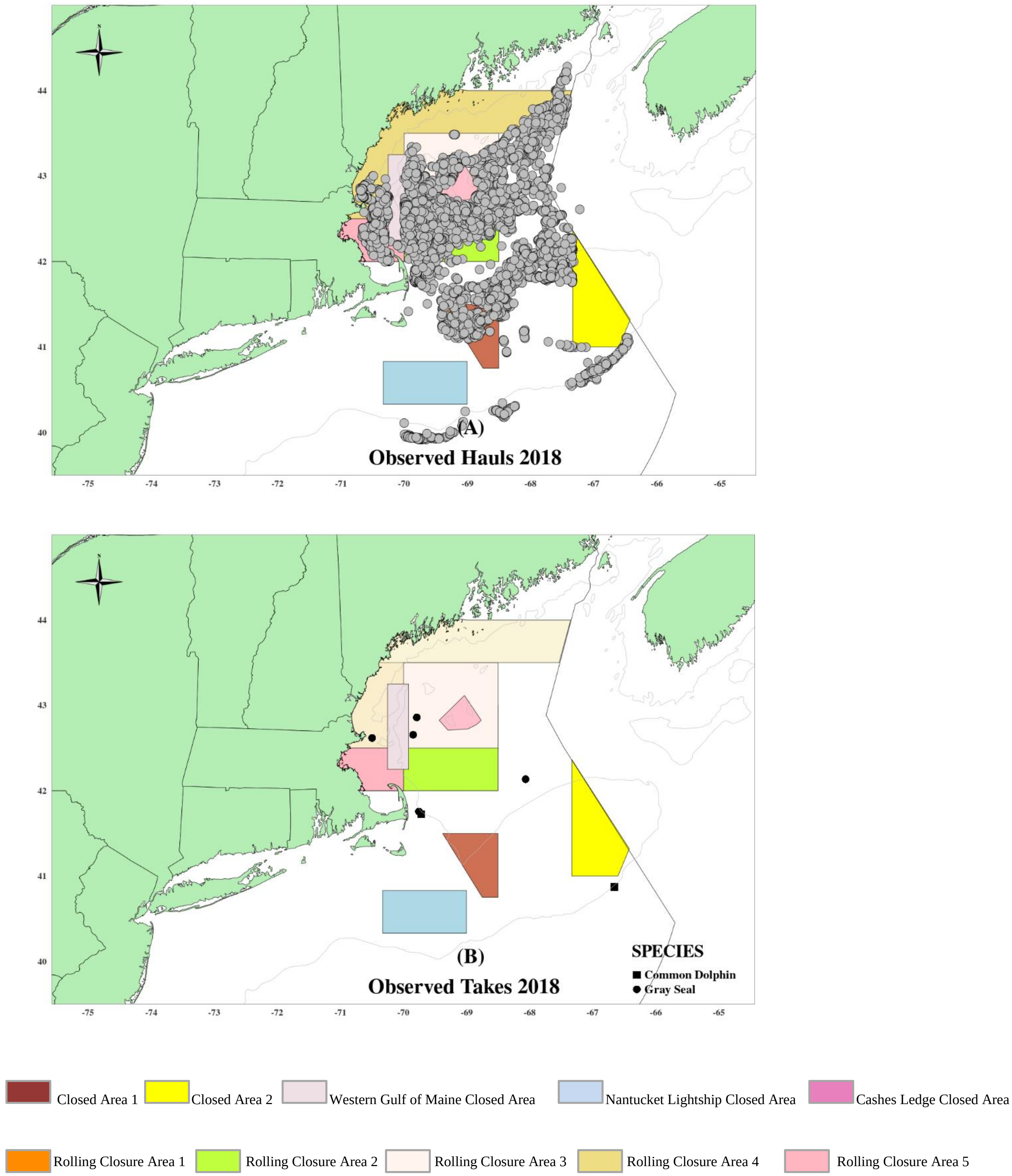


Figure 20. 2019 Northeast bottom trawl observed tows (A) and observed takes (B).

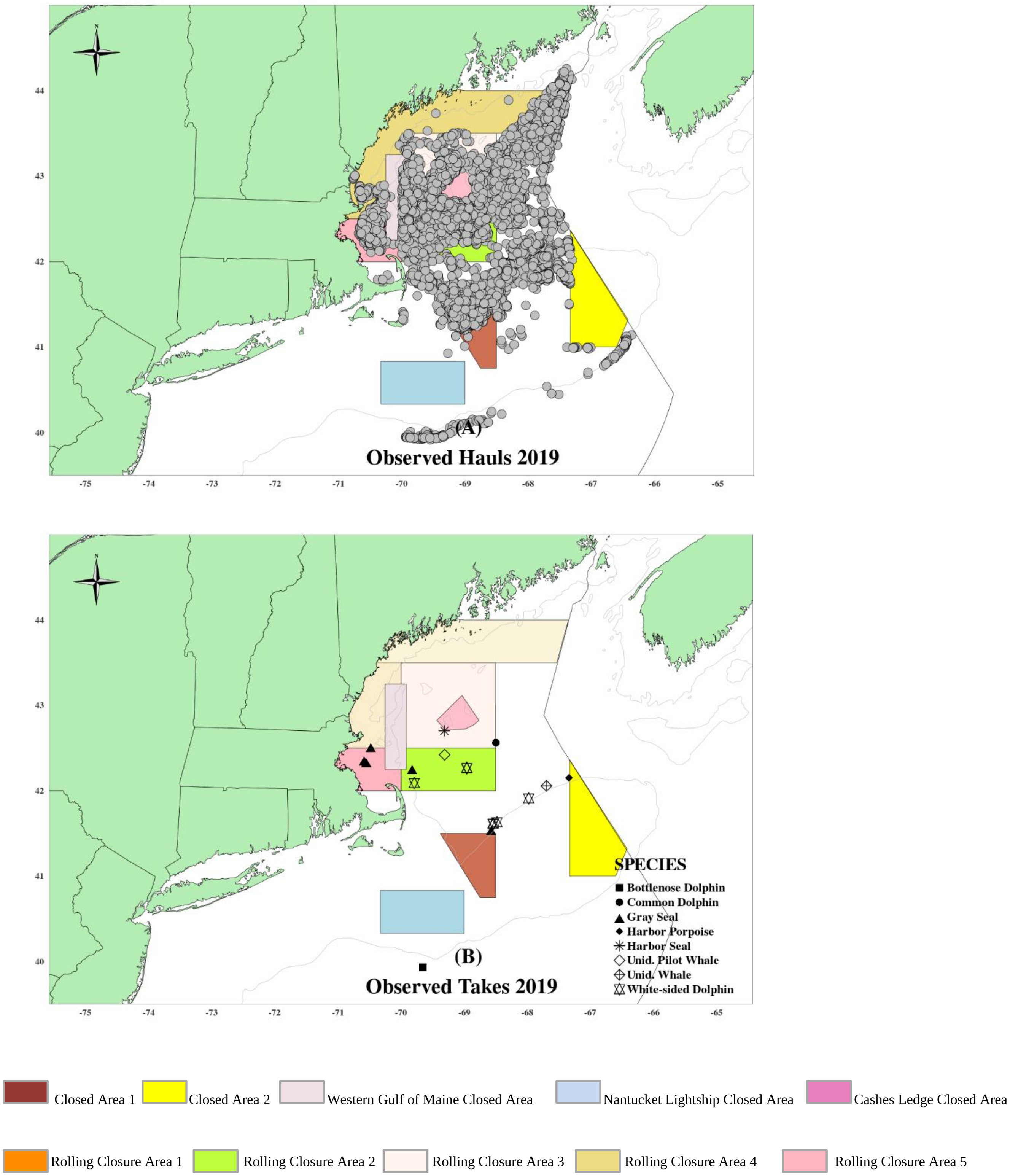


Figure 21. 2015 Northeast mid-water trawl observed tows (A) and observed takes (B).

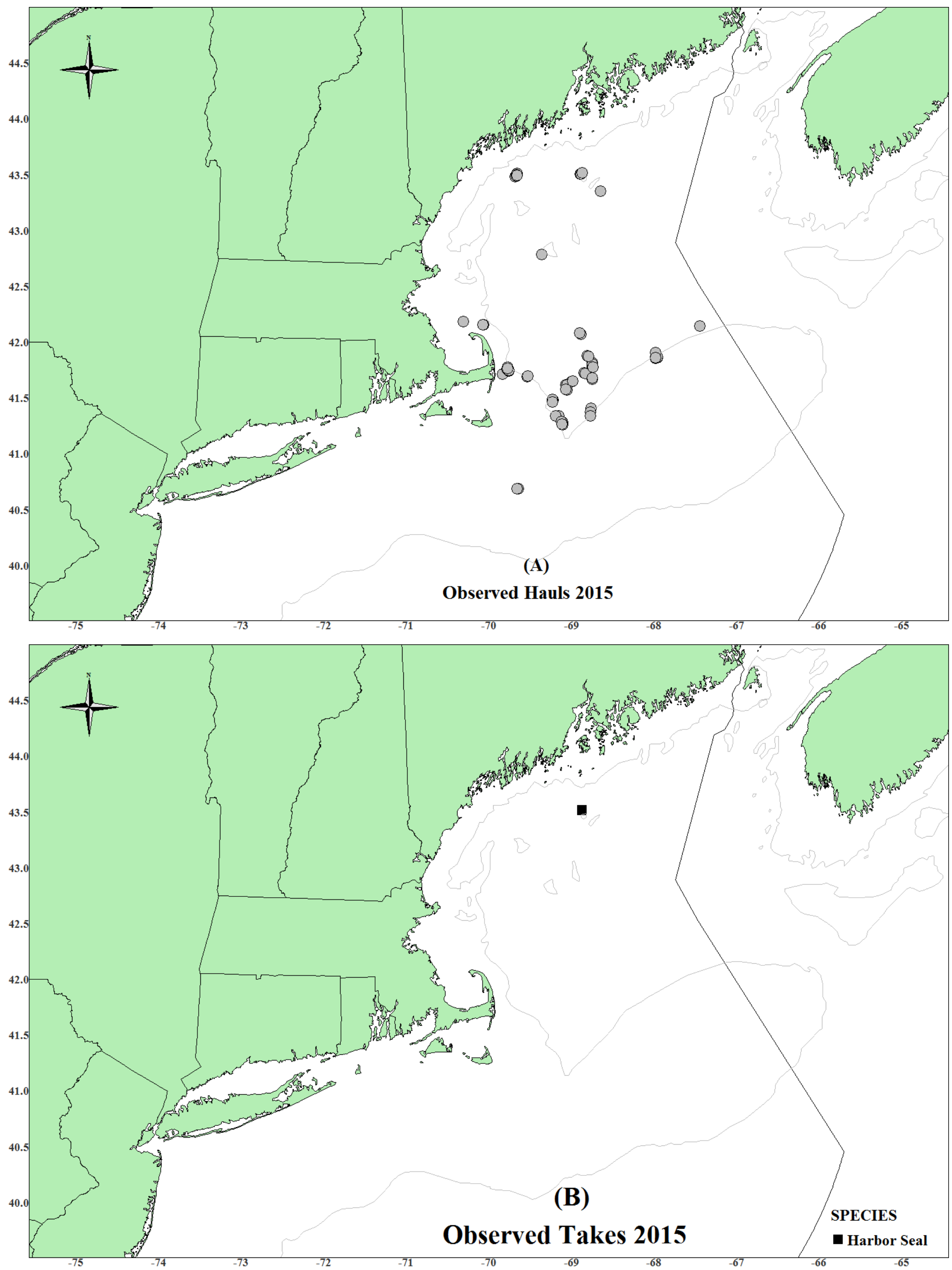


Figure 22. 2016 Northeast mid-water trawl observed tows (A) and observed takes (B).

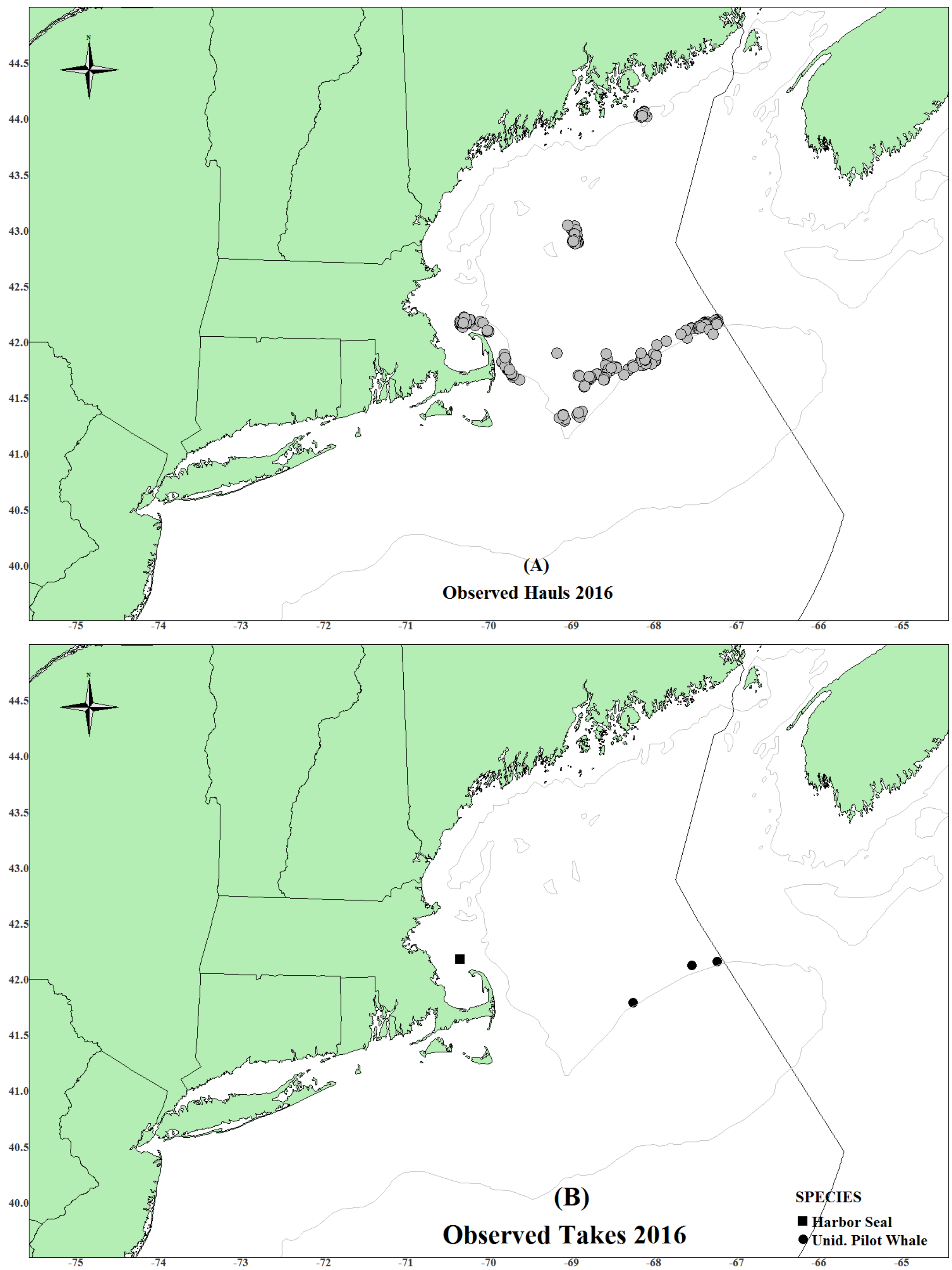
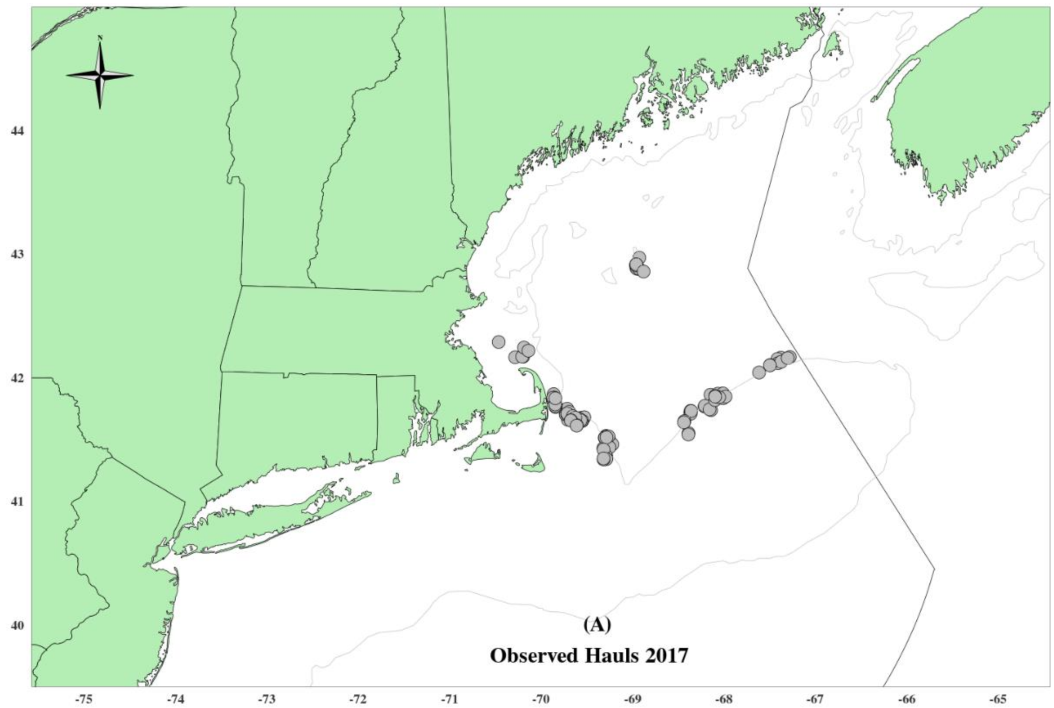


Figure 23. 2017 Northeast mid-water trawl observed tows (A) and observed takes (B).



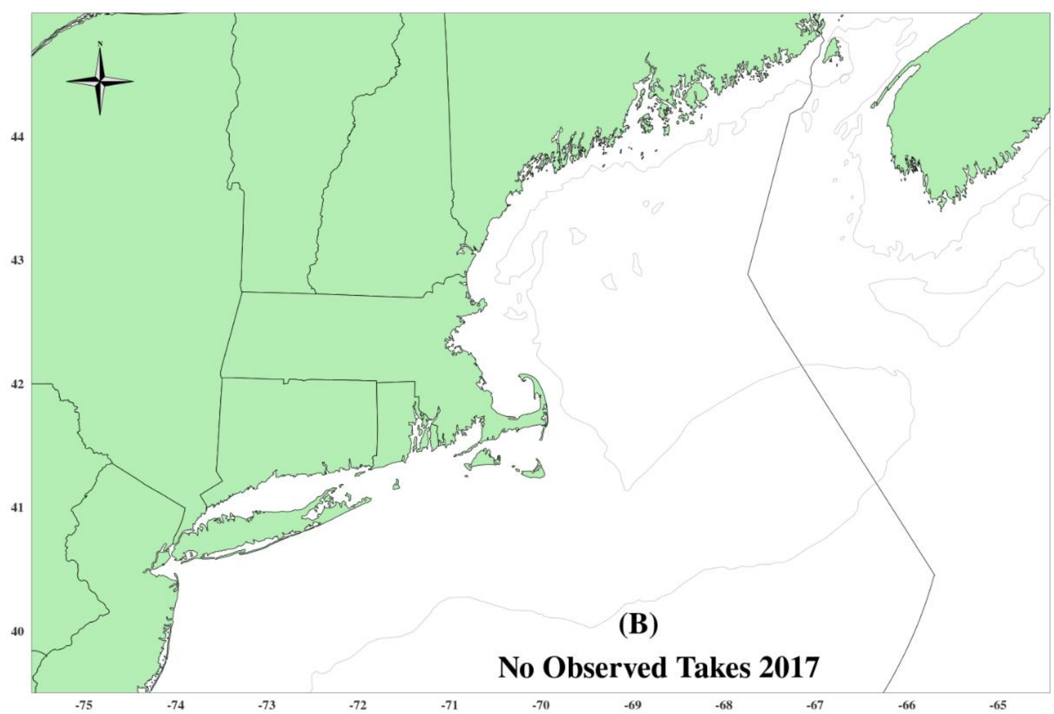


Figure 24. 2018 Northeast mid-water trawl observed tows (A) and observed takes (B).

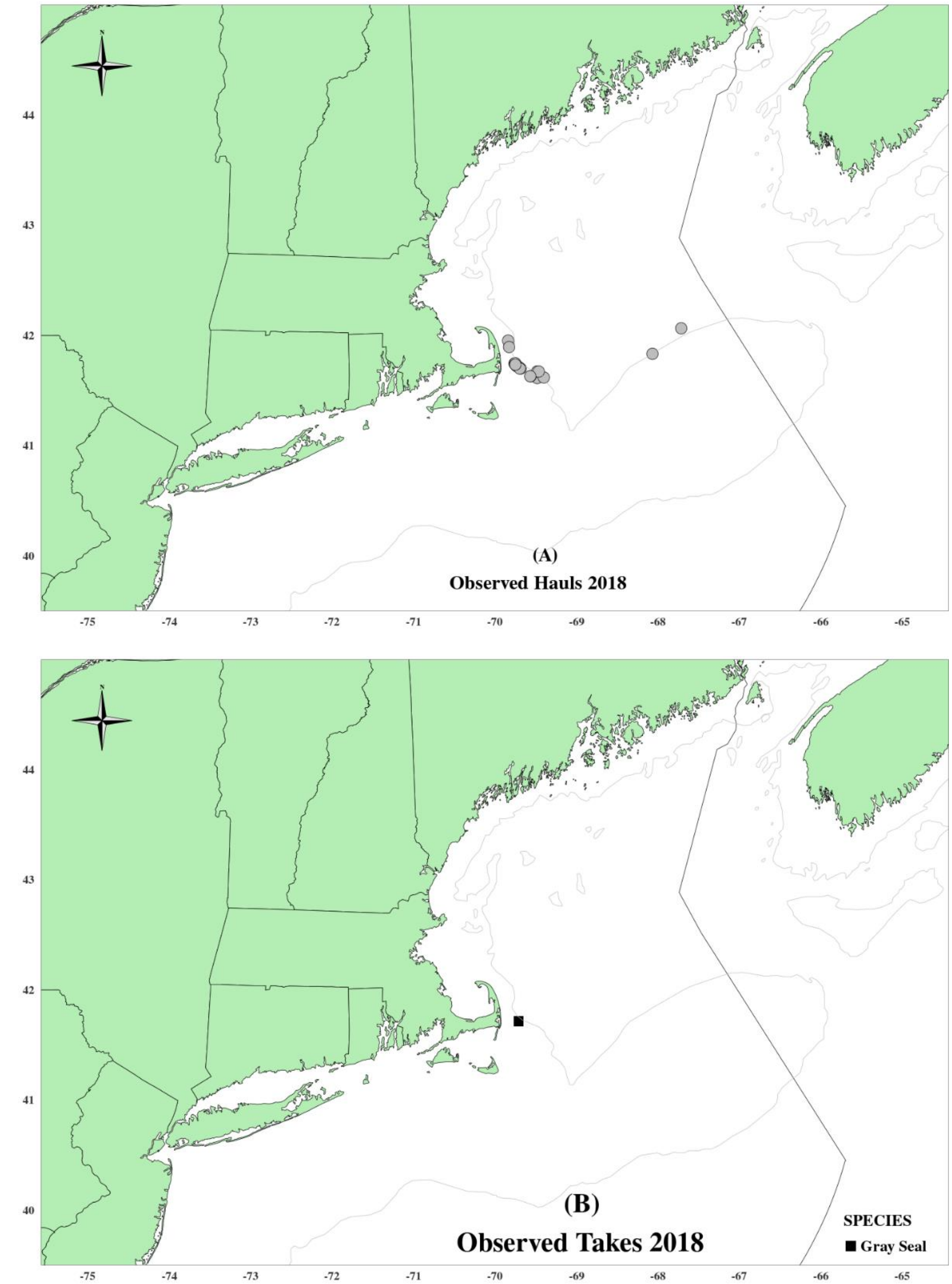


Figure 25. 2019 Northeast mid-water trawl observed tows (A) and observed takes (B).

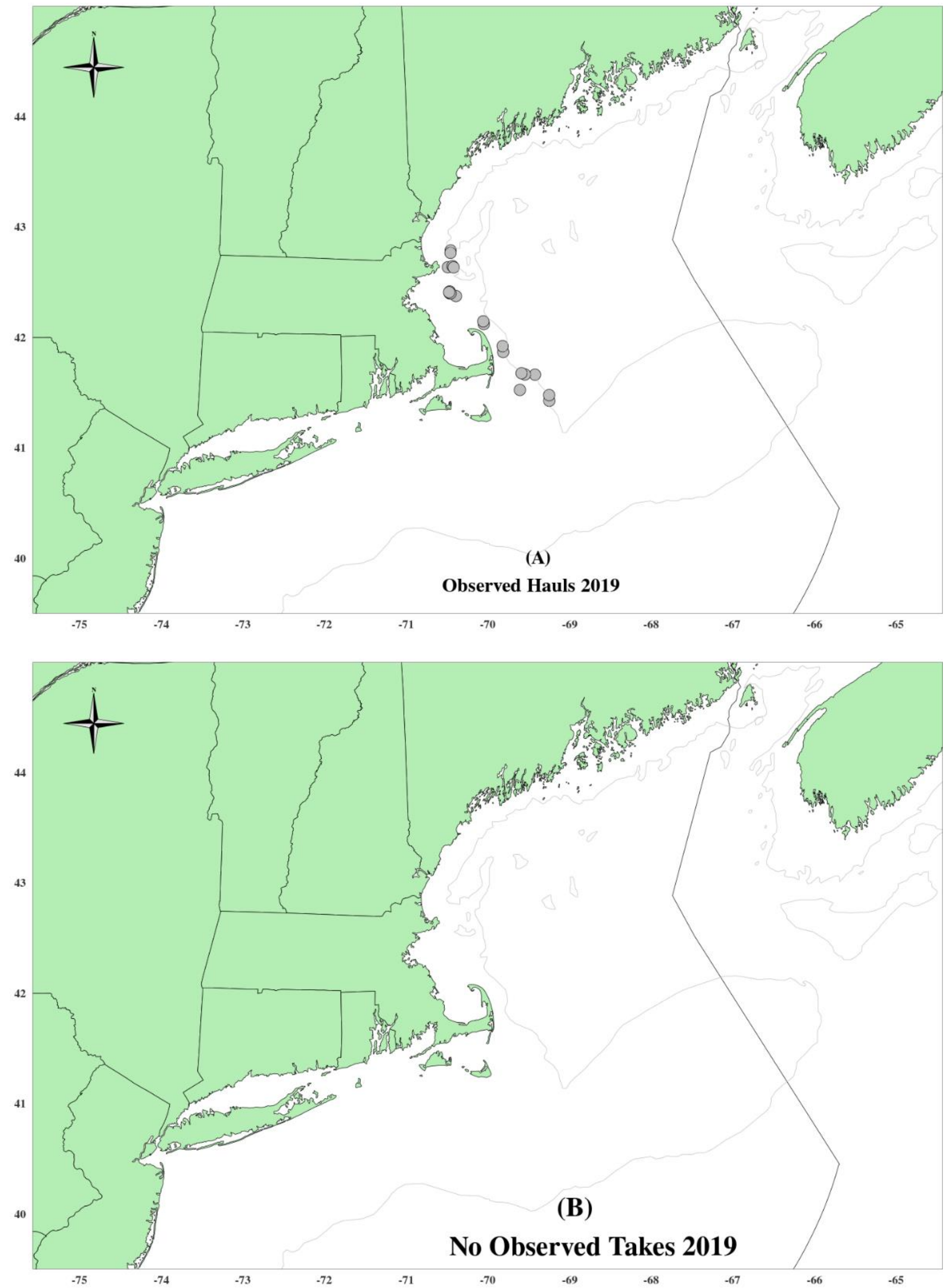
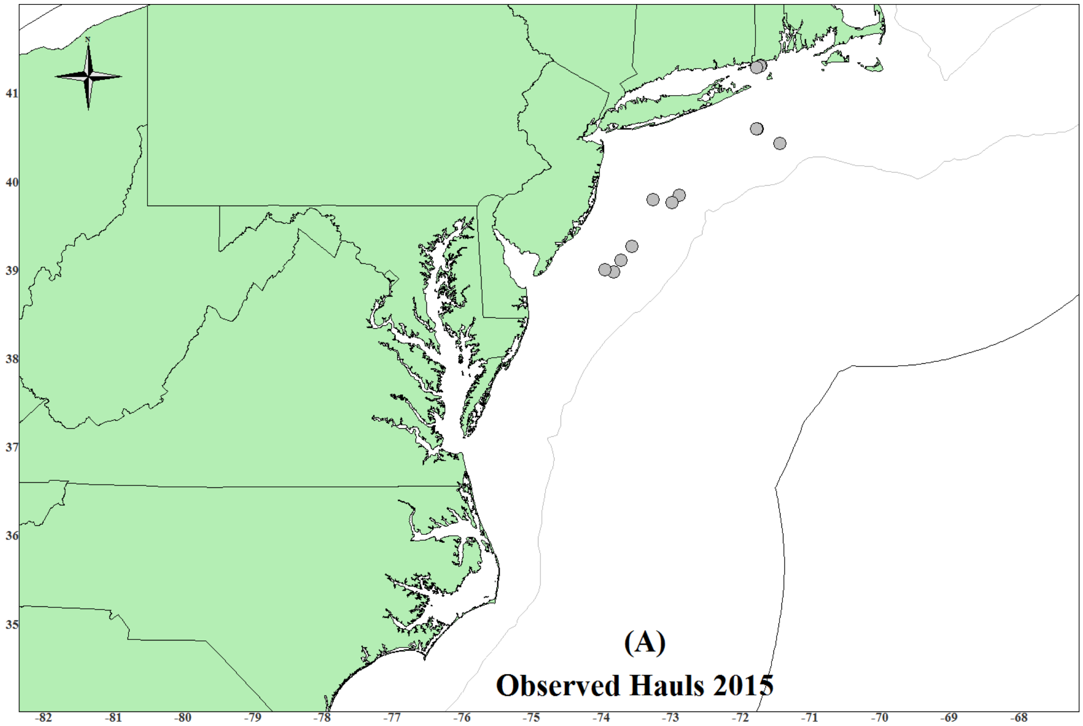


Figure 26. 2015 Mid-Atlantic mid-water trawl observed tows (A) and observed takes (B).



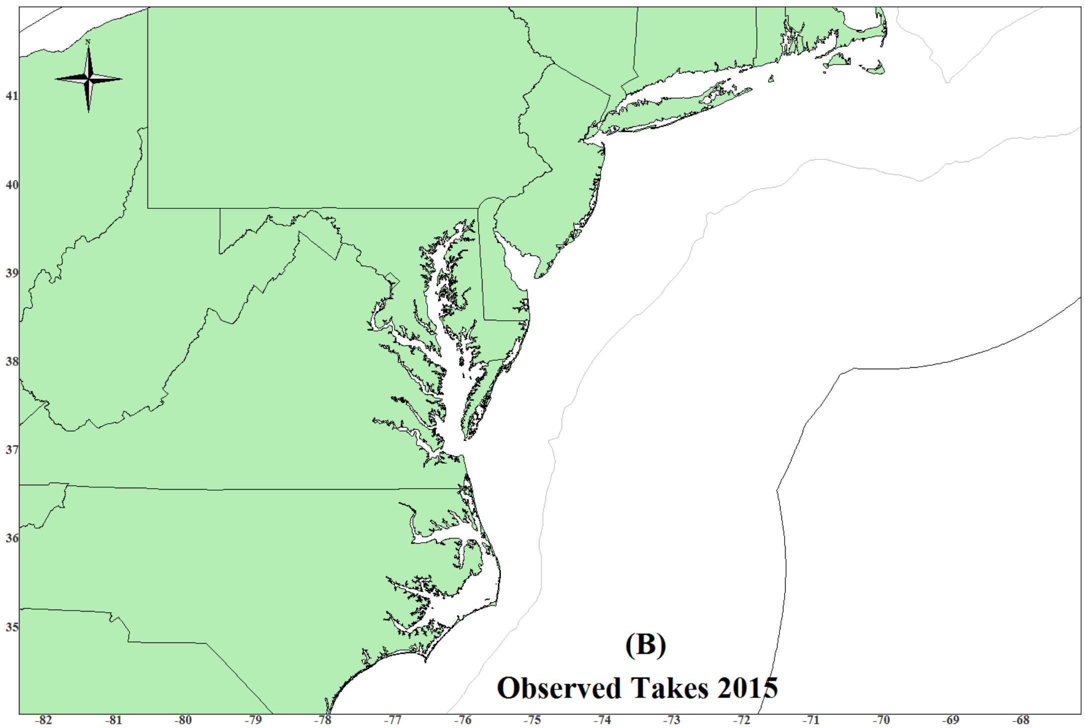
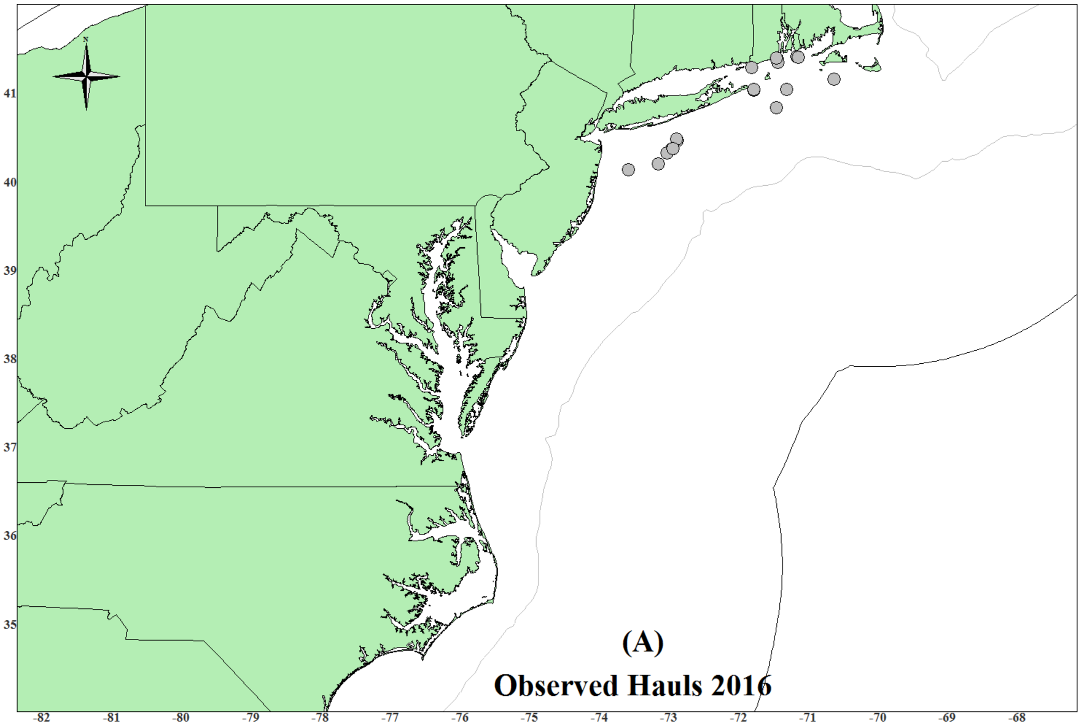


Figure 27. 2016 Mid-Atlantic mid-water trawl observed tows (A) and observed takes (B).



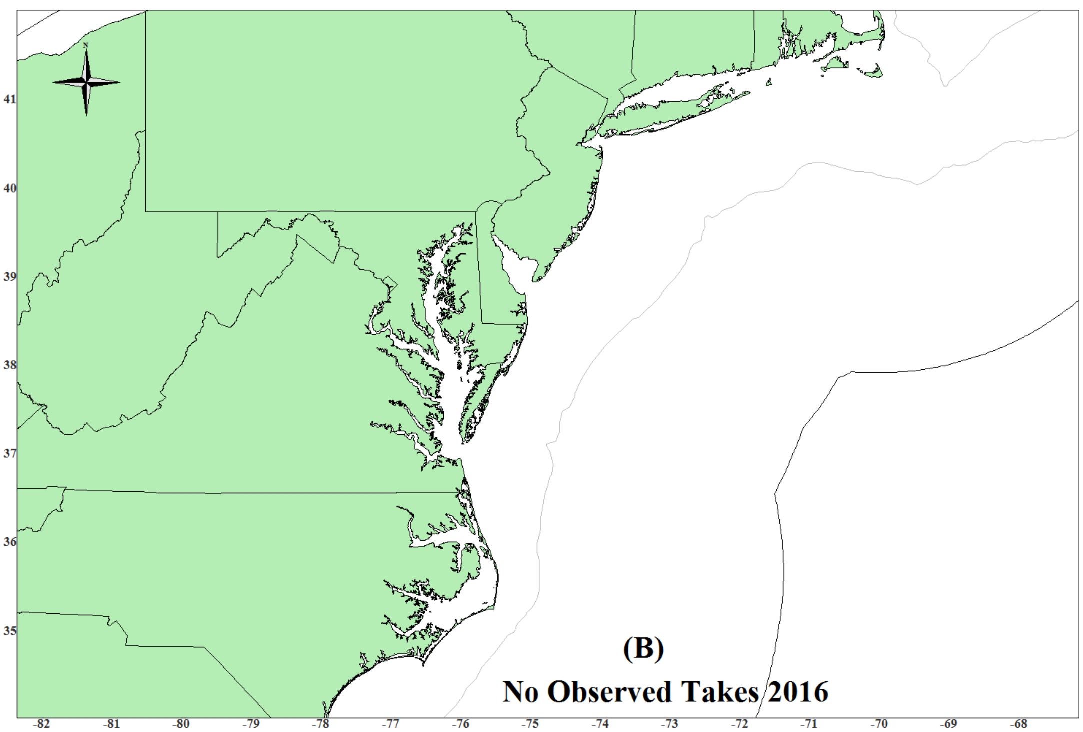
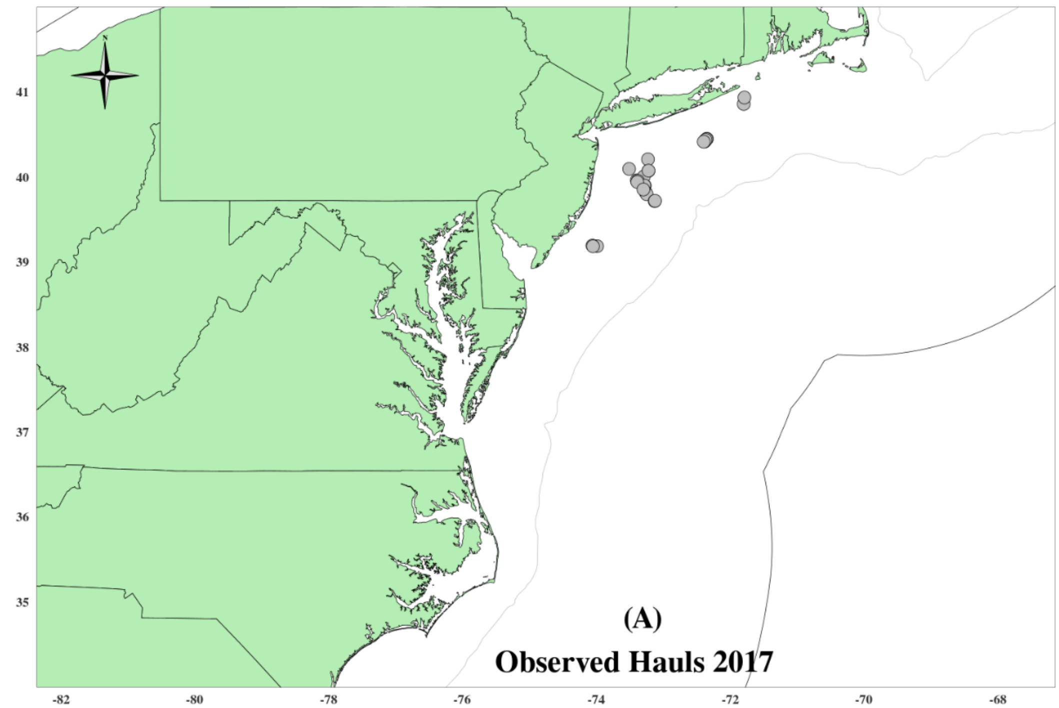


Figure 28. 2017 Mid-Atlantic mid-water trawl observed tows (A) and observed takes (B).



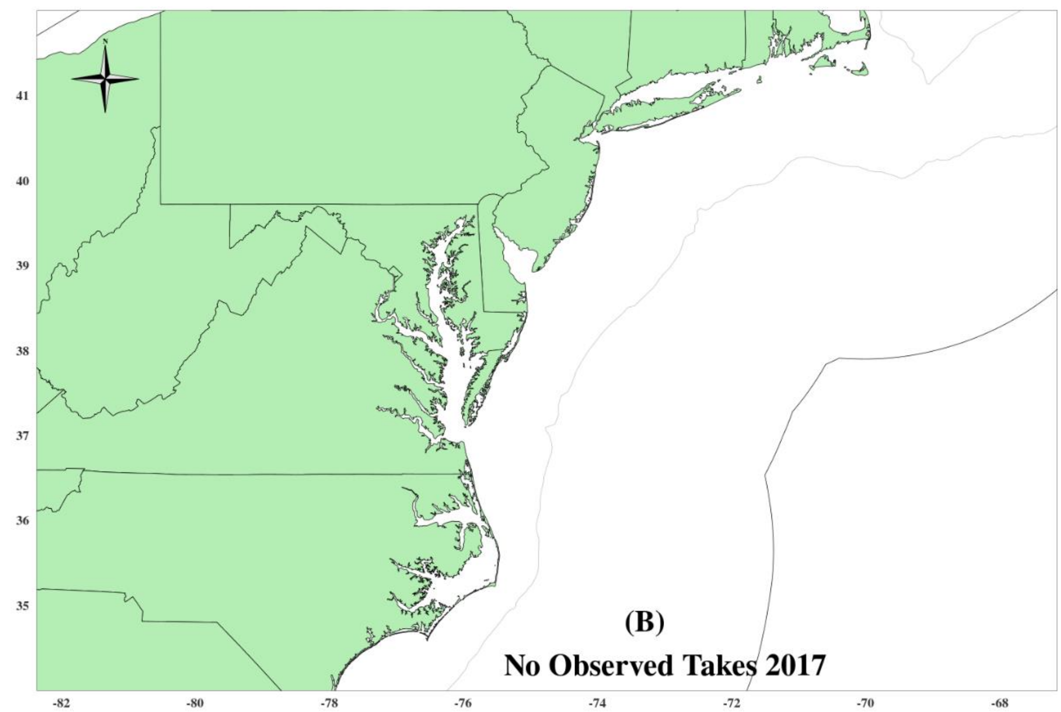
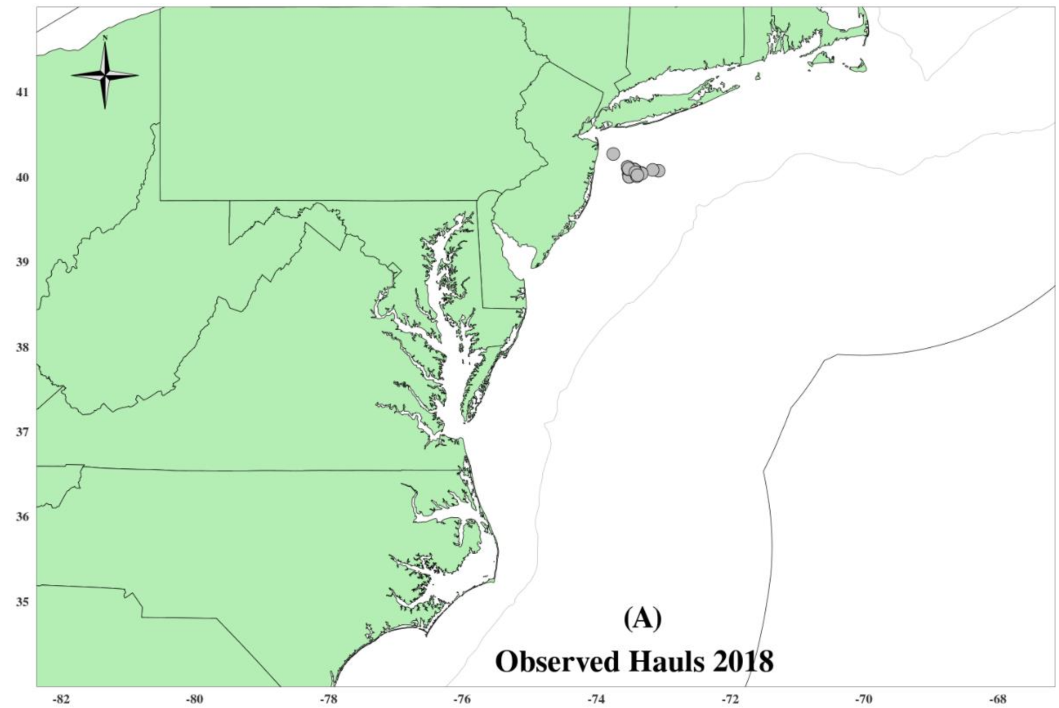


Figure 29. 2018 Mid-Atlantic mid-water trawl observed tows (A) and observed takes (B).



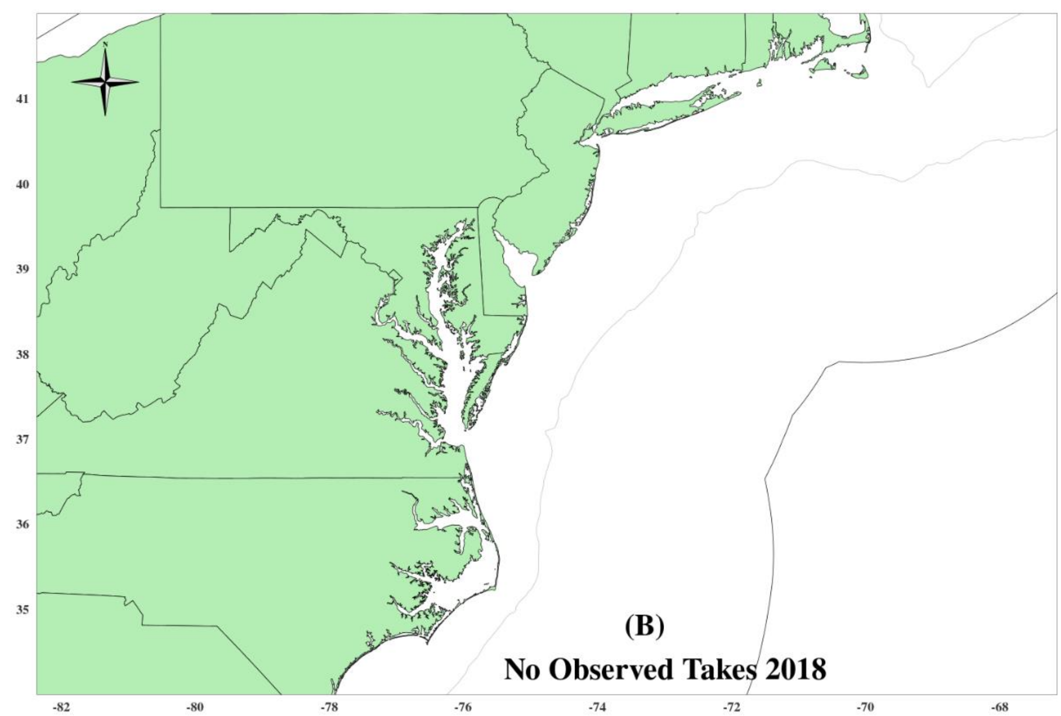


Figure 30. 2019 Mid-Atlantic mid-water trawl observed tows (A) and observed takes (B).

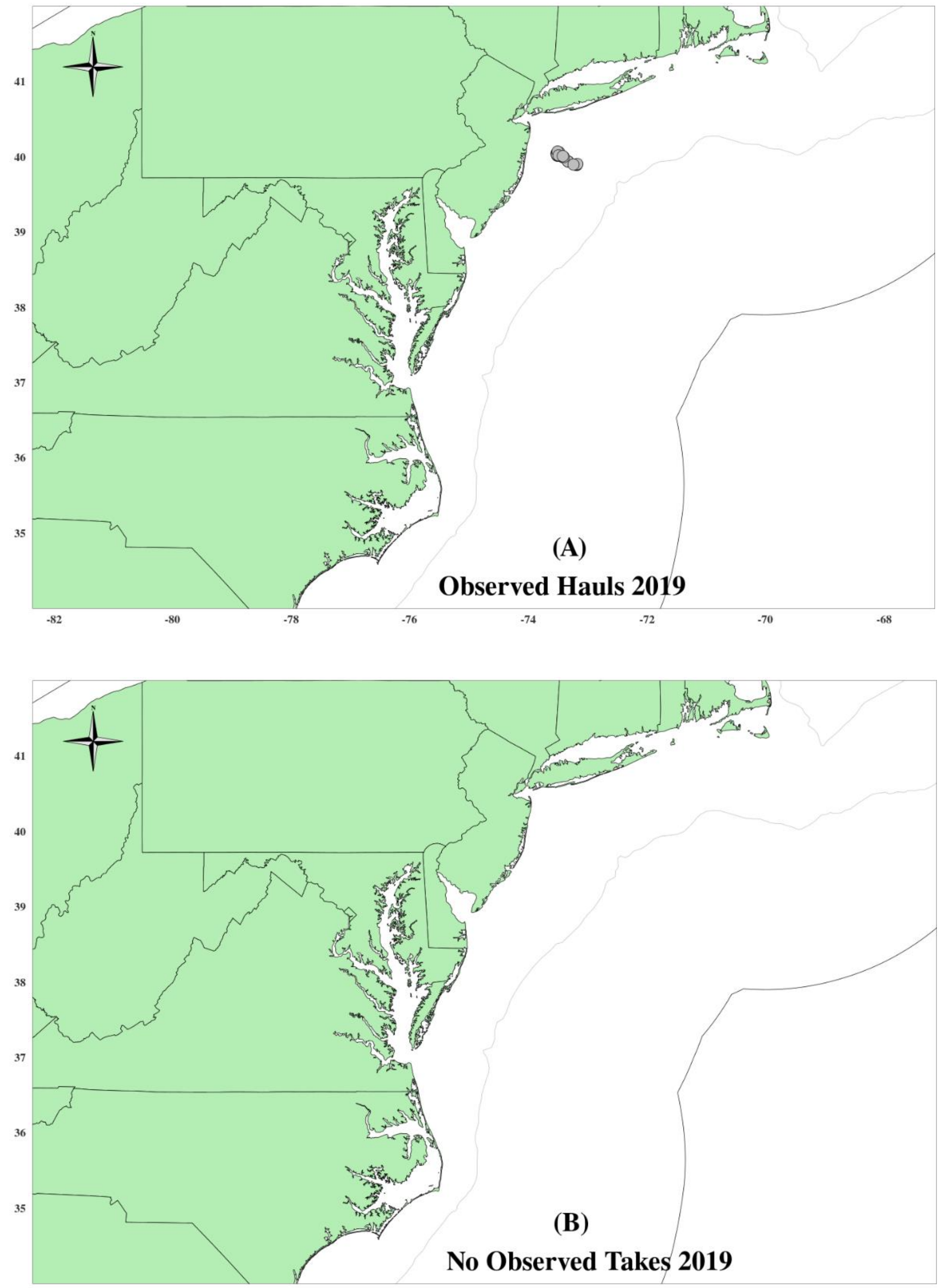


Figure 31. 2015 Herring Purse Seine observed hauls (A) and observed takes (B).

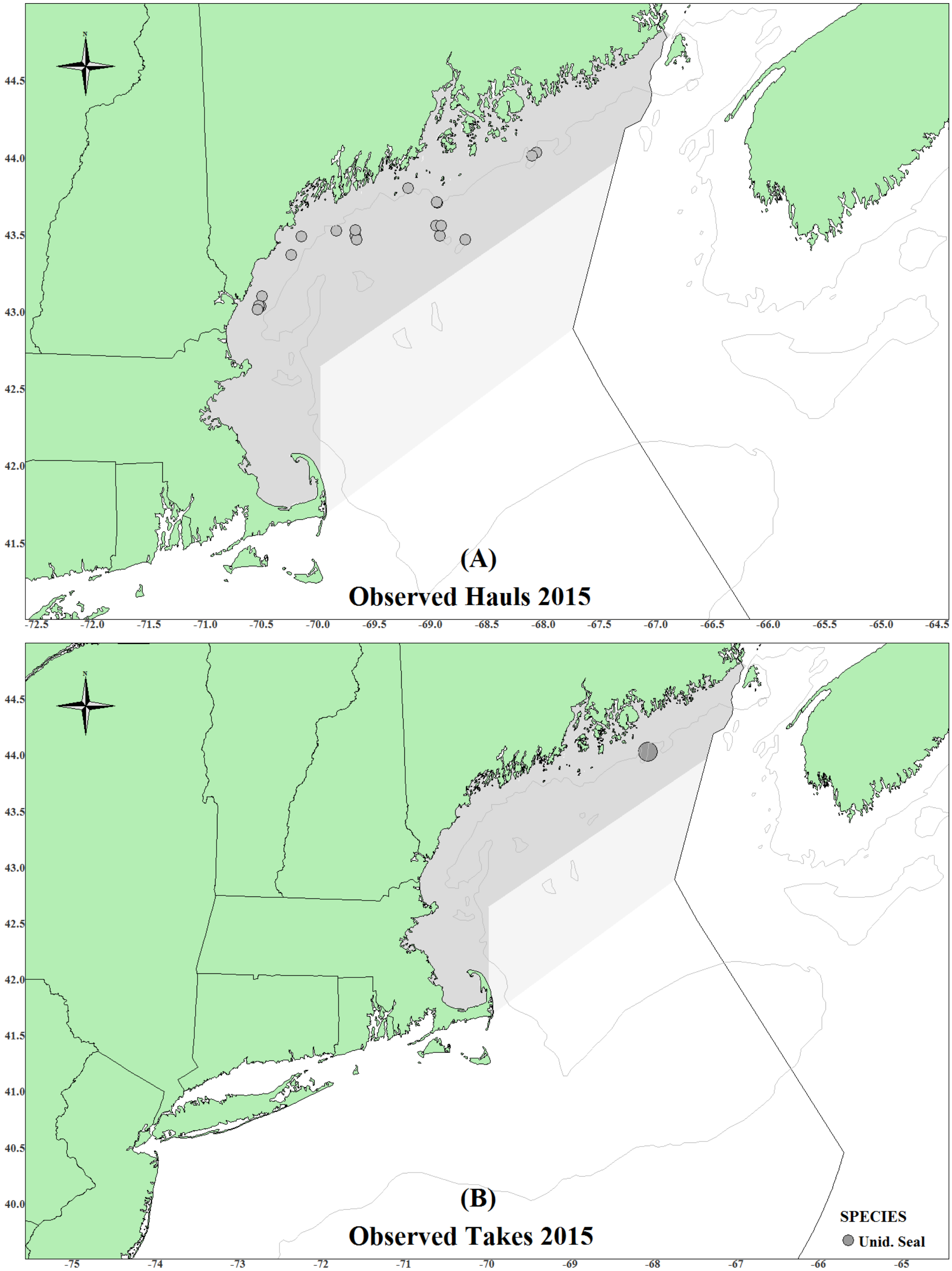
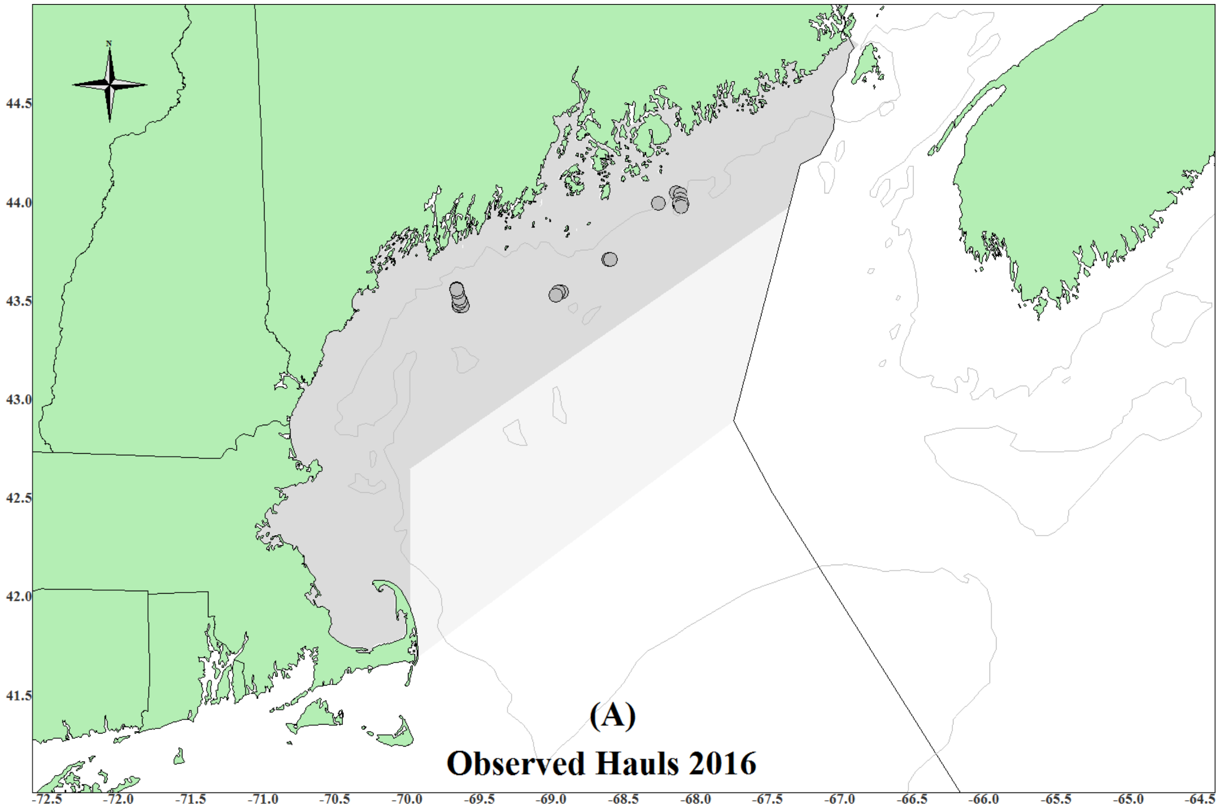


Figure 32. 2016 Herring Purse Seine observed hauls (A) and observed takes (B).



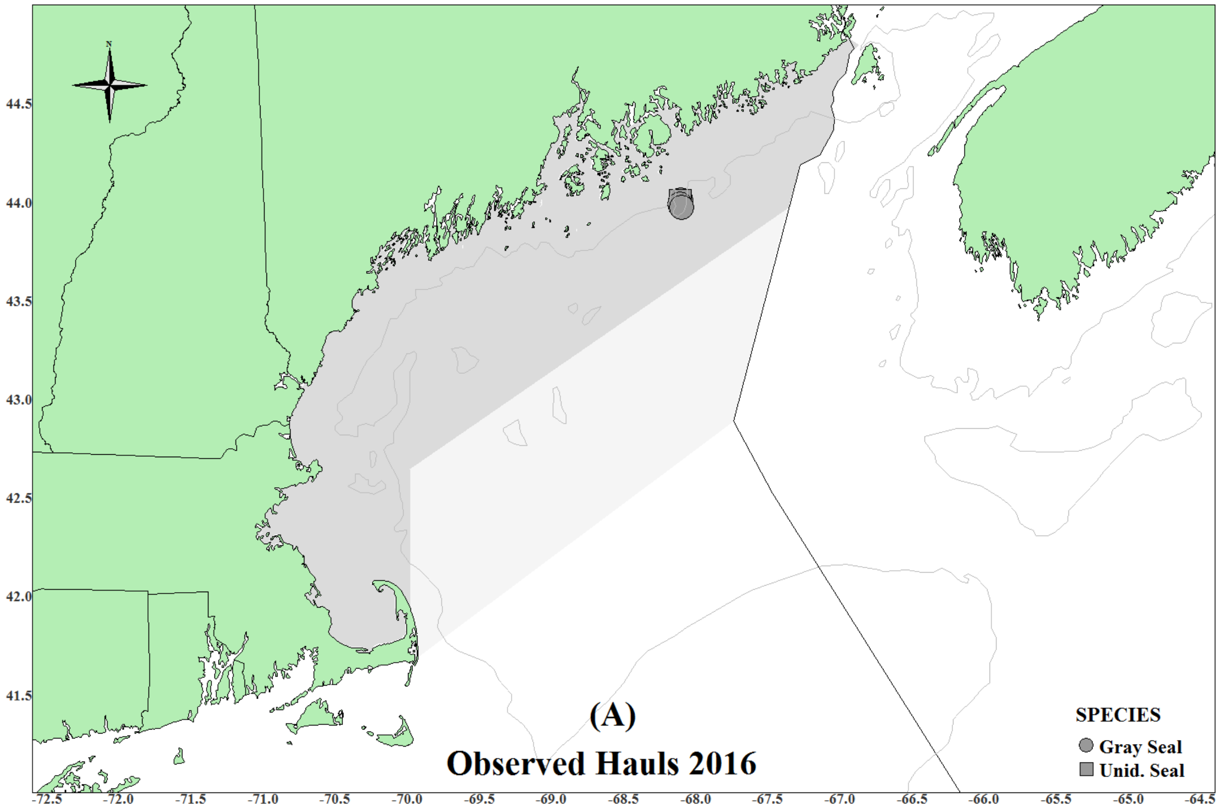
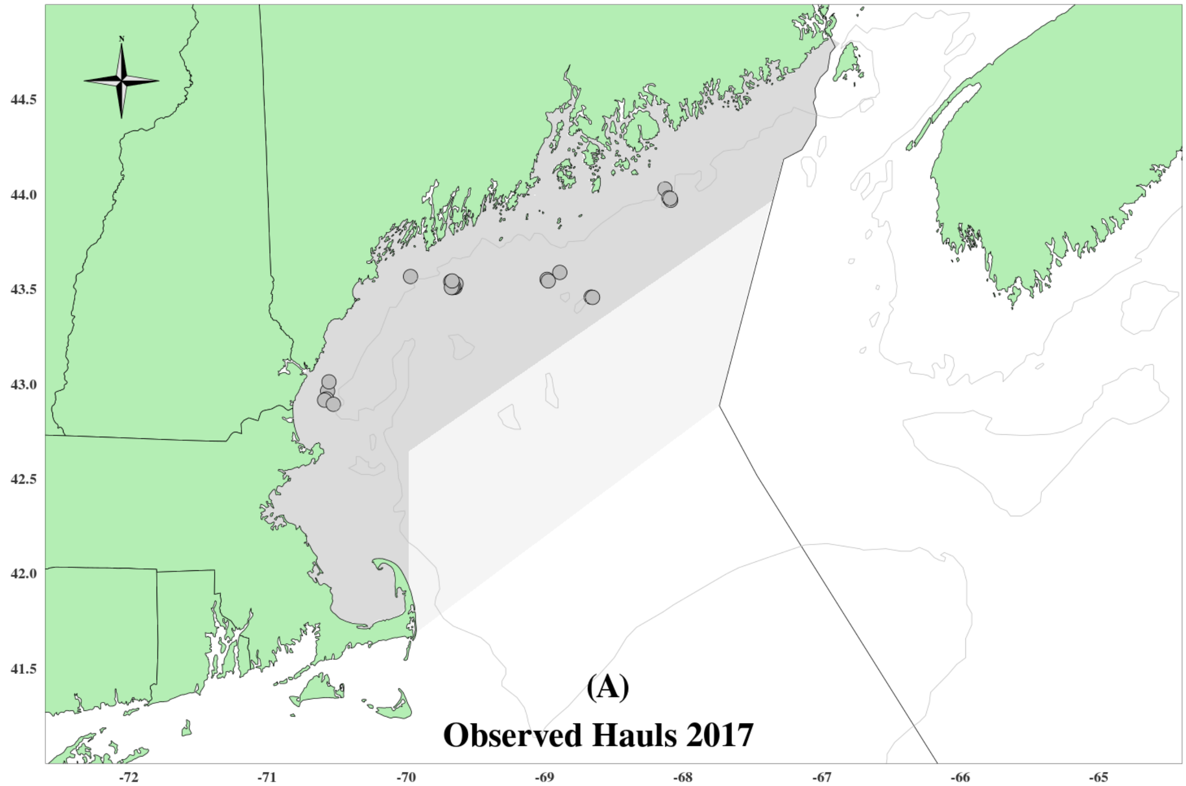


Figure 33. 2017 Herring Purse Seine observed hauls (A) and observed takes (B).



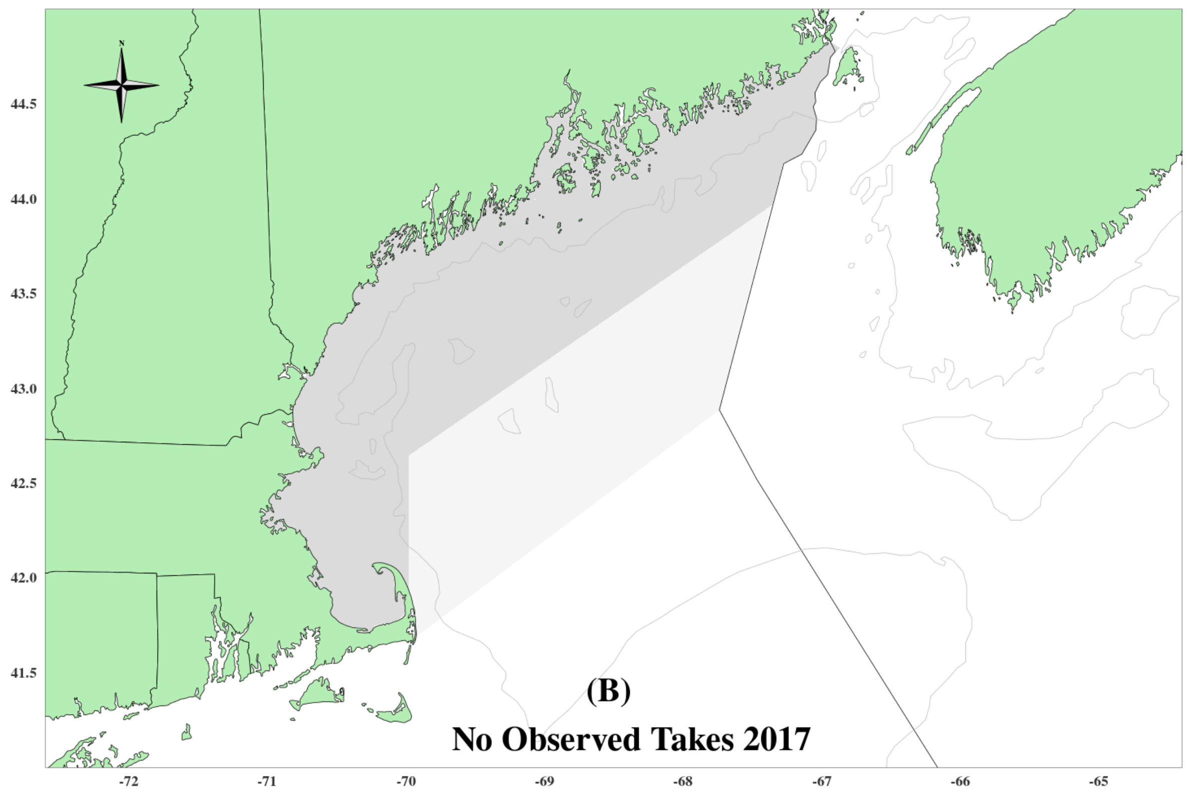
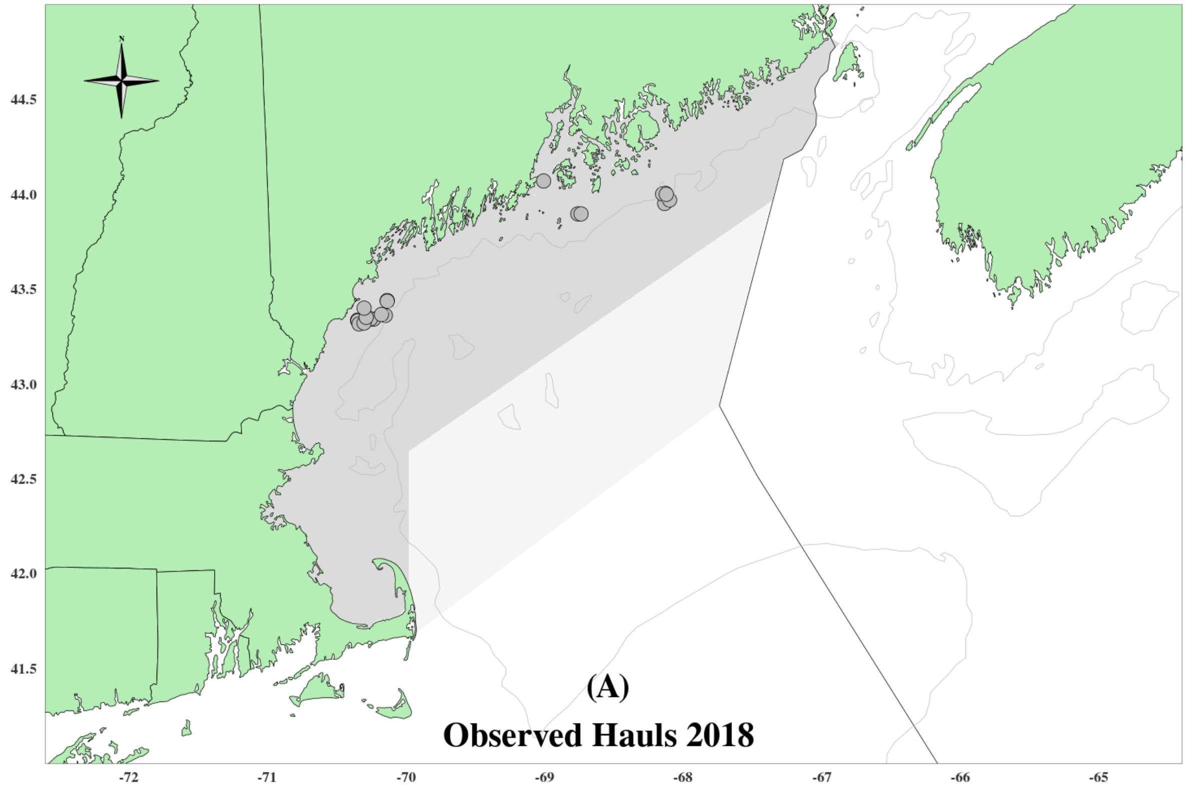


Figure 34. 2018 Herring Purse Seine observed hauls (A) and observed takes (B).



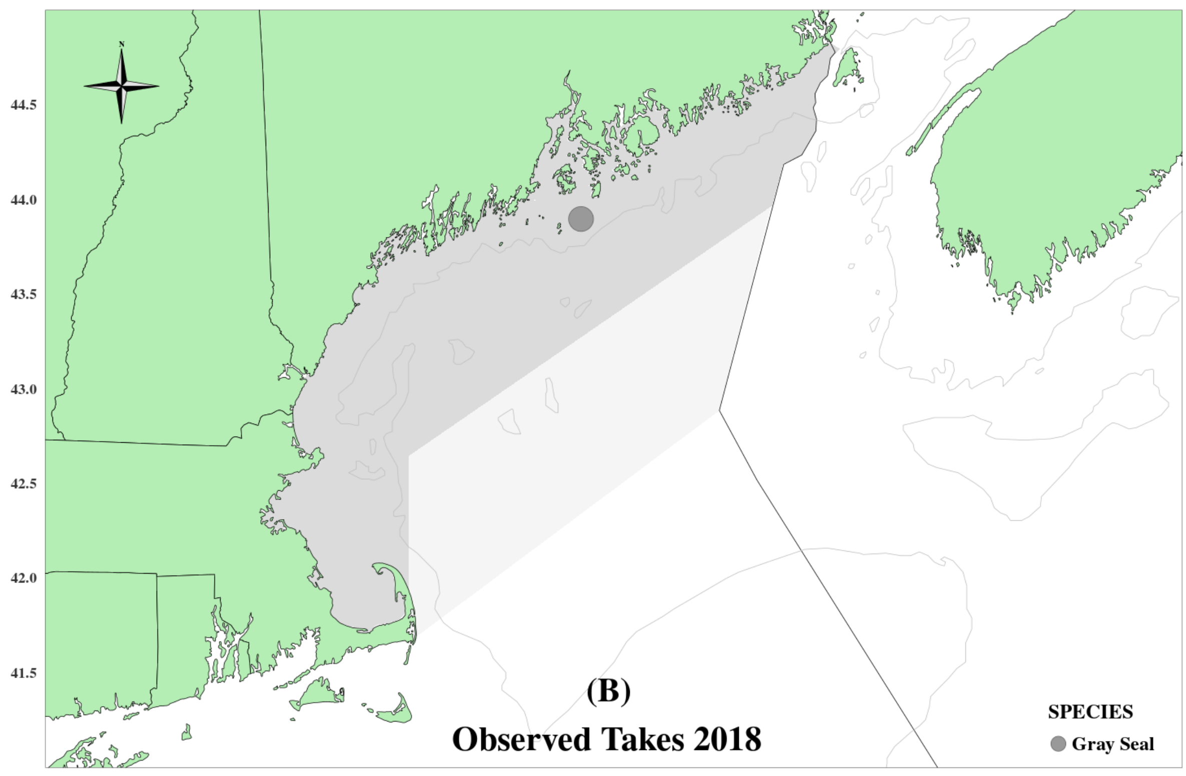
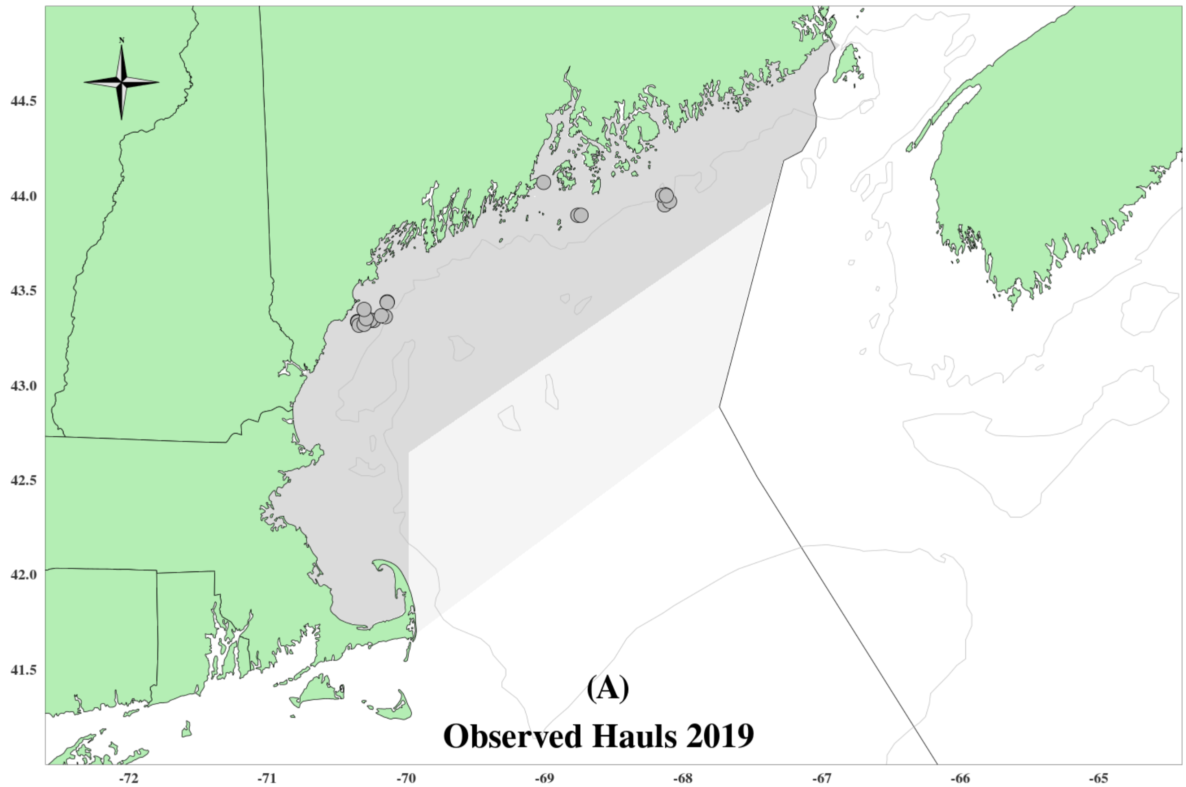


Figure 35. 2019 Herring Purse Seine observed hauls (A) and observed takes (B).



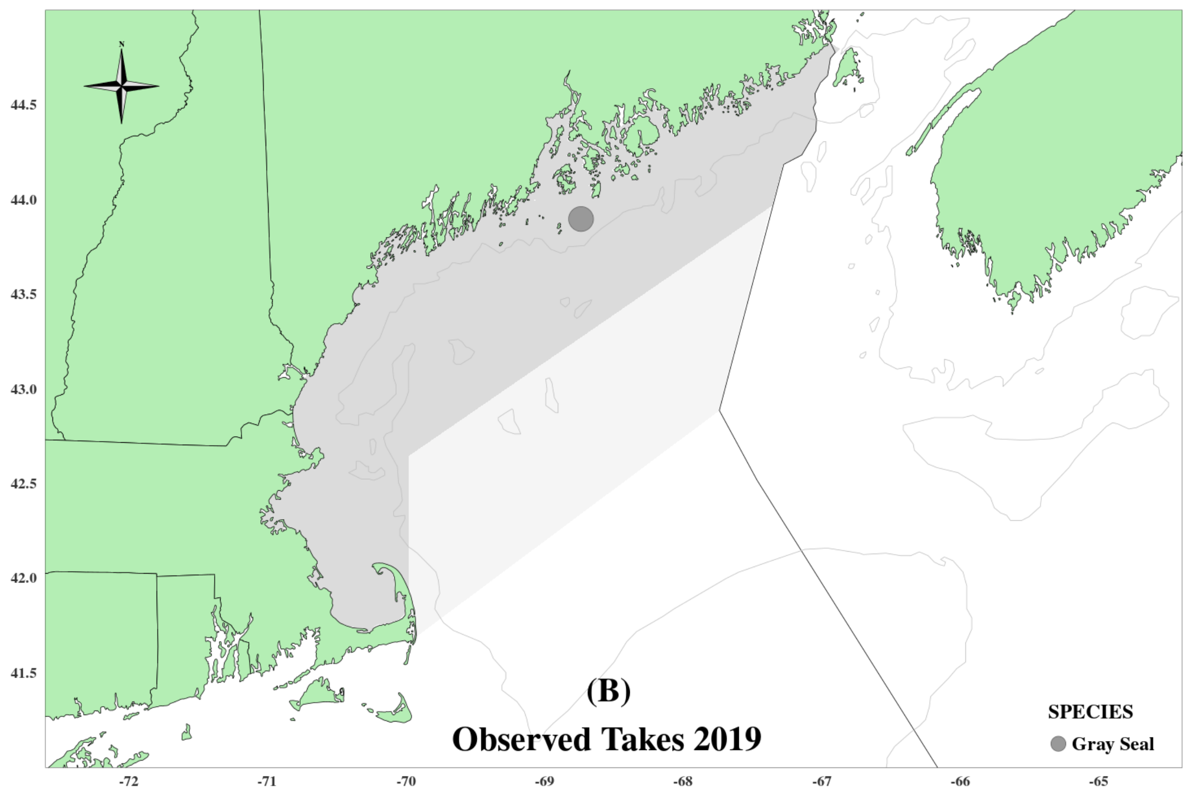


Figure 36. Observed sets and marine mammal interactions in the Pelagic Longline Fishery along the U.S. Atlantic coast during 2015. The boundaries of the Florida East Coast (FEC), South Atlantic Bight (SAB), Mid-Atlantic Bight (MAB), Northeast Coastal (NEC), and Sargasso Sea (SAR) fishing areas are shown. Seasonal closed areas instituted in 2001 under the HMS FMP are shown as hatched areas.

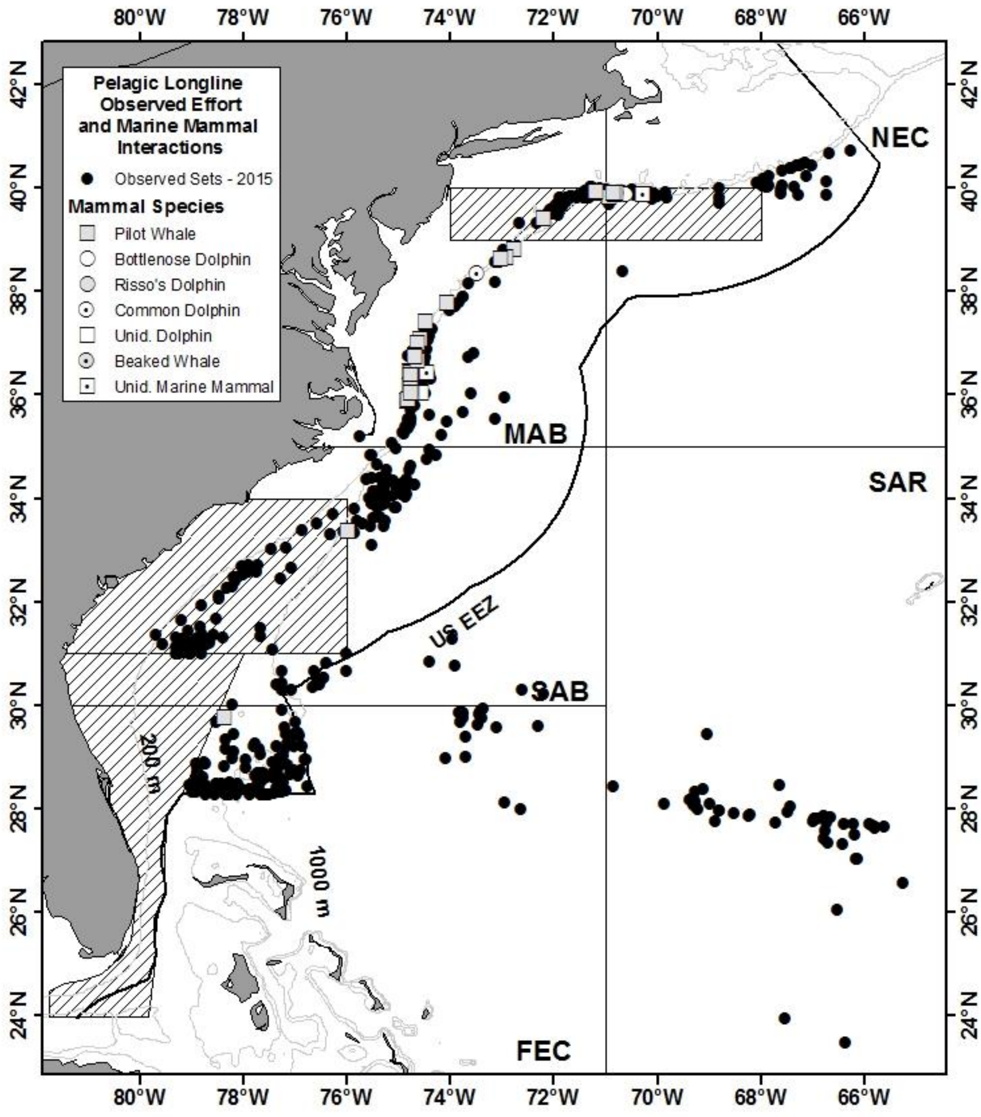


Figure 37. Observed sets and marine mammal interactions in the Pelagic Longline Fishery along the U.S. Atlantic coast during 2016. The boundaries of the Florida East Coast (FEC), South Atlantic Bight (SAB), Mid-Atlantic Bight (MAB), Northeast Coastal (NEC), and Sargasso Sea (SAR) fishing areas are shown. Seasonal closed areas instituted in 2001 under the HMS FMP are shown as hatched areas.

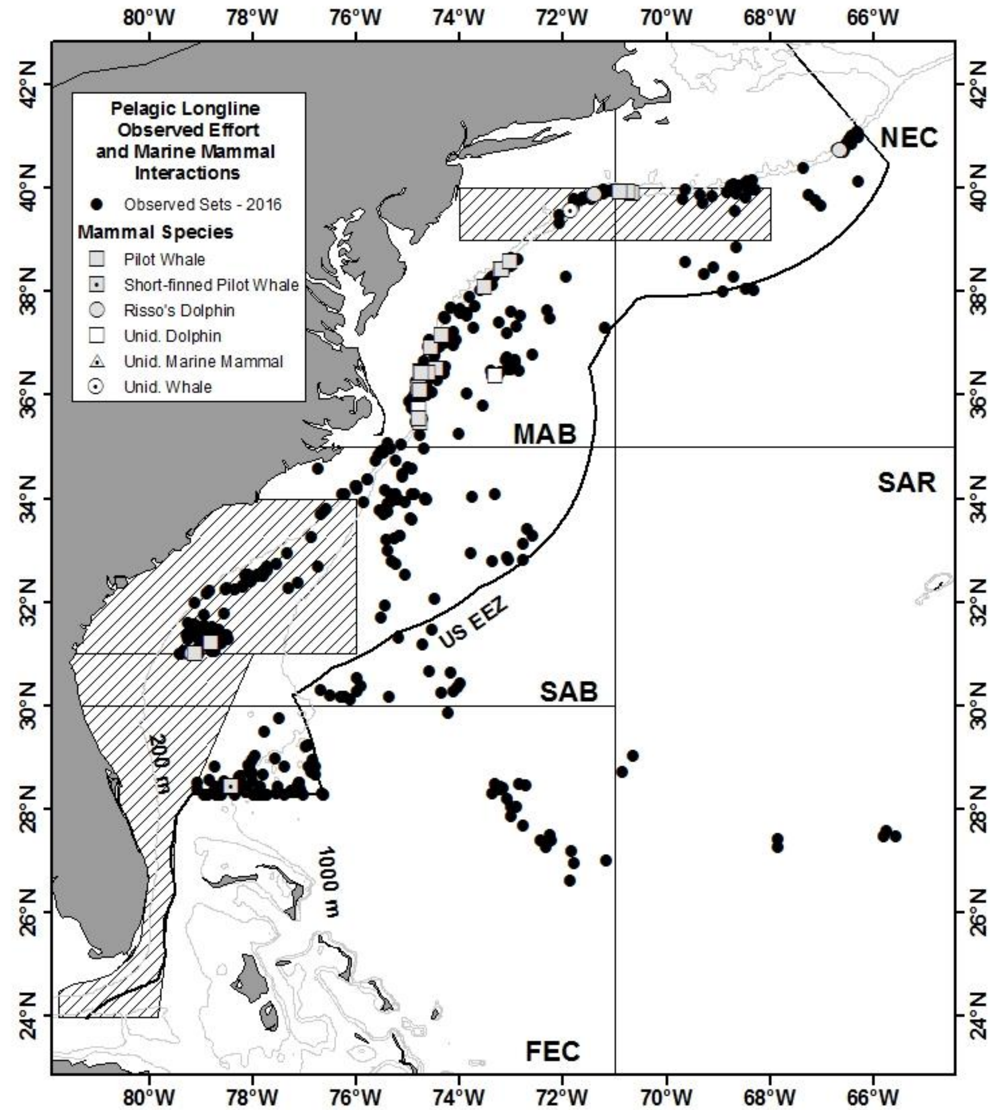


Figure 38. Observed sets and marine mammal interactions in the Pelagic Longline Fishery along the U.S. Atlantic coast during 2017. The boundaries of the Florida East Coast (FEC), South Atlantic Bight (SAB), Mid-Atlantic Bight (MAB), Northeast Coastal (NEC), and Sargasso Sea (SAR) fishing areas are shown. Seasonal closed areas instituted in 2001 under the HMS FMP are shown as hatched areas.

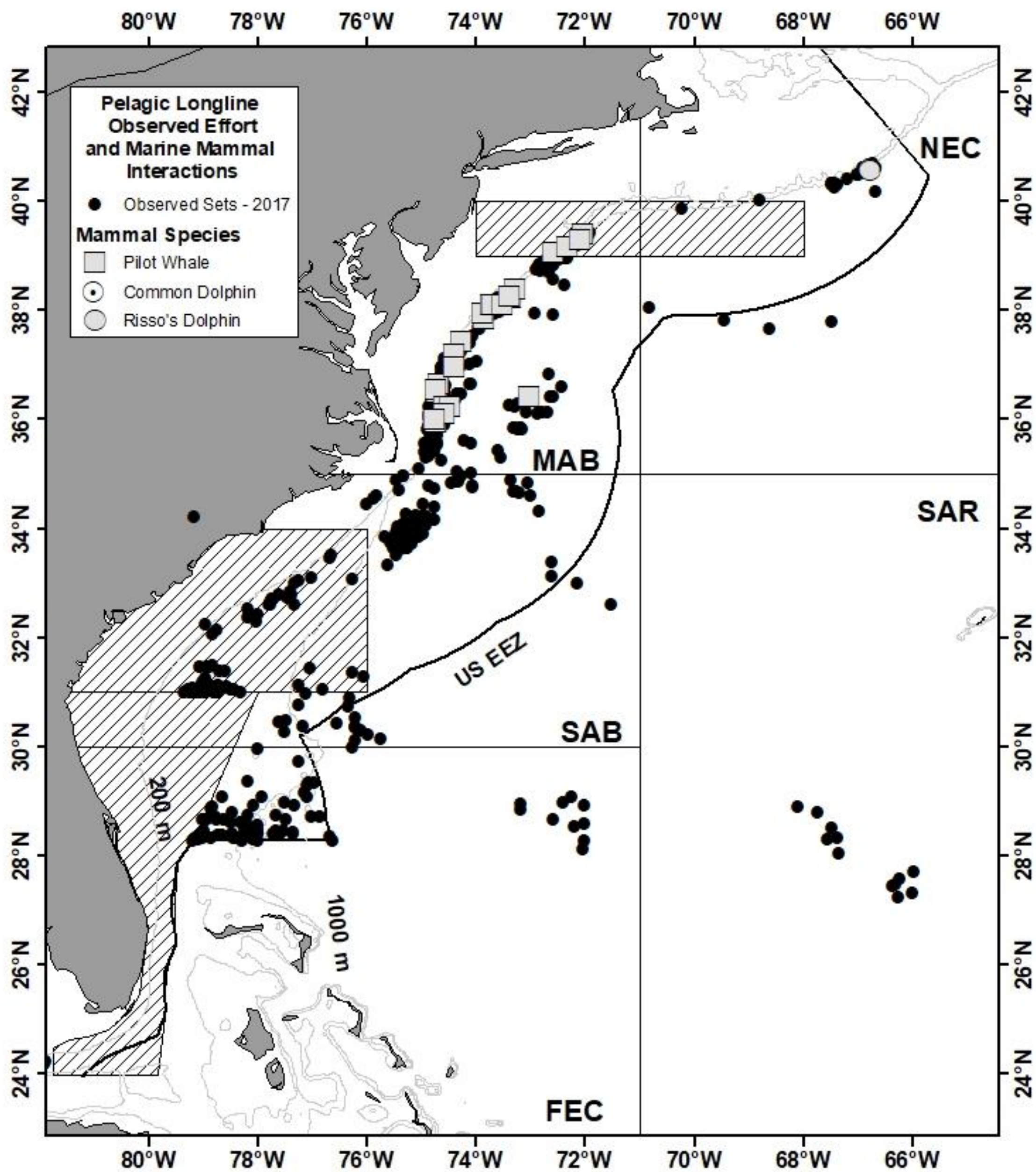


Figure 39. Observed sets and marine mammal interactions in the Pelagic Longline Fishery along the U.S. Atlantic coast during 2018. The boundaries of the Florida East Coast (FEC), South Atlantic Bight (SAB), Mid-Atlantic Bight (MAB), Northeast Coastal (NEC), and Sargasso Sea (SAR) fishing areas are shown. Seasonal closed areas instituted in 2001 under the HMS FMP are shown as hatched areas.

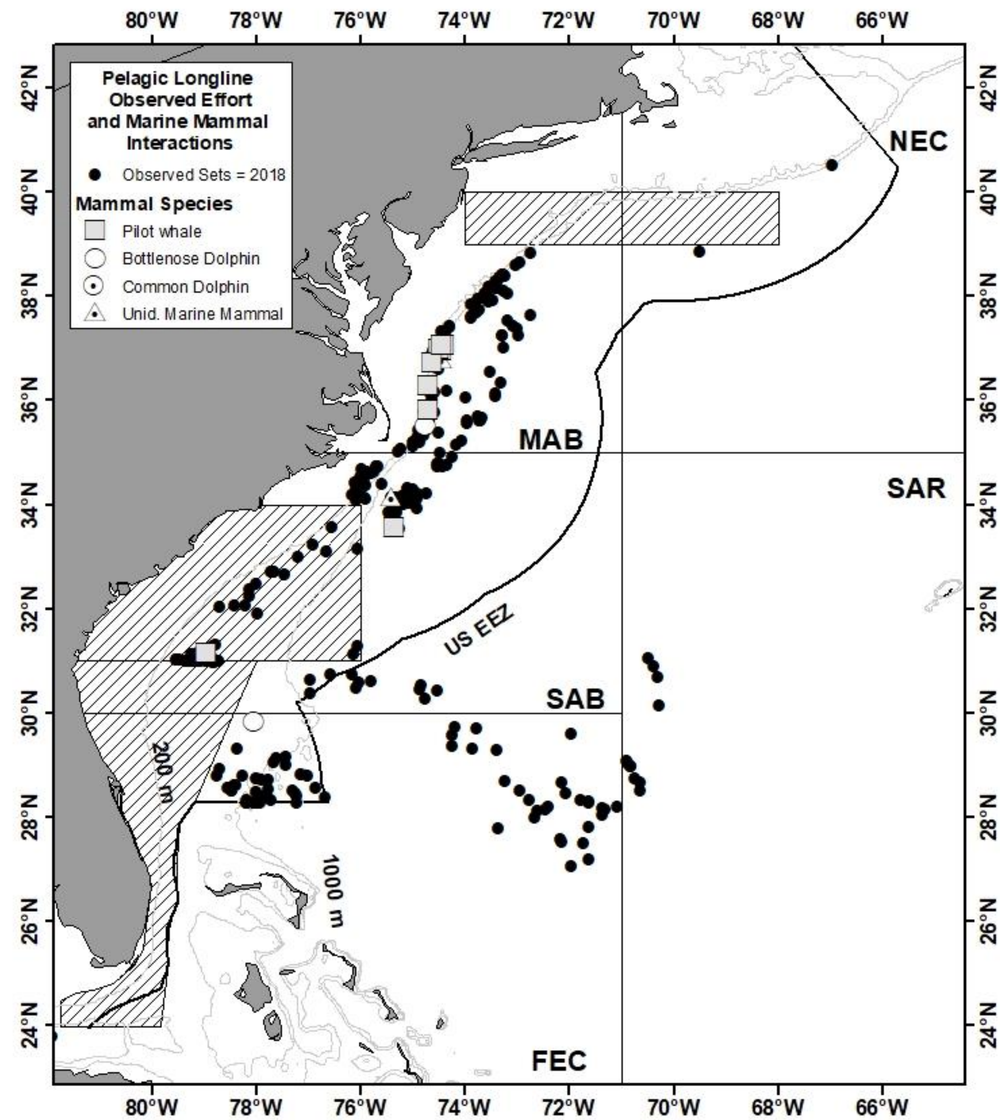


Figure 40. Observed sets and marine mammal interactions in the Pelagic Longline Fishery along the U.S. Atlantic coast during 2019. The boundaries of the Florida East Coast (FEC), South Atlantic Bight (SAB), Mid-Atlantic Bight (MAB), Northeast Coastal (NEC), and Sargasso Sea (SAR) fishing areas are shown. Seasonal closed areas instituted in 2001 under the HMS FMP are shown as hatched areas.

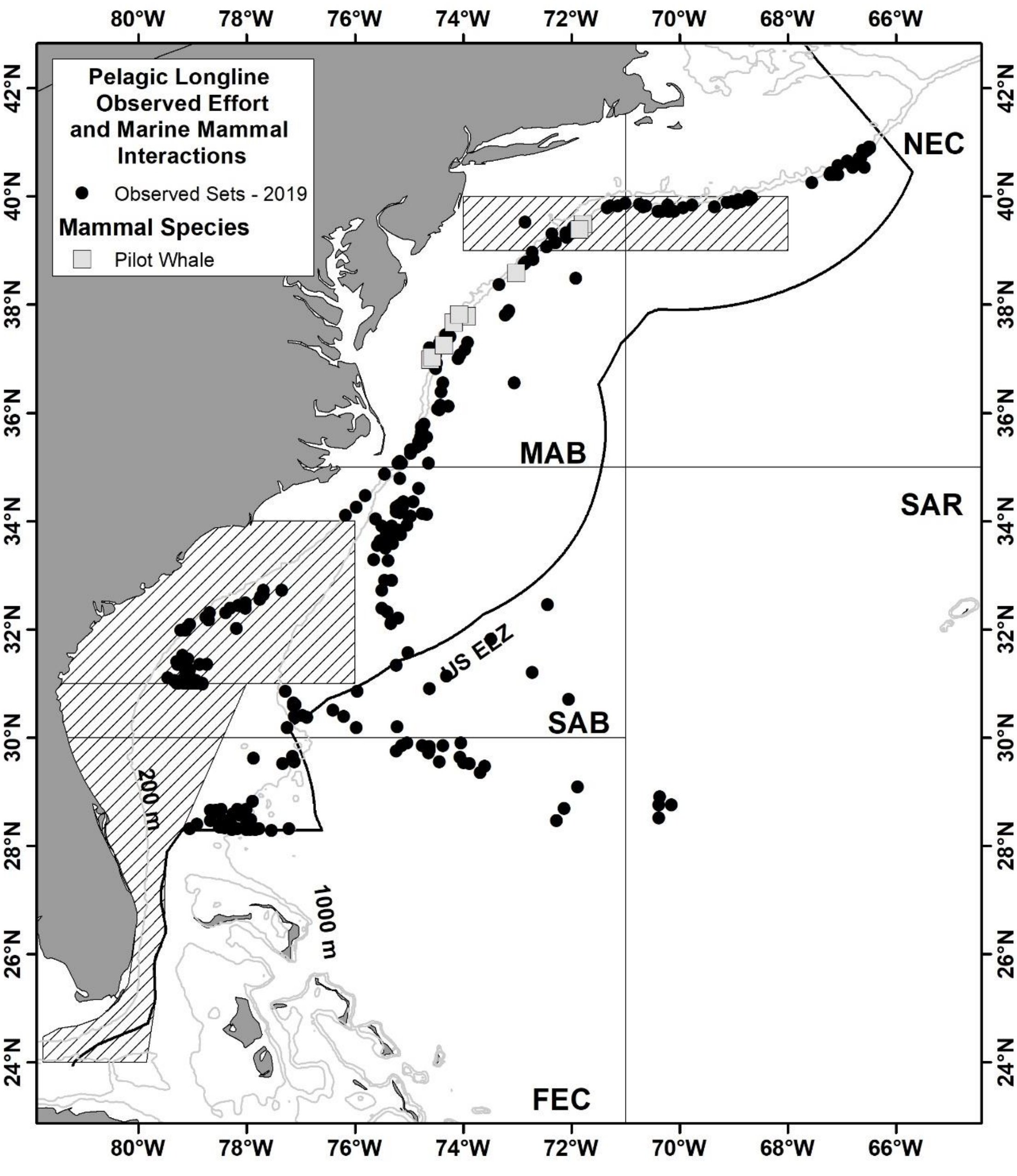


Figure 41. Observed sets in the Pelagic Longline Fishery in the Gulf of Mexico during 2015. Closed areas in the DeSoto Canyon instituted in 2001 are shown as hatched areas.

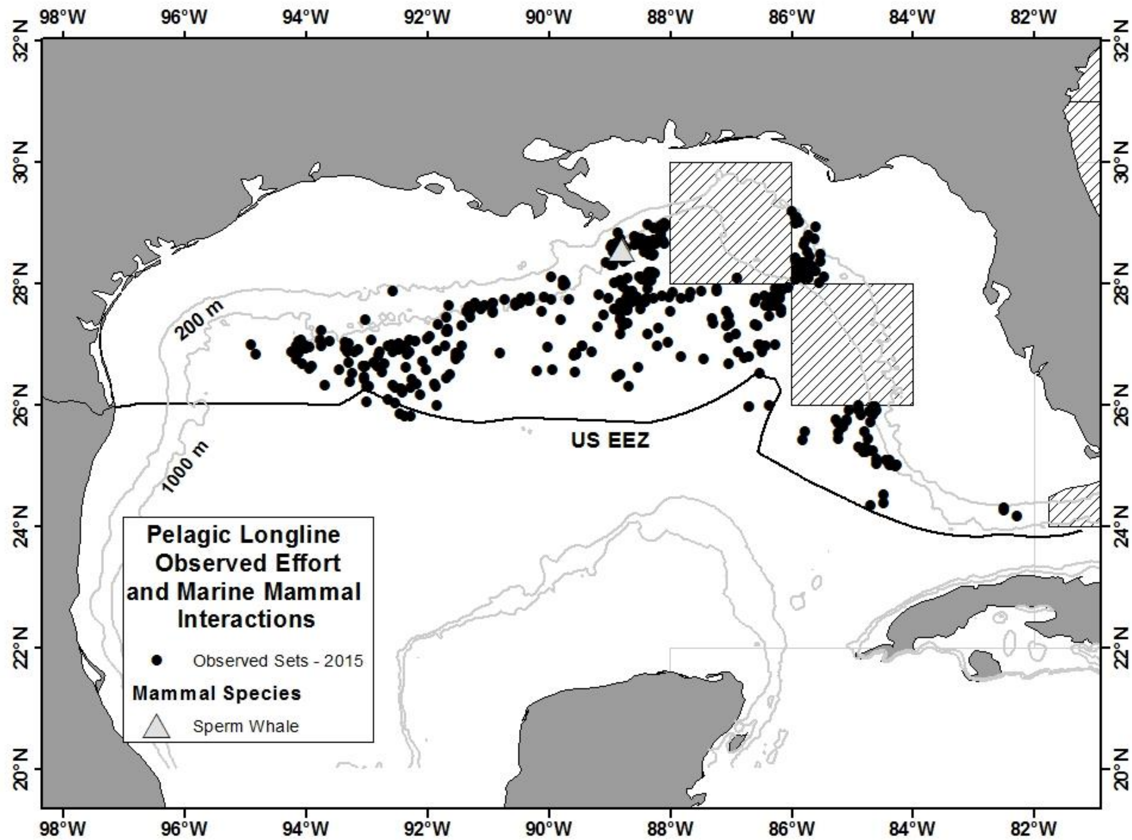


Figure 42. Observed sets in the Pelagic Longline Fishery in the Gulf of Mexico during 2016. Closed areas in the DeSoto Canyon instituted in 2001 are shown as hatched areas.

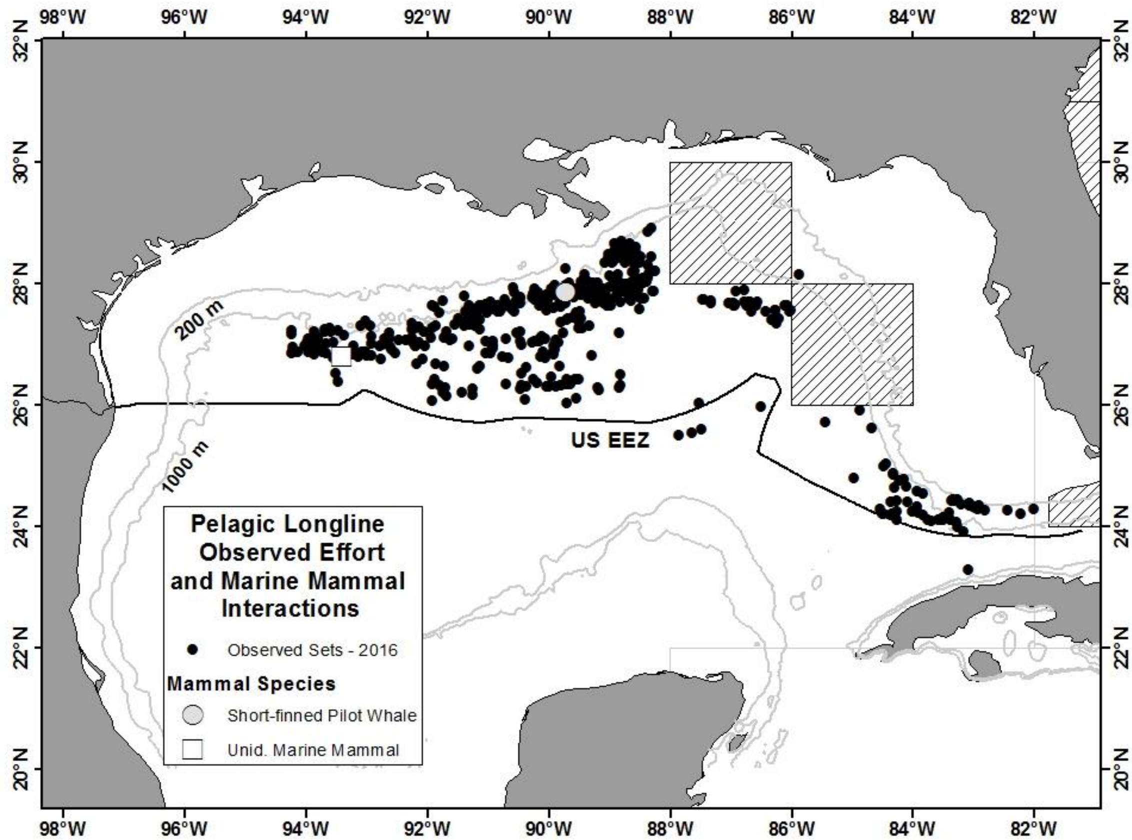


Figure 43. Observed sets in the Pelagic Longline Fishery in the Gulf of Mexico during 2017. Closed areas in the DeSoto Canyon instituted in 2001 are shown as hatched areas.

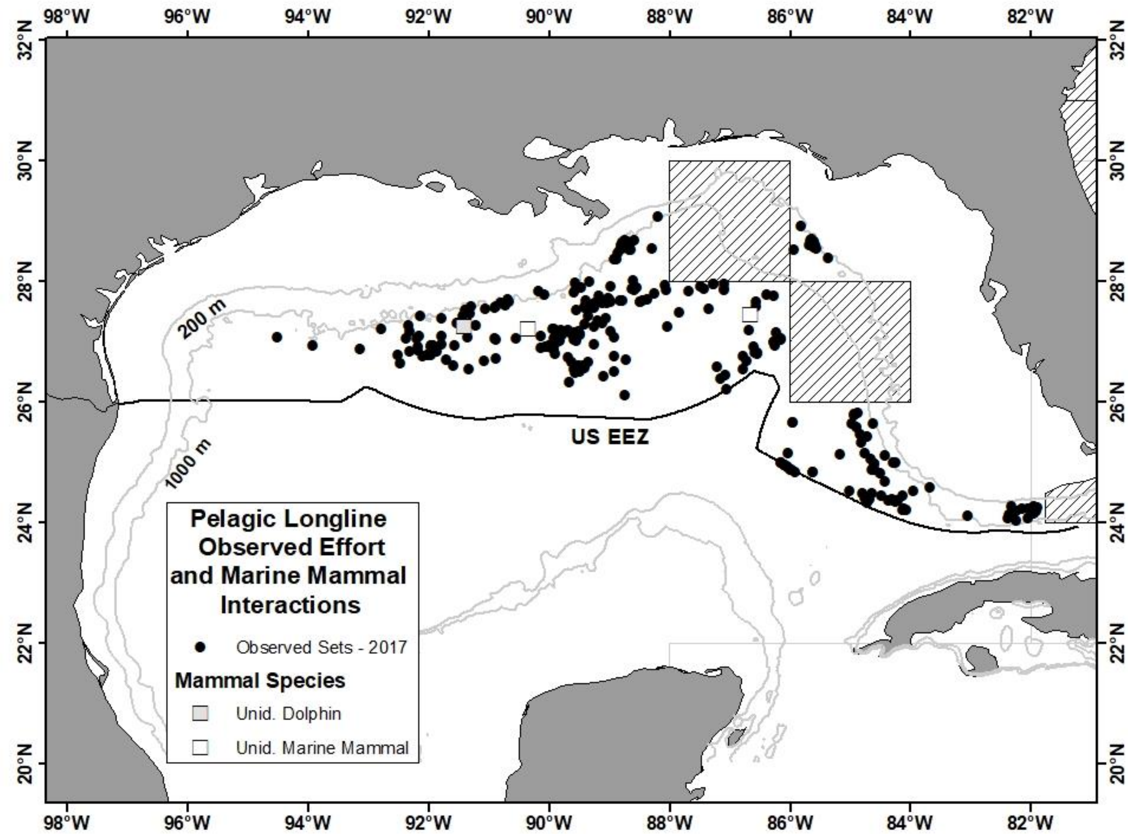


Figure 44. Observed sets in the Pelagic Longline Fishery in the Gulf of Mexico during 2018. Closed areas in the DeSoto Canyon instituted in 2001 are shown as hatched areas.

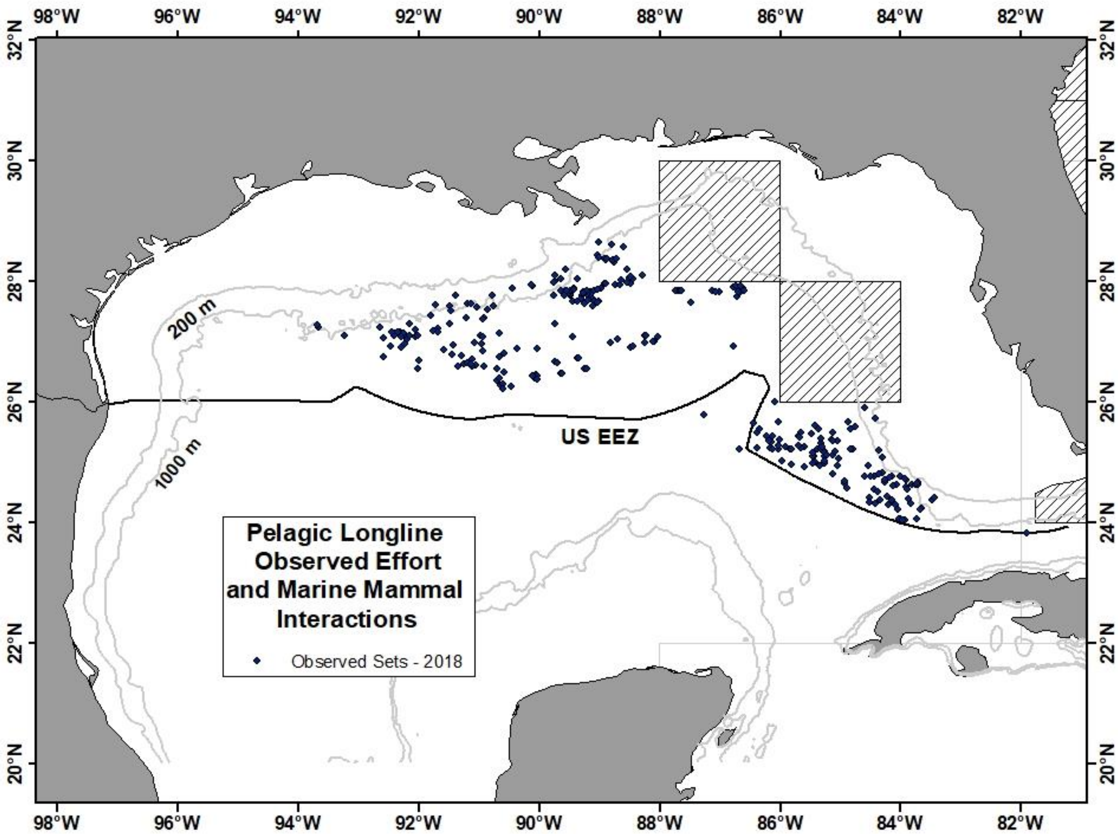
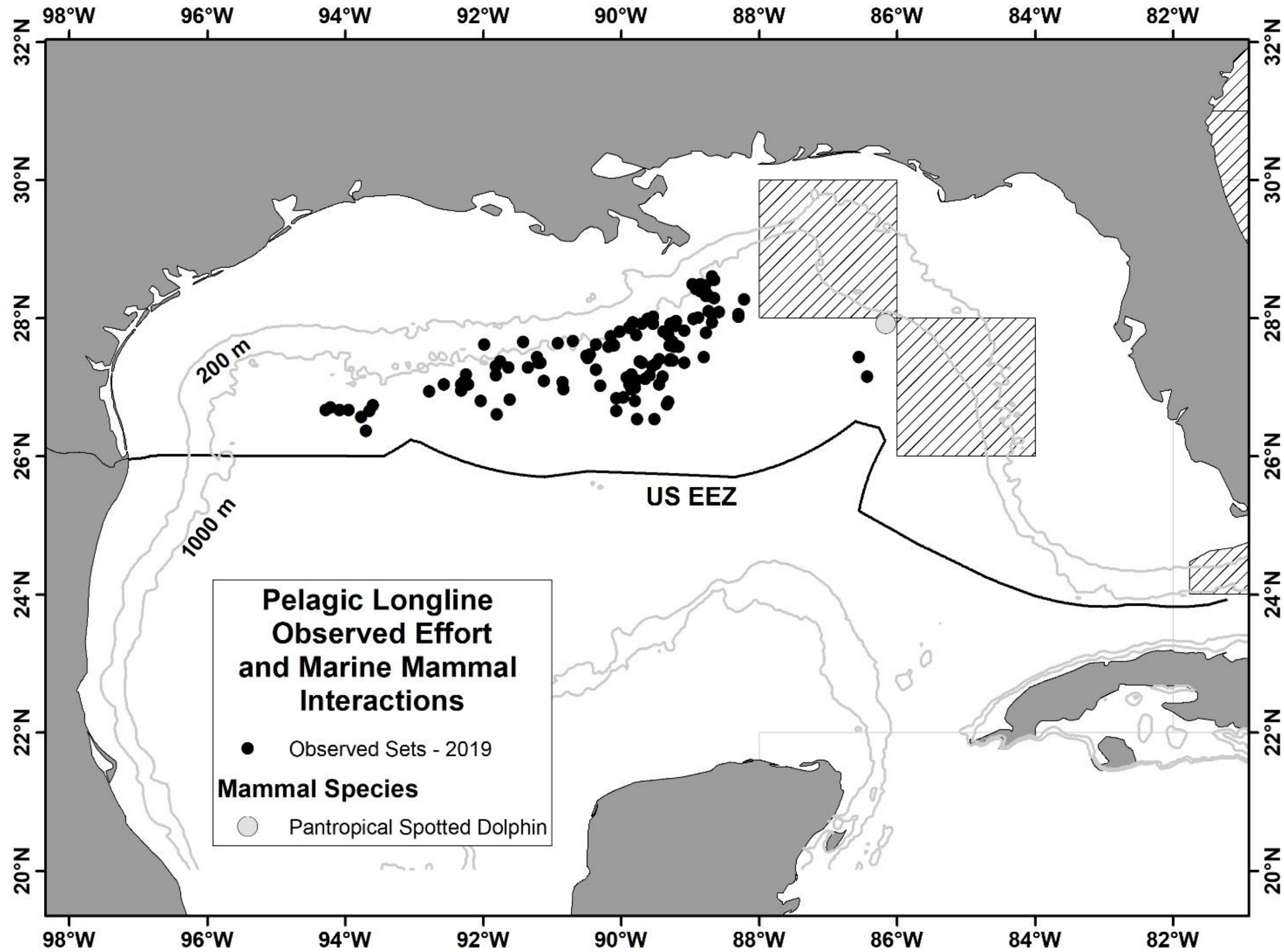


Figure 45. Observed sets in the Pelagic Longline Fishery in the Gulf of Mexico during 2019. Closed areas in the DeSoto Canyon instituted in 2001 are shown as hatched areas.



Appendix IV: Table A. Surveys.

Survey Number	Year(s)	Time of Year	Platform	Track Line Length (km)	Area	Agency/ Program	Analysis	Corrected for g(0)	Reference(s)
1	1982	year-round	Plane	211,585	Cape Hatteras, NC to Nova Scotia, (continental shelf & shelf edge waters)	CETAP	Line transect analyses of distance data	N	CETAP 1982
2	1990	Aug	Ship (Chapman)	2,067	Cape Hatteras, NC to Southern New England (north wall of Gulf Stream)	NEC	One team data analyzed by DISTANCE	N	NMFS 1990
3	1991	Jul–Aug	Ship (Abel-J)	1,962	Gulf of Maine, lower Bay of Fundy, southern Scotian Shelf	NEC	Two independent team data analyzed with modified direct duplicate method	Y	Palka 1995
4	1991	Aug	Boat (Sneak Attack)	640	Inshore bays of Maine	NEC	One team data analyzed by DISTANCE	Y	Palka 1995
5	1991	Aug–Sep	Plane 1 (AT-11)	9,663	Cape Hatteras, NC to Nova Scotia (continental shelf & shelf edge waters)	NEC/SEC	One team data analyzed by DISTANCE	N	NMFS 1991
6	1991	Aug–Sep	Plane 2 (Twin Otter)		Cape Hatteras, NC to Nova Scotia (continental shelf & shelf edge waters)	NEC/SEC	One team data analyzed by DISTANCE	N	NMFS 1991
7	1991	Jun–Jul	Ship (Chapman)	4,032	Cape Hatteras to Georges Bank, (between 200 & 2,000m isobaths)	NEC	One team data analyzed by DISTANCE	N	Waring *et al.* 1992; Waring 1998
8	1992	Jul–Sep	Ship (Abel-J)	3,710	N. Gulf of Maine & lower Bay of Fundy	NEC	Two independent team data analyzed with modified direct duplicate method	Y	Smith *et al.* 1993
9	1993	Jun–Jul	Ship (Delaware II)	1,874	S. edge of Georges Bank, across the Northeast Channel, to the SE edge of the Scotian Shelf	NEC	One team data analyzed by DISTANCE		NMFS 1993
10	1994	Aug–Sep	Ship (Relentless)	534	Georges Bank (shelf edge & slope waters)	NEC	One team data analyzed by DISTANCE	N	NMFS 1994
11	1995	Aug–Sep	Plane (Skymaster)	8,427	Gulf of St. Lawrence	DFO	One team data analyzed using Quenouille's Jackknife Bias Reduction Method that modeled the left truncated sighting curve	N	Kingsley and Reeves 1998

Survey Number	Year(s)	Time of Year	Platform	Track Line Length (km)	Area	Agency/ Program	Analysis	Corrected for g(0)	Reference(s)
12	1995	Jul–Sep	2 Ships (Abel-J & Pelican) & Plane (Twin Otter)	32,600	Virginia to the mouth of the Gulf of St. Lawrence	NEC	Ship: Two independent team data analyzed with modified direct duplicate method. Plane: One team data analyzed by DISTANCE.	Y/N	Palka 1996
13	1996	Jul–Aug	Plane	3,993	Northern Gulf of St. Lawrence	DFO	Quenouille's Jackknife Bias Reduction Method on line-transect methods that modeled the left truncated sighting curve	N	Kingsley and Reeves 1998
14	1998	Jul–Aug	Ship	4,163	South of Maryland	SEC	One team data analyzed by DISTANCE	N	Mullin and Fulling 2003
15	1998	Aug–Sep	Plane		Gulf of St. Lawrence	DFO			Kingsley and Reeves 1998
16	1998	Jul–Sep	Ship (Abel-J) & Plane (Twin Otter)	15,900	North of Maryland	NEC	Ship: Two independent team data analyzed with the modified direct duplicate or Palka & Hammond analysis methods, depending on the presence of responsive movement. Plane: One team data analyzed by DISTANCE.	Y	
17	1999	Jul–Aug	Ship (Abel-J) & Plane (Twin Otter)	6,123	South of Cape Cod to mouth of Gulf of St. Lawrence	NEC	Ship: Two independent team data analyzed with modified direct duplicate or Palka & Hammond analysis methods, depending on the presence of responsive movement. Plane: Circle-back data pooled with aerial data collected in 1999, 2002, 2004, 2006, 2007, and 2008 to calculate pooled g(0)'s and year-species specific abundance estimates for all years except 2008.	Y	
18	2002	Jul–Aug	Plane (Twin Otter)	7,465	Georges Bank to Maine	NEC	Same as for plane in survey 17	Y	Palka 2006
19	2002	Feb–Apr	Ship (Gunter)	4,592	SE US continental shelf - Delaware to Florida	SEC	One team data analyzed by DISTANCE	N	
20	2002	Jun–Jul	Plane	6,734	Florida to New Jersey	SEC	Two independent team data analyzed with modified direct duplicate method	Y	
21	2004	Jun–Aug	Ship (Gunter)	5,659	Florida to Maryland	SEC	Two independent team data analyzed with modified direct duplicate method	Y	Garrison *et al.* 2010
22	2004	Jun–Aug	Ship (Endeavor) & plane (Twin Otter)	10,761	Maryland to Bay of Fundy	NEC	Same methods used in survey 17	Y	Palka 2006
23	2006	Aug	Plane (Twin Otter)	10,676	Georges Bank to Bay of Fundy	NEC	Same as for plane in survey 17	Y	Palka 2005
24	2007	Aug	Ship (Bigelow) & Plane	8,195	Georges Bank to Bay of Fundy	NEC	Ship: Tracker data analyzed by DISTANCE. Plane: Same as for plane in survey 17	Y	Palka 2005

Survey Number	Year(s)	Time of Year	Platform	Track Line Length (km)	Area	Agency/ Program	Analysis	Corrected for g(0)	Reference(s)
			(Twin Otter)						
25	2007	Jul–Aug	Plane	46,804	Nova Scotia to Newfoundland	DFO	Uncorrected counts	N	Lawson and Gosselin 2009
26	2008	Aug	Plane (Twin Otter)	6,267	New York to Maine	NEC	Same as for plane in survey 17	Y	Palka 2005
27	2001	May–Jun	Plane		Maine Coast	NEC, UM	Corrected counts	N	Gilbert *et al.* 2005
28	1999	Mar	Plane		Cape Cod	NEC	Uncorrected counts	N	Barlas 1999
29	1983–1986	1983 (Fall), 1984 (Winter, Spring, Summer), 1985 (Summer, Fall), 1986 (Winter)	Plane (Beechcraft D-18S, modified with a bubblenose)	103,490	Northern Gulf of Mexico bays & sounds (coastal waters from shoreline to 18m isobath, & OCS waters from 18m isobath to 9.3km past the 18m isobath)	SEC	One team data analyzed with line-transect theory	N	Scott *et al.* 1989
30	1991–1994	Apr–Jun	Ship (Oregon II)	22,041	Northern Gulf of Mexico (from 200m to U.S. EEZ)	SEC	One team data analyzed by DISTANCE	N	Hansen *et al.* 1995
31	1992–1993	Sep–Oct	Plane (Twin Otter)		Northern Gulf of Mexico bays & sounds (coastal waters from shoreline to 18m isobath, & OCS waters from 18m isobath to 9.3km past the 18m isobath)	GOMEX92, GOMEX93	One team data analyzed by DISTANCE	N	Blaylock and Hoggard 1994
33	1996–1997, 1999–2001	Apr–Jun	Ship (Oregon II & Gunter)	12,162	Northern Gulf of Mexico (from 200m to U.S. EEZ)	SEC	One team data analyzed by DISTANCE	N	Mullin and Fulling 2004
34	1998–2001	End of Aug–Early Oct	Ship (Gunter & Oregon II)	2,196	Northern Gulf of Mexico (OCS waters from 20–200 m)	SEC	One team data analyzed by DISTANCE	N	Fulling *et al.* 2003
36	2004	12Jan–13 Jan	Helicopter		Sable Island	DFO	Pup count	na	Bowen *et al.* 2007
37	2004		Plane		Gulf of St Lawrence & Nova Scotia Eastern Shore	DFO	Pup count	na	Hammill 2005
38	2009	10Jun–13Aug	Ship	4,600	Northern Gulf of Mexico (from 200m to U.S. EEZ)	SEC	One team data analyzed by DISTANCE		
39	2007	17Jul–08Aug	Plane		Northern Gulf of Mexico (from shore to 200m, majority of effort 0–20m)	SEC	One team data analyzed by DISTANCE		

Survey Number	Year(s)	Time of Year	Platform	Track Line Length (km)	Area	Agency/ Program	Analysis	Corrected for g(0)	Reference(s)
40	2011	04Jun–01Aug	Ship (Bigelow)	3,107	Virginia to Massachusetts (waters that were deeper than the 100m depth contour out to beyond the US EEZ)	NEC	Two-independent teams, both using big-eyes. Analyzed using DISTANCE, the independent observer option assuming point independence	Y	Palka 2012
41	2011	07Aug–26Aug	Plane (Twin Otter)	5,313	Massachusetts to New Brunswick, Canada (waters north of New Jersey & shallower than the 100m depth contour, through the US & Canadian Gulf of Maine & up to & including the lower Bay of Fundy)	NEC	Two-independent teams, both using naked eye in the same plane. Analyzed using DISTANCE, the independent observer option assuming point independence.	Y	Palka 2012
42	2011	19Jun–01Aug	Ship (Gunter)	4,445	Florida to Virginia	SEC	Two-independent teams, both using naked eye in the same plane. Analyzed using DISTANCE, the independent observer option assuming point independence.	Y	Garrison 2016
43	2012	May–Jun	Plane		Maine Coast	NEC	Corrected counts	N	Waring *et al.* 2015
44	1992	Jan–Feb	Ship (Oregon II)	3,464	Cape Canaveral to Cape Hatteras, US EEZ	SEC		N	NMFS 1992
45	2010	24Jul–14Aug	Plane	7,944	Southeastern Florida to Cape May, New Jersey	SEC	Two-independent teams, both using naked eye in the same plane. Analyzed using DISTANCE, the independent observer option assuming point independence.		
46	2011	06Jul–29Jul	Plane	8,665	Southeastern Florida to Cape May, New Jersey	SEC	Two-independent teams, both using naked eye in the same plane. Analyzed using DISTANCE, the independent observer option assuming point independence.		Garrison 2016
47	2016	27Jun–25Aug	Ship & Plane	5,354	Central Virginia to the lower Bay of Fundy	NEC	Two-independent teams. Analyzed using DISTANCE, the independent observer option assuming point independence.		Palka 2020
48	2016	30Jun–19Aug	Ship & Plane	4,399	Central Florida to Virginia	SEC	Two-independent teams. Analyzed using DISTANCE, the independent observer option assuming point independence.		Garrison 2020

Survey Number	Year(s)	Time of Year	Platform	Track Line Length (km)	Area	Agency/ Program	Analysis	Corrected for g(0)	Reference(s)
49	2016	Aug & Sep	Plane	50,160	Gulf of St. Lawrence, Bay of Fundy, Scotian Shelf, Newfoundland, Labrador	DFO	NAISS		Lawson and Gosselin 2018
50	2017, 2018	02Jul–25Aug 2017, 11Aug–06Oct 2018	Ship (Gunter)	13,775	Northern Gulf of Mexico (waters from 200m to U.S. EEZ)	SEC	Two-independent teams. Analyzed using DISTANCE, the independent observer option assuming point independence.	Y	Garrison *et al.* 2020
51	2017, 2018	29Jun–17Aug 2017 18Jan–14Mar 2018 12Oct–28Nov 2018	Plane	14,590 km 8,046 km 10,781 km	Northern Gulf of Mexico (from shore to 200m, majority of effort 0–20m)	SEC	Two-independent teams, both using naked eye in the same plane. Analyzed using DISTANCE, the independent observer option assuming point independence.	Y	Garrison *et al.* 2021

Appendix IV: Table B. Abundance Estimates.

"Survey Number" refers to surveys described in Table A. "Best" estimate for each species is in bold font.

Species	Stock	Year	Nest	CV	Survey Number	Notes
Humpback Whale	Gulf of Maine	1992	501			Minimum population size estimated from photo-ID data
		1993	652	0.29		YONAH sampling (Clapham *et al.* 2003)
		1997	497			Minimum population size estimated from photo-ID data
		1999	902	0.45	17	
		2002	521	0.67	18	Palka 2006
		2004	359	0.75	22	Palka 2006
		2006	847	0.55	23	Palka 2005
		2008	823			Mark-recapture estimate (Robbins 2010)
		2011	335	0.42	40+41	Palka 2012
		2015	896			Minimum population size estimated from photo-ID data
		2016	2,368			
		2016	**1,396**	**na**		State-space mark-recapture (Pace 2017)
Fin Whale	Western North Atlantic	1995	2,200	0.24	12	Palka 1996
		1999	2,814	0.21	18	Palka 2006
		2002	2,933	0.49	18	Palka 2006
		2004	1,925	0.55	22	Palka 2006
		2006	2,269	0.37	23	Palka 2005
		2007	3,522	0.27	25	Lawson and Gosselin 2009
		2011	1,595	0.33	40+41	Palka 2012

Species	Stock	Year	Nest	CV	Survey Number	Notes
		2011	23	0.87	42	
		2011	1,618	0.33	40+41+42	Estimate summed from north and south surveys
		2016	3,006	0.40	47+48	Garrison 2020; Palka 2020
		2016	2,235	0.41	49	Bay of Fundy/Scotian Shelf (Lawson and Gosselin 2018)
		2016	2,177	0.47	49	Newfoundland/Labrador (Lawson and Gosselin 2018)
		2016	**7,418**	**0.25**	**47+48+49**	
Sei Whale	Nova Scotia Stock	1977	1,393–2,248			Based on tag-recapture data (Mitchell and Chapman 1977)
		1977	870			Based on census data (Mitchell and Chapman 1977)
		1982	280		1	CETAP 1982
		2002	71	1.01	18	Palka 2006
		2004	386	0.85	22	Palka 2006
		2006	207	0.62	23	Palka 2005
		2011	357	0.52	40+41	Palka 2012
		2010–2013	**6,292**	**1.02**		Springtime average abundance estimate generated from spatially- and temporally-explicit density models derived from visual two-team abundance survey data collected between 2010 and 2013 (Palka *et al*. 2017)
		1999–2013	627	0.14		Spring habitat-based density estimates (Roberts *et al*. 2016)
		1995–2013	717	0.30		Summer habitat-based density estimates (Roberts *et al*. 2016)
		2016	28	0.55	47	Palka 2016
Minke Whale	Canadian East Coast	1982	320	0.23	1	CETAP 1982
		1992	2,650	0.31	3+8	
		1993	330	0.66	9	
		1995	2,790	0.32	12	Palka 1996
		1995	1,020	0.27	11	
		1996	620	0.52	13	
		1999	2,998	0.19	17	
		2002	756	0.9	18	Palka 2006
		2004	600	0.61	22	Palka 2006
		2006	3,312	0.74	23	
		2007	20,741	0.3	25	Lawson and Gosselin 2009
		2011	2,591	0.81	40+41	Palka 2012
		2016	5,036	0.68	47	Palka 2020
		2016	6,158	0.40	49	Bay of Fundy/Scotian Shelf (Lawson and Gosselin 2018)
		2016	13,008	0.46	49	Newfoundland/Labrador (Lawson and Gosselin 2018)
		2016	**24,202**	**0.30**	**47+49**	
Sperm Whale	North Atlantic	1982	219	0.36	1	CETAP 1982
		1990	338	0.31	2	
		1991	736	0.33	7	Waring *et al*.1992, Warring 1998

Species	Stock	Year	Nest	CV	Survey Number	Notes
		1991	705	0.66	6	
		1991	337	0.5	5	
		1993	116	0.4	9	
		1994	623	0.52	10	
		1995	2,698	0.67	12	Palka 1996
		1998	2,848	0.49	16	
		1998	1,181	0.51	14	Mullin and Fulling 2003
		2004	2,607	0.57	22	Palka 2006
		2004	2,197	0.47	21	Garrison *et al.* 2010
		2004	4,804	0.38	21+22	Estimate summed from north and south surveys
		2011	1,593	0.36	40+41	Palka 2012
		2011	695	0.39	42	
		2011	2,288	0.28	40+41+42	Estimate summed from north and south surveys
		2016	3,321	0.35	47	Palka 2020
		2016	1,028	0.35	48	Garrison 2020
		2016	**4,349**	**0.28**	**47+48**	Estimate summed from north and south surveys
***Kogia* spp.**	Western North Atlantic	1998	115	0.61	16	
		1998	580	0.57	14	Mullin and Fulling 2003
		2004	358	0.44	22	Palka 2006
		2004	37	0.75	21	Garrison *et al.* 2010
		2004	395	0.4	21+22	Estimate summed from north and south surveys
		2011	1,783	0.62	40+41	Palka 2012
		2011	2,002	0.69	42	
		2011	3,785	0.47	40+41+42	Estimate summed from north and south surveys
		2016	4,548	0.49	47	Palka 2020
		2016	3,202	0.59	48	Garrison 2020
		2016	7,750	0.38	47+48	Estimate summed from north and south surveys
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, Warring 1998
		1991	370	0.65	6	
		1991	612	0.73	5	
		1993	330	0.66	9	
		1994	99	0.64	10	
		1995	1,519	0.69	12	Palka 1996
		1998	2,600	0.4	16	
		1998	541	0.55	14	Mullin and Fulling 2003
		2004	2,839	0.78	22	Palka 2006

Species	Stock	Year	Nest	CV	Survey Number	Notes
		2004	674	0.36	21	Garrison *et al.* 2010
		2004	3,513	0.63	21+22	Estimate summed from north and south surveys
		2006	922	1.47	23	
		2011	5,500	0.67	40+41	2011 estimates are for *Mesoplodon* spp. beaked whales alone (not including *Ziphias*; Palka 2012)
		2011	1,592	0.67	42	2011 estimates are for *Mesoplodon* spp. beaked whales alone (not including *Ziphias*)
		2011	7,092	0.54	40+41+42	2011 estimates are for *Mesoplodon* spp. beaked whales alone (not including *Ziphias*); Estimate summed from north and south surveys
		2016	6,760	0.37	47	Palka 2020
		2016	3,347	0.29	48	Garrison 2020
		2016	10,107	0.27	47+48	Estimate summed from north and south surveys
Cuvier's Beaked Whale	Western North Atlantic	2011	4,962	0.37	40+41	Palka 2012
		2011	1,570	0.65	42	
		2011	6,532	0.32	40+41+42	Estimate summed from north and south surveys
		2016	3,897	0.47	47	Palka 2020
		2016	1,847	0.49	48	Garrison 2020
		2016	**5,744**	**0.36**	**47+48**	Estimate summed from north and south surveys
Risso's Dolphin	Western North Atlantic	1982	4,980	0.34	1	CETAP 1982
		1991	11,017	0.58	7	Waring *et al.*1992, Warring 1998
		1991	6,496	0.74	5	
		1991	16,818	0.52	6	
		1993	212	0.62	9	
		1995	5,587	1.16	12	Palka 1996
		1998	18,631	0.35	17	
		1998	9,533	0.5	15	
		1998	28,164	0.29	15+17	Estimate summed from north and south surveys
		2002	69,311	0.76	18	Palka 2006
		2004	15,053	0.78	21	Garrison *et al.* 2010
		2004	5,426	0.54	22	Palka 2006
		2004	20,479	0.59	21+22	Estimate summed from north and south surveys
		2006	14,408	0.38	23	
		2011	15,197	0.55	40+41	Palka 2012
		2011	3,053	0.44	42	
		2011	18,250	0.46	40+41+42	Estimate summed from north and south surveys
		2016	7,245	0.44	48	Garrison 2020
		2016	22,175	0.23	47	Palka 2020
		2016	6,073	0.45	49	Lawson and Gosselin 2018
		2016	**35,493**	**0.19**	**47+48+49**	

Species	Stock	Year	Nest	CV	Survey Number	Notes
Pilot Whale	Western North Atlantic	1951	50,000			Derived from catch data from 1951–1961 drive fishery (Mitchell 1974)
		1975	43,000–96,000			Derived from population models (Mercer 1975)
		1982	11,120	0.29	1	CETAP 1982
		1991	3,636	0.36	7	Waring *et al.*1992, Warring 1998
		1991	3,368	0.28	5	
		1991	5,377	0.53	6	
		1993	668	0.55	9	
		1995	8,176	0.65	12	Palka 1996
		1995	9,776	0.55	12+16	Sum of US (#12) and Canadian (#16) surveys
		1998	1,600	0.65	16	
		1998	9,800	0.34	17	
		1998	5,109	0.41	15	
		2002	5,408	0.56	18	Palka 2006
		2004	15,728	0.34	22	Palka 2006
		2004	15,411	0.43	21	Garrison *et al.* 2010
		2004	31,139	0.27	21+22	Estimate summed from north and south surveys
		2006	26,535	0.35	23	Estimate summed from north and south surveys
		2007	16,058	0.79	25	Long-finned pilot whales (Lawson and Gosselin 2009)
		2011	5,636	0.63	40+41	Long-finned pilot whales
		2011	11,865	0.57	40+41	Unidentified pilot whales
		2011	4,569	0.57	40+41	Short-finned pilot whales
		2011	16,946	0.43	42	Short-finned pilot whales
		2011	21,515	0.37	40+41+42	Best estimate for short-finned pilot whales alone; Estimate summed from north and south surveys
		2016	3,810	0.42	47	Short-finned pilot whales (Garrison and Palka 2018)
		2016	25,114	0.27	48	Short-finned pilot whales (Garrison and Palka 2018)
		2016	**28,924**	**0.24**	**47+48**	Best estimate for short-finned pilot whales alone; Estimate summed from north and south surveys
		2016	10,997	0.51	47	Long-finned pilot whales (Garrison 2020; Palka 2020)
		2016	28,218	0.36	48	Long-finned pilot whales (Garrison 2020; Palka 2020)
		2016	**39,215**	**0.30**	**47+48**	Best estimate for long-finned pilot whales alone; Estimate summed from north and south surveys
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	

Species	Stock	Year	Nest	CV	Survey Number	Notes
		1999	51,640	0.38	17	
		2002	109,141	0.3	18	Palka 2006
		2004	2,330	0.8	22	Palka 2006
		2006	17,594	0.3	23	
		2006	63,368	0.27	(18+23)/2	Average of #18 and #23
		2007	5,796	0.43	25	Lawson and Gosselin 2009
		2011	48,819	0.61	40+41	Palka 2012
		2016	31,912	0.61	47	Palka 2020
		2016	61,321	1.04	49	Canadian part of Gulf of Maine and all of Gulf of St. Lawrence population (Lawson and Gosselin 2018)
		2016	**93,233**	**0.71**	**47+49**	
White-beaked Dolphin	Western North Atlantic	1982	573	0.69	1	CETAP 1982
			5,500			Alling and Whitehead 1987
		1982	3,486	0.22		Alling and Whitehead 1987
		2006	2,003	0.94	23	
		2007	11,842		25	
		2008			26	
		2016	**536,016**	**0.31**	**49**	Lawson and Gosselin 2018
Common Dolphin	Western North Atlantic	1982	29,610	0.39	1	
		1991	22,215	0.4	7	Waring *et al.*1992; Warring 1998
		1993	1,645	0.47	9	
		1995	6,741	0.69	12	Palka 1996
		1998	30,768	0.32	17	
		1998	0		15	
		2002	6,460	0.74	18	
		2004	90,547	0.24	22	Palka 2006
		2004	30,196	0.54	21	Garrison *et al.* 2010
		2004	120,743	0.23	21+22	Estimate summed from north and south surveys
		2006	84,000	0.36	24	
		2007	173,486	0.55	25	Lawson and Gosselin 2009
		2011	67,191	0.29	40+41	Palka 2012
		2011	2,993	0.87	42	
		2011	70,184	0.28	40+41+42	Estimate summed from north and south surveys
		2016	80,227	0.31	47	Palka 2020
		2016	900	0.57	48	Garrison 2020
		2016	48,574	0.48	49	Newfoundland/Labrador (Lawson and Gosselin 2018)
		2016	43,124	0.28	49	Bay of Fundy/Scotian Shelf (Lawson and Gosselin 2018)
		2016	**172,825**	**0.21**	**47+48+49**	Estimate summed from north, south and Canadian surveys

Species	Stock	Year	Nest	CV	Survey Number	Notes
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
		2016	8,247	0.24	47	Palka 2020
		2016	31,674	0.33	48	Garrison 2020
		2016	39,921	0.27	47+48	Estimate summed from north and south surveys
Pantropical Spotted Dolphin	Western North Atlantic	1982	6,107	0.27	1	CETAP 1982
		1995	4,772	1.27	12	Palka 1996
		1998	343	1.03	16	
		1998	12,747	0.56	14	Mullin and Fulling 2003
		2004	0		22	Palka 2006
		2004	4,439	0.49	21	Garrison *et al.* 2010
		2004	4,439	0.49	21+22	Estimate summed from north and south surveys
		2011	0	0	40+41	Palka 2012
		2011	3,333	0.91	42	
		2011	3,333	0.91	40+41+42	Estimate summed from north and south surveys
		2016	0	-	47	Palka 2020
		2016	6,593	0.52	48	Garrison 2020
		2016	6,593	0.52	47+48	Estimate summed from north and south surveys
Striped Dolphin	Western North Atlantic	1982	36,780	0.27	1	
		1995	31,669	0.73	12	Palka 1996
		1998	39,720	0.45	16	
		1998	10,225	0.91	14	Mullin and Fulling 2003
		2004	52,055	0.57	22	
		2004	42,407	0.53	21	Garrison *et al.* 2010
		2004	94,462	0.4	21+22	Estimate summed from north and south surveys
		2011	46,882	0.33	40+41	Palka 2012
		2011	7,925	0.66	42	
		2011	54,807	0.3	40+41+42	Estimate summed from north and south surveys
		2016	42,783	0.25	47	Palka 2020
		2016	24,163	0.66	48	Garrison 2020
		2016	67,036	0.29	47+48	Estimate summed from north and south surveys

Species	Stock	Year	Nest	CV	Survey Number	Notes
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
		2016	17,958	0.33	47	Palka 2020
		2016	44,893	0.29	48	Garrison 2020
		2016	62,851	0.23	47+48	Estimate summed from north and south surveys
Harbor Porpoise	Gulf of Maine, Bay of Fundy	1991	37,500	0.29	3	Palka 1995
		1992	67,500	0.23	8	Smith *et al.* 1993
		1995	74,000	0.2	12	Palka 1996
		1995	12,100	0.26	11	
		1996	21,700	0.38	14	Mullin and Fulling 2003
		1999	89,700	0.22	17	Survey discovered portions of the range not previously surveyed (Palka 2006)
		2002	64,047	0.48	21	Palka 2006
		2004	51,520	0.65	23	Palka 2006
		2006	89,054	0.47	24	
		2007	4,862	0.31	25	Lawson and Gosselin 2009
		2011	79,883	0.32	40+41	Palka 2012
		2016	75,079	0.38	47	Palka 2020
		2016	20,464	0.39	48	Garrison 2020
		2016	**95,543**	**0.31**	**47+48**	Estimate summed from north and south surveys
Harbor Seal	Western North Atlantic	2001	99,340	0.097	27	Gilbert *et al.* 2005
		2012	**75,834**	**0.15**	**43**	Waring *et al.* 2015
Gray Seal	Western North Atlantic	1999	5,611		28	Barlas 1999
		2001	1,731		27	Gilbert *et al.* 2005
		2004	52,500	0.15	37	Gulf of St Lawrence and Nova Scotia Eastern Shore
		2004	208,720–223,220	0.08–0.14	36	Sable Island
		2012	331,000	95%CI= 263,000-458,000		Gulf of St Lawrence + Nova Scotia Eastern Shore + Sable Island (DFO 2013)

Species	Stock	Year	Nest	CV	Survey Number	Notes
		2014	505,000	95%CI= 329,000–682,000		Gulf of St Lawrence + Nova Scotia Eastern Shore + Sable Island (DFO 2014)
		2016	424,300	95%CI= 263,600–578,300		Gulf of St Lawrence + Nova Scotia Eastern Shore + Sable Island (DFO 2017)
		2016	**27,131**	**95%CI= 18,768–39,221**		Derived from total population size to pup ratios in Canada applied to U.S. pup counts
Bryde's Whale	Northern Gulf of Mexico	1991–1994	35	1.1	30	Hansen *et al.* 1995
		1996–2001	40	0.61	33	Mullin and Fulling 2004
		2003–2004	15	1.98	35	
		2009	33	1.07	38	
		2017–2018	51	0.50	50	Garrison *et al.* 2020a
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	
		2017–2018	1,307	0.33	50	Garrison *et al.* 2020a
***Kogia* spp.**	Northern Gulf of Mexico	1991–1994	547	0.28	30	Hansen *et al.* 1995
		1996–2001	742	0.29	33	Mullin and Fulling 2004
		2003–2004	453	0.35	35	
		2009	186	1.04	38	
		2017–2018	336	0.35	50	Garrison *et al.* 2020a
Cuvier's Beaked Whale	Northern Gulf of Mexico	1991–1994	30	0.5	30	Hansen *et al.* 1995
		1996–2001	95	0.47	33	Mullin and Fulling 2004
		2003–2004	65	0.67	35	
		2009	74	1.04	38	
		2017–2018	18	0.75	50	Garrison *et al.* 2020a
***Mesoplodon* spp.**	Northern Gulf of Mexico	1996–2001	106	0.41	33	Mullin and Fulling 2004
		2003–2004	57	1.4	35	
		2009	149	0.91	38	
		2017–2018	98	0.46	50	Garrison *et al.* 2020a
Killer Whale	Northern Gulf of Mexico	1991–1994	277	0.42	30	Hansen *et al.* 1995
		1996–2001	133	0.49	33	Mullin and Fulling 2004
		2003–2004	49	0.77	35	
		2009	28	1.02	38	
		2017–2018	267	0.75	50	Garrison *et al.* 2020a
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

Species	Stock	Year	Nest	CV	Survey Number	Notes
		2003–2004	777	0.56	35	
		2017–2018	494	0.79	50	Garrison *et al.* 2020a
Short-finned Pilot Whale	Northern Gulf of Mexico	1991–1994	353	0.89	30	Hansen *et al.* 1995
		1996–2001	2,388	0.48	33	Mullin and Fulling 2004
		2003–2004	716	0.34	35	
		2009	2,415	0.66	38	
		2017–2018	1,321	0.43	50	Garrison *et al.* 2020a
Melon-headed Whale	Northern Gulf of Mexico	1991–1994	3,965	0.39	30	Hansen *et al.* 1995
		1996–2001	3,451	0.55	33	
		2003–2004	2,283	0.76	35	
		2009	2,235	0.75	38	
		2017–2018	1,749	0.68	50	Garrison *et al.* 2020a
Pygmy Killer Whale	Northern Gulf of Mexico	1991–1994	518	0.81	30	Hansen *et al.* 1995
		1996–2001	408	0.6	33	Mullin and Fulling 2004
		2003–2004	323	0.6	35	
		2009	152	1.02	38	
		2017–2018	613	1.15	50	Garrison *et al.* 2020a
Risso's Dolphin	Northern Gulf of Mexico	1991–1994	2,749	0.27	30	Hansen *et al.* 1995
		1996–2001	2,169	0.32	33	Mullin and Fulling 2004
		2003–2004	1,589	0.27	35	
		2009	2,442	0.57	38	
		2017–2018	1,974	0.46	50	Garrison *et al.* 2020a
Pantropical Spotted Dolphin	Northern Gulf of Mexico	1991–1994	31,320	0.2	30	Hansen *et al.* 1995
		1996–2001	91,321	0.16	33	Mullin and Fulling 2004
		2003–2004	34,067	0.18	35	
		2009	50,880	0.27	38	
		2017–2018	37,195	0.24	50	Garrison *et al.* 2020a
Striped Dolphin	Northern Gulf of Mexico	1991–1994	4,858	0.44	30	Hansen *et al.* 1995
		1996–2001	6,505	0.43	33	Mullin and Fulling 2004
		2003–2004	3,325	0.48	35	
		2009	1,849	0.77	38	
		2017–2018	1,817	0.56	50	Garrison *et al.* 2020a
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	

Species	Stock	Year	Nest	CV	Survey Number	Notes
		2017–2018	2,991	0.54	50	Garrison *et al.* 2020a
Clymene Dolphin	Northern Gulf of Mexico	1991–1994	5,571	0.37	30	Hansen *et al.* 1995
		1996–2001	17,355	0.65	33	Mullin and Fulling 2004
		2003–2004	6,575	0.36	35	
		2009	129	1	38	
		2017–2018	513	1.03	50	Garrison *et al.* 2020a
Atlantic Spotted Dolphin	Northern Gulf of Mexico	Oceanic (1991–1994)	3,213	0.44	30	Hansen *et al.* 1995
		Oceanic (1996–2001)	175	0.84	33	Mullin and Fulling 2004
		OCS (1998–2001)	37,611	0.28	34	Abundance estimate is from 2000-2001 surveys only (from Fulling *et al.* 2003). Current best population size estimate is unknown because data from the continental shelf portion of this species' range are more than 8 years old.
		Oceanic (2003–2004)	0	-	35	
		2009	2968	0.67	38	
		2017–2018	**21,506**	**0.26**	**50+51**	Garrison *et al.* 2020a and Garrison *et al.* 2021
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	
		2017–2018	213	1.03	50	Garrison *et al.* 2020a
Rough-toothed Dolphin	Northern Gulf of Mexico	Oceanic (1991–1994)	852	0.31	30	
		Oceanic (1996–2001)	985	0.44	33	Mullin and Fulling 2004
		OCS (1998–2001)	1,145	0.83	34	Abundance estimate is from 2000-2001 surveys only (from Fulling *et al.* 2003). Current best population size estimate is unknown because data from the continental shelf portion of this species' range are more than 8 years old.
		Oceanic (2003–2004)	1,508	0.39	35	
		2009	624	0.99	38	
Bottlenose Dolphin	Northern Gulf of Mexico: Oceanic	1996–2001	2,239	0.41	33	Mullin and Fulling 2004
		2003–2004	3,708	0.42	35	
		2009	5,806	0.39	38	
		2017–2018	213	1.03	50	Garrison *et al.* 2020a
Bottlenose Dolphin	Northern Gulf of Mexico: Continental Shelf	1998–2001	17,777	0.32	34	Abundance estimate is from 2000-2001 surveys only (from Fulling *et al.* 2003). Current best population size estimate is unknown because data from the continental shelf are more than 8 years old.
		2017–2018	**63,280**	**0.11**	**51**	Garrison *et al.* 2021
		Eastern (1994)	9,912	0.12	32	

Species	Stock	Year	Nest	CV	Survey Number	Notes
Bottlenose Dolphin	Northern Gulf of Mexico: Coastal (3 stocks)	Eastern (2007)	7,702	0.19	39	
		Eastern (2017–2018)	**16,407**	**0.17**	**51**	Garrison *et al.* 2021
		Northern (1993)	4,191	0.21	31	Current best population size estimate for this stock is unknown because data are more than 8 years old (Blaylock and Hoggard 1994)
		Northern (2007)	2,473	0.25	39	
		Northern (2017–2018)	**11,543**	**0.19**	**51**	Garrison *et al.* 2021
		Western (1992)	3,499	0.21	31	Current best population size estimate for this stock is unknown because data are more than 8 years old (Blaylock and Hoggard 1994)
		Western (2017–2018)	**20,759**	**0.13**	**51**	Garrison *et al.* 2021
Bottlenose Dolphin	Northern Gulf of Mexico: Bay, Sound and Estuarine (31 stocks)	Choctawhatchee Bay (2007)	179	0.04		Conn *et al.* 2011
		St. Joseph Bay (2011)	142	0.17		Balmer *et al. 2018*
		Sarasota Bay, Little Sarasota Bay (2015)	158	0.27		Tyson and Wells 2016
		Mississippi River Delta (2017–2018)	**1,446**	**0.19**	**51**	Garrison *et al.* 2021
		Mississippi Sound, Lake Borgne, Bay Boudreau (2018)	**1,265**	**0.35**	**51**	Garrison *et al.* 2021
		Barataria Bay (2019)	**2,071**	**0.06**		Garrison *et al.* 2020b
		West Bay (2014–2015)	37	0.05		Ronje *et al.* 2020
		Galveston Bay, East Bay, Trinity Bay (2016)	**842**	**0.8**		Ronje *et al.* 2020
		Terrebonne Bay, Timbalier Bay (2016)	3,870	0.15		Litz *et al.* 2018
		St. Andrew Bay (2016)	199	0.09		Balmer *et al.* 2019
		Sabine Lake (2017)	122	0.19		Ronje *et al.* 2020
		Remaining 20 stocks	unknown	undetermined	31	Current best population size estimate for each of these 20 stocks is unknown because data are more than 8 years old (Blaylock and Hoggard 1994)

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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	**M/SI (est)**	**CV**	**M/SI (est)**	**CV**	**M/SI (est)**	**CV**	**M/SI (est)**	**CV**	**M/SI (est)**	**CV**	**M/SI (est)**	**CV**	**M/SI (est)**	**CV**	**M/SI (est)**	**CV**	**M/SI (est)**	**CV**
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	.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	0	0	0	0	55	0.54	0	0	0	0	474	0.17	1021	0.25	119	0.34
2016	125	0.34	0	0	0	0	80	0.38	0	0	0	0	245	0.29	498	0.33	85	0.5
2017	136	0.28	8	0.92	0	0	133	0.28	0	0	0	0	298	0.18	930	0.16	44	0.37

	Harbor Porpoise		Bottlenose Dolphin, Atlantic Offshore Stock		White-sided Dolphin		Common Dolphin		Risso's Dolphin		Long-finned Pilot Whale		Harbor Seal		Gray Seal		Harp Seal	
Year	**M/SI (est)**	**CV**	**M/SI (est)**	**CV**	**M/SI (est)**	**CV**	**M/SI (est)**	**CV**	**M/SI (est)**	**CV**	**M/SI (est)**	**CV**	**M/SI (est)**	**CV**	**M/SI (est)**	**CV**	**M/SI (est)**	**CV**
2018	92	0.52	0	0	0	0	93	0.45	0	0	0	0	188	0.36	1113	0.32	14	0.8
2019	145	0.14	2	4.6	0	0	73	0.19	1	3.5	0	0	304	0.1	1116	0.11	85	.16

Note: This table only includes observed bycatch. For a complete list of marine mammal species interactions with this fishery, please see https://www.fisheries.noaa.gov/national/marine-mammal-protection/northeast-sink-gillnet-fishery-mmpa-list-fisheries.
[a] Unextrapolated mortalities
na=not applicable; unk= observer coverage was absent or too low to detect bycatch, or no estimate generated; tbd= to be determined

Mid-Atlantic Sink Gillnet

	Harbor Porpoise		Bottlenose Dolphin, Atlantic Offshore Stock		White-sided Dolphin		Common Dolphin		Risso's Dolphin		Pilot Whale, Unidentified		Harbor Seal		Gray Seal		Harp Seal	
Year	**M/SI**	**CV**	**M/SI**	**CV**	**M/SI**	**CV**	**M/SI**	**CV**	**M/SI**	**CV**	**M/SI**	**CV**	**M/SI**	**CV**	**M/SI**	**CV**	**M/SI**	**CV**
1994	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
1995	103	0.57	56	1.66	0	0	7.4	0.69	0	0	0	0	0	0	0	0	0	0
1996	311	0.31	64	0.83	0	0	43	0.79	0	0	0	0	0	0	0	0	0	0
1997	572	0.35	0	0	45	0.82	0	0	0	0	0	0	0	0	0	0	0	0
1998	446	0.36	63	0.94	0	0	0	0	0	0	7	0	11	0.77	0	0	17	1.02
1999	53	0.49	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
2000	21	0.76	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
2001	26	0.95	na	na	0	0	0	0	0	0	0	0	0	0	0	0	0	0
2002	unk	na	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
2003	76	1.13	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
2004	137	0.91	0	0	0	0	0	0	0	0	0	0	15	0.86	69	0.92	0	0
2005	470	0.51	1[a]	na	0	0	0	0	0	0	0	0	63	0.67	0	0	0	0
2006	511	0.32	0	0	0	0	0	0	0	0	0	0	26	0.98	0	0	0	0
2007	58	1.03	0	0	0	0	0	0	34	0.73	0	0	0	0	0	0	38	0.9
2008	350	0.75	0	0	0	0	0	0	0	0	0	0	88	0.74	0	0	176	0.74
2009	201	0.55	0	0	0	0	0	0	0	0	0	0	47	0.68	0	0	0	0
2010	259	0.88	0	0	0	0	30	0.48	0	0	0	0	89	0.39	267	0.75	0	0
2011	123	0.41	0	0	0	0	29	0.53	0	0	0	0	21	0.67	19	0.60	0	0
2012	63.41	0.83	0	0	0	0	15	0.93	0	0	0	0	0	0	14	0.98	0	0
2013	19	1.06	26	0.95	0	0	62	0.67	0	0	0	0	0	0	0	0	0	0
2014	22	1.03	0	0	0	0	17	0.86	0	0	0	0	19	1.06	22	1.09	0	0
2015	60	1.16			0	0	30	0.55	0	0	0	0	48	0.52	15	1.04	0	0
2016	23	0.64			0	0	7	0.97	0	0	0	0	18	0.95	7	0.93	0	0
2017	9	0.95	0	0	0	0	22	0.71	0	0	0	0	3	0.62	0	0	0	0
2018	0	0	0	0	0	0	8	0.91	0	0	0	0	26	0.52	0	0	0	0
2019	16	0.68	0	0	0	0	17	0.31	0	0	0	0	22	0.3	8	0.76	6	4.2

Note: This table only includes observed bycatch. For a complete list of marine mammal species interactions with this fishery, please see https://www.fisheries.noaa.gov/national/marine-mammal-protection/mid-atlantic-gillnet-fishery-mmpa-list-fisheries. For bottlenose dolphin stocks not listed in this table (Northern Migratory Coastal Stock, Southern Migratory Coastal Stock, Northern NC Estuarine Stock, Southern NC Estuarine Stock), see Lyssikatos & Garrison 2018 and Lyssikatos 2021.

[a] Unextrapolated mortalities

na=not applicable; unk= observer coverage was absent or too low to detect bycatch, or no estimate generated; tbd= to be determined

New England/North Atlantic Bottom Trawl

	Harbor Porpoise		Bottlenose Dolphin, Atlantic Offshore Stock		White-sided Dolphin		Common Dolphin		Risso's Dolphin, Atlantic		Pilot Whale, Unidentified		Long-finned Pilot Whale		Harbor Seal		Gray Seal		Harp Seal		Minke Whale	
Year	**M/SI**	**CV**	**M/SI**	**CV**	**M/SI**	**CV**	**M/SI**	**CV**	**M/SI**	**CV**	**M/SI**	**CV**	**M/SI**	**CV**	**M/SI**	**CV**	**M/SI**	**CV**	**M/SI**	**CV**	**M/SI**	**CV**
1990	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
1991	0	0	91	0.97	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
1992	0	0	0	0	110	0.97	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
1993	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
1994	0	0	0	0	182	0.71	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
1995	0	0	0	0	0	0	142	0.77	0	0	0	0	0	0	0	0	0	0	0	0	0	0
1996	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
1997	0	0	0	0	0	0	93	1.06	0	0	0	0	0	0	0	0	0	0	0	0	0	0
1998	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
1999	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
2000	0	0	0	0	137	0.34	27	0.29	0	0	0	0	0	0	0	0	0	0	0	0	0	0
2001	0	0	0	0	161	0.34	30	0.3	0	0	21	0.27	0	0	0	0	0	0	49	1.1	0	0
2002	0	0	0	0	70	0.32	26	0.29	0	0	22	0.26	0	0	0	0	0	0	0	0	0	0
2003	*	*	0	0	216	0.27	26	0.29	0	0	20	0.26	0	0	0	0	0	0	0	0	0	0
2004	0	0	0	0	200	0.30	26	0.29	0	0	15	0.29	0	0	0	0	0	0	0	0	0	0
2005	7.2	0.48	0	0	213	0.28	32	0.28	0	0	15	0.30	0	0	0	0	unk	unk	unk	unk	0	0
2006	6.5	0.49	0	0	40	0.50	25	0.28	0	0	14	0.28	0	0	0	0	0	0	0	0	0	0
2007	5.6	0.46	48	0.95	29	0.66	24	0.28	3	0.52	0	0	0	0	0	0	unk	unk	0	0	0	0
2008	5.6	0.97	19	0.88	13	0.57	6	0.99	2	0.56	0	0	21	0.51	0	0	16	0.52	0	0	7.8	0.69
2009	0	0	18	0.92	171	0.28	24	0.60	3	0.53	0	0	13	0.70	0	0	22	0.46	5	1.02	0	0
2010	0	0	4	0.53	37	0.32	114	0.32	2	0.55	0	0	30	0.43	0	0	30	0.34	0	0	0	0
2011	5.9	0.71	10	0.84	141	0.24	72	0.37	3	0.55	0	0	55	0.18	9	0.58	58	0.25	3	1.02	0	0
2012	0	0	0	0	27	0.47	40	0.54	0	0	0	0	33	0.32	3	1	37	0.49	0	0	0	0
2013	7	0.98	0	0	33	0.31	17	0.54	0	0	0	0	16	0.42	4	0.89	20	0.37	0	0	0	0
2014	5.5	0.86	0	0	16	0.5	17	0.53	4.2	0.91	0	0	32	0.44	11	0.63	19	0.45	0	0	0	0
2015	3.7	0.49	19	0.65	15	0.52	22	0.45	0	0	0	0	0	0	0	0	23	0.46	0	0	0	0
2016	0	0	33.5	0.89	28	0.46	16	0.46	17	0.88	0	0	29	0.58	0	0	0	0	0	0	0	0
2017	0	0	0	0	15	0.64	0	0	0	0	0	0	0	0	8.3	0	16	0.24	0	0	0	0
2018	0	0	0	0	0	0	28	0.54	0	0	0	0	0	0	0	0	32	0.42	0	0	0	0
2019	2.2	0.63	11	0.56	27	0.21	15	0.27	3.4	0.88	0	0	6.9	0.51	2.7	0.68	20	0.23	1	0.89	0.2 [a]	na

Note: This table only includes observed bycatch. For a complete list of marine mammal species interactions with this fishery, please see https://www.fisheries.noaa.gov/national/marine-mammal-protection/northeast-bottom-trawl-fishery-mmpa-list-fisheries

na=not applicable; unk= observer coverage was absent or too low to detect bycatch, or no estimate generated; tbd= to be determined

[a] Unextrapolated mortalities

Mid-Atlantic Bottom Trawl

	Bottlenose Dolphin, Atlantic Offshore Stock		White-sided Dolphin		Common Dolphin		Risso's Dolphin, Atlantic		Pilot Whale, Unidentified		Harbor Seal		Gray Seal	
Year	**M/SI**	**CV**	**M/SI**	**CV**	**M/SI**	**CV**	**M/SI**	**CV**	**M/SI**	**CV**	**M/SI**	**CV**	**M/SI**	**CV**
1997	0	0	161	1.58	0	0	0	0	0	0	0	0	0	0
1998	0	0	0	0	0	0	0	0	0	0	0	0	0	0
1999	0	0	0	0	0	0	0	0	228	1.03	0	0	0	0
2000	0	0	27	0.17	0	0	0	0	0	0	0	0	0	0
2001	0	0	27	0.19	103	0.27	0	0	39	0.3	0	0	0	0
2002	0	0	25	0.17	87	0.27	0	0	38	0.36	0	0	0	0
2003	0	0	31	0.25	99	0.28	0	0	31	0.31	0	0	0	0
2004	0	0	26	0.2	159	0.3	0	0	35	0.33	0	0	0	0
2005	0	0	38	0.29	141	0.29	0	0	31	0.31	0	0	0	0
2006	0	0	3	0.53	131	0.28	0	0	37	0.34	0	0	0	0
2007	11	0.42	2	1.03	66	0.27	33	0.34	0	0	0	0	0	0
2008	16	0.36	0	0	23	1	39	0.69	0	0	0	0	0	0
2009	21	0.45	0	0	167	0.46	23	0.5	0	0	24	0.92	38	0.7
2010	20	0.34	0	0	21	0.96	54	0.74	0	0	11	1.1	0	0
2011	34	0.31	0	0	271	0.25	62	0.56	0	0	0	0	25	0.57
2012	16	1.00	0	0	323	0.26	8	1	0	0	23	1	30	1.1
2013	0	0	0	0	269	0.29	42	0.71	0	0	11	0.96	29	0.67
2014	25	0.66	9.7	0.94	329	0.29	21	0.93	0	0	10	0.95	7	0.96
2015	0	0	0	0	250	0.32	40	0.63	0	0	7.4	1.0	0	0
2016	7.3	0.93	0	0	177	0.33	39	0.56	0	0	0	0	26	0.57
2017	22.1	0.66	0	0	380	0.23	43	0.51	0	0	0	0	26	0.40
2018	6.33	0.91	0	0	205	0.21	0	0	0	0	5.6	0.94	56	0.58
2019	7.2	0.48	0	0	281	0.12	24	0.33	0	0	4.1	0.56	26	0.30

Note: This table only includes observed bycatch. For a complete list of marine mammal species interactions with this fishery, please see https://www.fisheries.noaa.gov/national/marine-mammal-protection/mid-atlantic-bottom-trawl-fishery-mmpa-list-fisheries
na=not applicable; unk= observer coverage was absent or too low to detect bycatch, or no estimate generated; tbd= to be determined

Northeast Mid-Water Trawl

	White-sided Dolphin		Common Dolphin		Pilot Whale, Unidentified		Long-finned Pilot Whale		Harbor Seal		Gray Seal	
Year	**M/SI**	**CV**	**M/SI**	**CV**	**M/SI**	**CV**	**M/SI**	**CV**	**M/SI**	**CV**	**M/SI**	**CV**
1999	0	0	0	0	0	0	0	0	0	0	0	0
2000	0	0	0	0	4.6	0.74	0	0	0	0	0	0
2001	unk	na	0	0	11	0.74	0	0	0	0	0	0
2002	unk	na	0	0	8.9	0.74	0	0	0	0	0	0
2003	22	0.97	0	0	14	0.56	0	0	0	0	0	0
2004	0	0	0	0	5.8	0.58	0	0	0	0	0	0
2005	9.4	1.03	0	0	1.1	0.68	0	0	0	0	0	0
2006	0	0	0	0	0	0	0	0	0	0	0	0
2007	0	0	0	0	0	0	0	0	0	0	0	0
2008	0	0	0	0	0	0	16	0.61	0	0	0	0
2009	0	0	0	0	0	0	0	0	1.3	0.81	0	0
2010	0	0	1[a]	na	0	0	0	0	2[a]	na	0	0
2011	0	0	0	0	0	0	1	0	0	0	0	0
2012	0	0	1[a]	na	0	0	1	0	1[a]	na	1[a]	na
2013	0	0	0	0	0	0	3	0	0	0	1[a]	na
2014	0	0	0	0	0	0	4	na	1[a]	na	0	0
2015	0	0	0	0	0	0	0	na	2[a]	na	0	0
2016	0	0	0	0	0	0	3	na	1[a]	na	0	0
2017	0	0	0	0	0	0	0	na	0	na	0	0
2018	0	0	0	0	0	0	0	0	0	0	1[a]	na
2019	0	0	0	0	0	0	0	0	0	0	0	0

Note: This table only includes observed bycatch. For a complete list of marine mammal species interactions with this fishery, please see https://www.fisheries.noaa.gov/national/marine-mammal-protection/northeast-mid-water-trawl-fishery-mmpa-list-fisheries
[a] Unextrapolated mortalities
na = not applicable; unk = observer coverage was absent or too low to detect bycatch, or no estimate generated; tbd = to be determined

Mid-Atlantic Mid-Water Trawl

	White-sided Dolphin		Common Dolphin		Risso's Dolphin, Atlantic		Harbor Seal		Gray Seal	
Year	**M/SI**	**CV**	**M/SI**	**CV**	**M/SI**	**CV**	**M/SI**	**CV**	**M/SI**	**CV**
1999	0	0	0	0	0	0	0	0	0	0
2000	0	0	0	0	0	0	0	0	0	0
2001	unk	na	0	0	0	0	0	0	0	0
2002	unk	na	0	0	0	0	0	0	0	0
2003	0	0	0	0	0	0	0	0	0	0
2004	22	0.99	0	0	0	0	0	0	0	0
2005	58	1.02	0	0	0	0	0	0	0	0
2006	29	0.74	0	0	0	0	0	0	0	0
2007	12	0.98	3.2	0.7	0	0	0	0	0	0
2008	15	0.73	0	0	1[a]	na	0	0	0	0
2009	4	0.92	0	0	0	0	0	0	0	0
2010	0	0	0	0	0	0	1[a]	na	1[a]	na
2011	0	0	0	0	0	0	0	0	0	0
2012	0	0	0	0	0	0	0	0	0	0
2013	0	0	0	0	0	0	0	0	0	0
2014	0	0	0	0	0	0	0	0	0	0
2015	0	0	0	0	0	0	0	0	0	0
2016	0	0	0	0	0	0	0	0	0	0
2017	0	0	0	0	0	0	0	0	0	0
2018	0	0	0	0	0	0	0	0	0	0
2019	0	0	0	0	0	0	0	0	0	0

Note: This table only includes observed bycatch. For a complete list of marine mammal species interactions with this fishery, please see https://www.fisheries.noaa.gov/national/marine-mammal-protection/mid-atlantic-mid-water-trawl-includes-pair-trawl-fishery-mmpa

[a] Unextrapolated mortalities

na = not applicable; unk = observer coverage was absent or too low to detect bycatch, or no estimate generated; tbd = to be determined

Pelagic Longline

	Pantropical Spotted Dolphin, GMex		Bottlenose Dolphin, Atlantic Offshore Stock		Common Dolphin		Risso's Dolphin, Atlantic		Risso's Dolphin, Gmex		Pilot Whale, Unidentified & Long-finned, Atlantic		Short-finned Pilot Whale, Atlantic		Beaked Whale, Unidentified	
Year	**M/SI**	**CV**	**M/SI**	**CV**	**M/SI**	**CV**	**M/SI**	**CV**	**M/SI**	**CV**	**M/SI**	**CV**	**M/SI**	**CV**	**M/SI**	**CV**
1992	0	0	0	0	0	0	0	0	0	0	22	0.23	0	0	0	0
1993	0	0	0	0	0	0	13	0.19	0	0	0	0	0	0	0	0
1994	0	0	0	0	0	0	7	1	0	0	137	0.44	0	0	0	0
1995	0	0	0	0	0	0	103	0.68	0	0	345	0.51	0	0	0	0
1996	0	0	0	0	0	0	99	1	0	0	0	0	0	0	0	0
1997	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
1998	0	0	0	0	0	0	57	1	0	0	0	0	0	0	0	0
1999	0	0	0	0	0	0	22	1	0	0	381	0.79	0	0	0	0
2000	0	0	0	0	0	0	64	1	0	0	133	0.88	0	0	0	0
2001	0	0	0	0	0	0	69	0.57	0	0	79	0.48	0	0	0	0
2002	0	0	0	0	0	0	28	0.86	0	0	54	0.46	0	0	0	0
2003	0	0	0	0	0	0	40	0.63	0	0	21	0.77	0	0	5.3	1
2004	0	0	0	0	0	0	28	0.72	0	0	74	0.42	0	0	0	0
2005	0	0	0	0	0	0	3	1	0	0	212	0.21	0	0	0	0
2006	0	0	0	0	0	0	0	0	0	0	185	0.47	0	0	0	0
2007	0	0	0	0	0	0	9	0.65	0	0	57	0.65	0	0	0	0
2008	0	0	0	0	0	0	16.8	0.73	8.3	0.63	0	0	80	0.42	0	0
2009	16	0.69	8.8	1	8.5	1	11.8	0.711	0	0	0	0	17	0.7	0	0
2010	0	0	0	0	0	0	0	0	0	0	0	0	127	0.78	0	0
2011	0	0	0	0	0	0	12	0.70	1.5	1	0	0	305	0.29	0	0
2012	0	0	62	0.68	0	0	15	1	30	1	0	0	170.1	0.33	0	0
2013	2.1	1	0	0	0	0	1.9	1	15	1	0	0	124	0.32	0	0
2014	0	0	0	0	0	0	7.7	1	0	0	9.6	0.43	233	0.24	0	0
2015	0	0	0	0	9.05	1	8.4	0.71	0	0	2.2	0.49	200	0.24	0	0
2016	0	0	0	0	0	0	16	0.57	0	0	1.1	0.6	111	0.31	0	0
2017	0	0	0	0	4.92	1	0.2	1	0	0	3.3	0.98	133	0.29	0	0
2018	0	0	17.3	0.73	1.44	1	0.2	0.94	0	0	0.4	0.93	102	0.39	0	0

	Pantropical Spotted Dolphin, GMex		Bottlenose Dolphin, Atlantic Offshore Stock		Common Dolphin		Risso's Dolphin, Atlantic		Risso's Dolphin, Gmex		Pilot Whale, Unidentified & Long-finned, Atlantic		Short-finned Pilot Whale, Atlantic		Beaked Whale, Unidentified	
Year	**M/SI**	**CV**	**M/SI**	**CV**	**M/SI**	**CV**	**M/SI**	**CV**	**M/SI**	**CV**	**M/SI**	**CV**	**M/SI**	**CV**	**M/SI**	**CV**
2019	12.9	1	0	0	0	0	0	0	0	0	0.4	1	131	0.37	0.3	1

na = not applicable; unk = observer coverage was absent or too low to detect bycatch, or no estimate generated; tbd = to be determined

Pelagic Drift Gillnet

	White-sided Dolphin		Common Dolphin		Risso's Dolphin, Atlantic		Pilot Whale, Unidentified		Long-finned Pilot Whale		Bottlenose Dolphin, Atlantic Offshore Stock		Beaked Whale, Unidentified		Sowerby's Beaked Whales		Harbor Porpoise	
Year	**M/SI (est)**	**CV**	**M/SI (est)**	**CV**	**M/SI (est)**	**CV**	**M/SI (est)**	**CV**	**M/SI (est)**	**CV**	**M/SI (est)**	**CV**	**M/SI (est)**	**CV**	**M/SI (est)**	**CV**	**M/SI (est)**	**CV**
1989	4.4	0.71	0	0	87	0.52	0	0	0	0	72	0.18	60	0.21	0	0	0.7	7
1990	6.8	0.71	0	0	144	0.46	0	0	0	0	115	0.18	76	0.26	0	0	1.7	2.65
1991	0.9	0.71	223	0.12	21	0.55	30	0.26	0	0	26	0.15	13	0.21	0	0	0.7	1
1992	0.8	0.71	227	0.09	31	0.27	33	0.16	0	0	28	0.1	9.7	0.24	0	0	0.4	1
1993	2.7	0.17	238	0.08	14	0.42	31	0.19	0	0	22	0.13	12	0.16	0	0	1.5	0.34
1994	0	0.71	163	0.02	1.5	0.16	20	0.06	0	0	14	0.04	0	0	3	0.09	0	0
1995	0	0	83	0	6	0	9.1	0	0	0	5	0	3	0	6	0	0	0
1996	0	0	0	0	0	0	0	0	0	0	0	0	2	0.25	9	0.12	0	0
1997	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
1998	0	0	0	0	9	0	0	0	0	0	3	0	7	0	2	0	0	0
1999	0	0	0	0	0	0	20	0	0	0	0	0	0	0	0	0	0	0

Note: This table only includes observed bycatch.

Pelagic Pair Trawl

	White-sided Dolphin		Common Dolphin		Risso's Dolphin, Atlantic		Pilot Whale, Unidentified		Long-finned Pilot Whale		Bottlenose Dolphin, Atlantic Offshore	
Year	**M/SI (est)**	**CV**	**M/SI (est)**	**CV**	**M/SI (est)**	**CV**	**M/SI (est)**	**CV**	**M/SI (est)**	**CV**	**M/SI (est)**	**CV**
1989	0	0	0	0	0	0	0	0	0	0	0	0
1990	0	0	0	0	0	0	0	0	0	0	0	0
1991	0	0	0	0	0.6	1	0	0	0	0	13	0.52
1992	0	0	0	0	4.3	0.76	0	0	0	0	73	0.49
1993	0	0	0	0	3.2	1	0	0	0	0	85	0.41
1994	0	0	0	0	0	0	2	0.49	0	0	4	0.4
1995	0	0	0	0	3.7	0.45	22	0.33	0	0	17	0.26
1996	0	0	0	0	0	0	0	0	0	0	0	0
1997	0	0	0	0	0	0	0	0	0	0	0	0
1998	0	0	0	0	0	0	0	0	0	0	0	0
1999	0	0	0	0	0	0	0	0	0	0	0	0

Note: This table only includes observed bycatch.
na = not applicable; unk = observer coverage was absent or too low to detect bycatch, or no estimate generated; tbd = to be determined

Gulf of Mexico Shrimp Otter Trawl

	Atlantic Spotted Dolphin		Bottlenose Dolphin, Continental Shelf Stock		Bottlenose Dolphin, Western Coastal Stock		Bottlenose Dolphin, Northern Coastal Stock		Bottlenose Dolphin, Eastern Coastal Stock		Bottlenose Dolphin, TX BSE Stocks		Bottlenose Dolphin, LA BSE Stocks		Bottlenose Dolphin, AL/MS BSE Stocks		Bottlenose Dolphin, FL BSE Stocks	
Year	**M/SI (est)**	**CV**	**M/SI (est)**	**CV**	**M/SI (est)**	**CV**	**M/SI (est)**	**CV**	**M/SI (est)**	**CV**	**M/SI (est)**	**CV**	**M/SI (est)**	**CV**	**M/SI (est)**	**CV**	**M/SI (est)**	**CV**
1997	128	0.44	172	0.42	217	0.84	13	0.80	18	0.99	0	-	29	1.00	37	0.82	3	0.99
1998	146	0.44	180	0.43	148	0.80	20	0.95	23	0.99	0	-	31	0.99	37	0.83	2	0.99
1999	120	0.44	159	0.42	289	0.91	31	0.72	11	0.99	0	-	38	0.89	52	0.85	3	0.99
2000	105	0.44	156	0.43	242	0.86	15	0.72	15	0.99	0	-	21	0.86	47	0.77	8	0.99
2001	115	0.45	169	0.42	291	0.85	15	0.79	11	0.99	0	-	28	0.99	55	0.74	6	0.99
2002	128	0.44	166	0.42	223	0.80	29	0.84	12	0.99	0	-	118	0.98	69	0.84	6	0.99
2003	75	0.45	122	0.43	133	0.79	15	0.71	5	0.99	0	-	72	1.00	52	0.82	5	0.99
2004	84	0.46	132	0.43	111	0.80	14	0.88	5	0.99	0	-	77	0.90	26	0.90	2	0.99
2005	55	0.49	94	0.43	66	0.84	11	0.64	1	0.99	0	-	57	0.96	15	0.72	3	0.99
2006	49	0.44	77	0.43	105	0.89	16	0.67	6	0.99	0	-	55	0.97	17	0.64	3	0.99
2007	43	0.45	60	0.43	81	0.85	20	0.67	3	0.99	0	-	47	0.90	26	0.77	1	0.99
2008	37	0.53	46	0.44	56	0.80	22	0.77	1	0.99	0	-	61	1.00	28	0.76	1	0.99
2009	49	0.50	56	0.43	77	0.89	35	0.67	3	0.99	0	-	116	1.02	45	0.73	6	0.99
2010	44	0.42	57	0.40	57	0.83	17	0.64	3	0.99	0	-	113	1.09	58	0.64	6	0.99
2011	35	0.48	63	0.44	67	0.91	13	0.65	1	0.99	0	-	104	0.98	47	0.64	3	0.99
2012	28	0.44	49	0.37	48	0.79	12	0.68	0.6	1.01	0	-	31	0.76	12	0.80	0.2	1.01
2013	27	0.43	57	0.38	23	0.74	6.0	0.83	0.7	1.01	0	-	19	0.74	14	0.95	1.1	1.01
2014	23	0.43	58	0.40	57	0.84	8.3	0.74	1.1	0.98	0	-	40	0.94	2.8	0.66	1.2	0.98
2015	24	0.39	62	0.34	18	0.55	4.5	0.57	4.1	1.00	0.3	1.01	32	0.64	20	0.67	0.1	1.00
2016	43	0.41	70	0.33	46	0.47	7.2	0.56	8.1	1.00	1.1	1.00	53	0.63	46	0.63	1.7	1.00
2017	46	0.40	72	0.30	46	0.48	5.4	0.55	9.8	1.00	0.6	1.00	63	0.52	29	0.57	0.9	1.00
2018	36	0.40	64	0.30	33	0.47	5.6	0.55	8.7	0.98	0.1	0.99	45	0.53	35	0.62	0.2	0.98
2019	29	0.38	50	0.33	17	0.47	9.9	0.55	7.2	0.98	0.1	1.02	34	0.61	33	0.63	0.5	0.98

Note: This table only includes observed bycatch. For a complete list of marine mammal species interactions with this fishery, please see https://www.fisheries.noaa.gov/national/marine-mammal-protection/southeastern-us-atlantic-gulf-mexico-shrimp-trawl-fishery-mmpa.
na = not applicable; unk = observer coverage was absent or too low to detect bycatch, or no estimate generated; tbd = to be determined

Appendix V: Fishery Bycatch Summaries

Part B: By Species

Harbor Porpoise

	Mid-Atlantic Gillnet		North Atlantic Bottom Trawl		NE Sink Gillnet		Pelagic Drift Gillnet	
Year	**M/SI (est)**	**CV**	**M/SI (est)**	**CV**	**M/SI (est)**	**CV**	**M/SI (est)**	**CV**
1990	na	na	0	0	2900	0.32	1.7	2.65
1991	na	na	0	0	2000	0.35	0.7	1
1992	na	na	0	0	1200	0.21	0.4	1
1993	na	na	0	0	1400	0.18	1.5	0.34
1994	na	na	0	0	2100	0.18		
1995	103	0.57	0	0	1400	0.27		
1996	311	0.31	0	0	1200	0.25		
1997	572	0.35	0	0	782	0.22		
1998	446	0.36	0	0	332	0.46		
1999	53	0.49	0	0	270	0.28		
2000	21	0.76	0	0	507	0.37		
2001	26	0.95	0	0	53	0.97		
2002	unk	na	0	0	444	0.37		
2003	76	1.13	*	*	592	0.33		
2004	137	0.91	0	0	654	0.36		
2005	470	0.51	7.2	0.48	630	0.23		
2006	511	0.32	6.5	0.49	514	0.31		
2007	58	1.03	5.6	0.46	395	0.37		
2008	350	0.75	5.6	0.97	666	0.48		
2009	201	0.55	0	0	591	0.23		
2010	259	0.88	0	0	387	0.27		
2011	123	0.41	5.9	0.71	273	0.2		
2012	63.41	0.83	0	0	277.3	0.59		
2013	19	1.06	7	0.98	399	0.33		
2014	22	1.03	5.5	0.86	128	0.27		
2015	60	1.16	3.7	0.49	177	0.28		
2016	23	0.64	0	0	125	0.34		
2017	9	0.95	0	0	136	0.28		
2018	0	0	0	0	92	0.52		
2019	13	0.51	0	0	195	0.22		

Note: This table only includes observed bycatch.

na = not applicable; unk = observer coverage was absent or too low to detect bycatch, or no estimate generated; tbd = to be determined

Common Bottlenose Dolphin, Atlantic Offshore Stock

	Mid-Atlantic Bottom Trawl		Mid-Atlantic Gillnet		North Atlantic Bottom Trawl		NE Sink Gillnet		Pelagic Drift Gillnet		Pelagic Longline	
Year	**M/SI (est)**	**CV**	**M/SI (est)**	**CV**	**M/SI (est)**	**CV**	**M/SI (est)**	**CV**	**M/SI (est)**	**CV**	**M/SI (est)**	**CV**
1991	na	na	na	na	91	0.97	0	0	26	0.15	0	0
1992	na	na	na	na	0	0	0	0	28	0.1	0	0
1993	na	na	na	na	0	0	0	0	22	0.13	0	0
1994	na	na	na	na	0	0	0	0	14	0.04	0	0
1995	na	na	56	1.66	0	0	0	0	5	0	0	0
1996	na	na	64	0.83	0	0	0	0	0	0	0	0
1997	0	0	0	0	0	0	0	0			0	0
1998	0	0	63	0.94	0	0	0	0			0	0
1999	0	0	0	0	0	0	0	0			0	0
2000	0	0	0	0	0	0	132	1.16			0	0
2001	0	0	na	na	0	0	0	0			0	0
2002	0	0	0	0	0	0	0	0			0	0
2003	0	0	0	0	0	0	0	0			0	0
2004	0	0	0	0	0	0	1[a]	na			0	0
2005	0	0	1[a]	na	0	0	0	0			0	0
2006	0	0	0	0	0	0	0	0			0	0
2007	11	0.42	0	0	48	0.95	0	0			0	0
2008	16	0.36	0	0	19	0.88	0	0			0	0
2009	21	0.45	0	0	18	0.92	0	0			8.8	1
2010	20	0.34	0	0	4	0.53	0	0			0	0
2011	34	0.31	0	0	10	0.84	0	0			0	0
2012	16	1	0	0	0	0	0	0			61.8	0.68
2013	0	0	0	0	0	0	26	0.95			0	0
2014	25	0.66	0	0	0	0	0	0			0	0
2015	0	0	0	0	18.6	0.65	0	0			0	0
2016	7.3	0.93	0	0	33.5	0.89	0	0			0	0
2017	22.1	0.66	0	0	0	0	8	0.92			0	0
2018	6.3	0.91	0	0	0	0	0	0			17.3	0.73
2019	0	0	0	0	5.6	0.92	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 Gillnet		Northeast Midwater Trawl		Pelagic Drift Gillnet	
Year	**M/SI (est)**	**CV**	**M/SI (est)**	**CV**	**M/SI (est)**	**CV**	**M/SI (est)**	**CV**	**M/SI (est)**	**CV**	**M/SI (est)**	**CV**	**M/SI (est)**	**CV**
1990	na	na	na	na	na	na	0	0	0	0	na	na		
1991	na	na	na	na	na	na	0	0	49	0.46	na	na	0	0
1992	na	na	na	na	na	na	110	0.97	154	0.35	na	na	110	0.97
1993	na	na	na	na	na	na	0	0	205	0.31	na	na	0	0
1994	na	na	0	0	na	na	182	0.71	240	0.51	na	na	182	0.71
1995	na	na	0	0	na	na	0	0	80	1.16	na	na	0	0
1996	na	na	0	0	na	na	0	0	114	0.61	na	na		
1997	161	1.58	45	0.82	na	na	0	0	140	0.61	na	na		
1998	0	0	0	0	na	na	0	0	34	0.92	na	na		
1999	0	0	0	0	0	0	0	0	69	0.7	0	0		
2000	27	0.17	0	0	0	0	137	0.34	26	1	0	0		
2001	27	0.19	0	0	unk	na	161	0.34	26	1	unk	na		
2002	25	0.17	0	0	unk	na	70	0.32	30	0.74	unk	na		
2003	31	0.25	0	0	0	0	216	0.27	31	0.93	22	0.97		
2004	26	0.2	0	0	22	0.99	200	0.3	7	0.98	0	0		
2005	38	0.29	0	0	58	1.02	213	0.28	59	0.49	9.4	1.03		
2006	3	0.53	0	0	29	0.74	40	0.5	41	0.71	0	0		
2007	2	1.03	0	0	12	0.98	29	0.66	0	0	0	0		
2008	0	0	0	0	15	0.73	13	0.57	81	0.57	0	0		
2009	0	0	0	0	4	0.92	171	0.28	0	0	0	0		
2010	0	0	0	0	0	0	37	0.32	66	0.9	0	0		
2011	0	0	0	0	0	0	141	0.24	18	0.43	0	0		
2012	0	0	0	0	0	0	27	0.47	9	0.92	0	0		
2013	0	0	0	0	0	0	33	0.31	4	1.03	0	0		
2014	9.7	0.94	0	0	0	0	16	0.50	10	0.66	0	0		
2015	0	0	0	0	0	0	15	0.52	0	0	0	0		
2016	0	0	0	0	0	0	28	0.46	0	0	0	0		
2017	0	0	0	0	0	0	15	0.64	0	0	0	0		

	Mid-Atlantic Bottom Trawl		Mid-Atlantic Gillnet		Mid-Atlantic Midwater Trawl		North Atlantic Bottom Trawl		NE Sink Gillnet		Northeast Midwater Trawl		Pelagic Drift Gillnet	
Year	**M/SI (est)**	**CV**	**M/SI (est)**	**CV**	**M/SI (est)**	**CV**	**M/SI (est)**	**CV**	**M/SI (est)**	**CV**	**M/SI (est)**	**CV**	**M/SI (est)**	**CV**
2018	0	0	0	0	0	0	0	0	0	0	0	0	0	0
2019	0	0	0	0	0	0	79	0.28	0	0	0	0	0	0

Note: This table only includes observed bycatch.

na = not applicable; unk = observer coverage was absent or too low to detect bycatch, or no estimate generated; tbd = to be determined

Risso's Dolphin, Western North Atlantic Stock

	Mid-Atlantic Bottom Trawl		Mid-Atlantic Gillnet		North Atlantic Bottom Trawl		NE Sink Gillnet		Pelagic Longline	
Year	**M/SI (est)**	**CV**	**M/SI (est)**	**CV**	**M/SI (est)**	**CV**	**M/SI (est)**	**CV**	**M/SI (est)**	**CV**
1996	0	0	0	0	0	0	0	0	99	1
1997	0	0	0	0	0	0	0	0	0	0
1998	0	0	0	0	0	0	0	0	57	1
1999	0	0	0	0	0	0	0	0	22	1
2000	0	0	0	0	0	0	15	1.06	64	1
2001	0	0	0	0	0	0	0	0	69	0.57
2002	0	0	0	0	0	0	0	0	28	0.86
2003	0	0	0	0	0	0	0	0	40	0.63
2004	0	0	0	0	0	0	0	0	28	0.72
2005	0	0	0	0	0	0	15	0.93	3	1
2006	0	0	0	0	0	0	0	0	0	0
2007	33	0.34	34	0.73	3	0.52	0	0	9	0.65
2008	39	0.69	0	0	2	0.56	0	0	16.8	0.732
2009	23	0.5	0	0	3	0.53	0	0	11.8	0.711
2010	54	0.74	0	0	2	0.55	0	0	0	0
2011	62	0.56	0	0	3	0.55	0	0	11.8	0.699
2012	8	1	0	0	0	0	6	0.87	15.1	1
2013	42	0.71	0	0	0	0	23	0.97	1.9	1
2014	21	0.93	0	0	4.2	0.91	0	0	7.7	1.0
2015	40	0.63	0	0	0	0	0	0	8.4	0.71
2016	39	0.56	0	0	17	0.88	0	0	16.1	0.57
2017	31	0.51	0	0	0	0	0	0	0.2	1
2018	0	0	0	0	0	0	0	0	0.2	0.94
2019	0	0	0	0	0	0	5.3	0.7	0	0

Note: This table only includes observed bycatch.
na = not applicable; unk = observer coverage was absent or too low to detect bycatch, or no estimate generated; tbd = to be determined

Long-finned Pilot Whale, Western North Atlantic Stock

	Mid-Atlantic Bottom Trawl		Mid-Atlantic Midwater Trawl		North Atlantic Bottom Trawl		NE Sink Gillnet		Northeast Midwater Trawl		Pelagic Longline	
Year	**M/SI (est)**	**CV**	**M/SI (est)**	**CV**	**M/SI (est)**	**CV**	**M/SI (est)**	**CV**	**M/SI (est)**	**CV**	**M/SI (est)**	**CV**
2008	0	0	0	0	21	0.51	0	0	16	0.61	na	na
2009	0	0	0	0	13	0.7	0	0	0	0	na	na
2010	0	0	0	0	30	0.43	3	0.82	0	0	na	na
2011	0	0	0	0	55	0.18	0	0	1	0	na	na
2012	0	0	0	0	33	0.32	0	0	1	0	na	na
2013	0	0	0	0	16	0.42	0	0	3	0	na	na
2014	0	0	0	0	32	0.44	0	0	4	na	9.6	0.43
2015	0	0	0	0	0	0	0	0	0	na	2.2	0.49
2016	0	0	0	0	29	0.58	0	0	3	na	1.1	0.6
2017	0	0	0	0	0	0	0	0	0	na	3.3	0.98
2018	0	0	0	0	0	0	0	0	0	0	0.4	0.93
2019	0	0	0	0	5.4	0.88	0	0	0	0	0.4	1

Note: This table only includes observed bycatch.
na = not applicable; unk = observer coverage was absent or too low to detect bycatch, or no estimate generated; tbd = to be determined

Short-finned Pilot Whale, Western North Atlantic Stock

	Pelagic Longline	
Year	**M/SI (est)**	**CV**
2008	80	0.42
2009	17	0.7
2010	127	0.78
2011	305	0.29
2012	170	0.33
2013	124	0.32
2014	233	0.24
2015	200	0.24
2016	111	0.31
2017	133	0.29
2018	102	0.39
2019	131	0.37

Note: This table only includes observed bycatch.
na=not applicable; unk= observer coverage was absent or too low to detect bycatch, or no estimate generated; tbd= to be determined

Common Dolphin, Western North Atlantic Stock

	Mid-Atlantic Bottom Trawl		Mid-Atlantic Gillnet		North Atlantic Bottom Trawl		NE Sink Gillnet		Northeast Midwater Trawl		Pelagic Drift Gillnet		Pelagic Longline	
Year	**M/SI (est)**	**CV**	**M/SI (est)**	**CV**	**M/SI (est)**	**CV**	**M/SI (est)**	**CV**	**M/SI (est)**	**CV**	**M/SI (est)**	**CV**	**M/SI (est)**	**CV**
1990	na	na	na	na	0	0	0	0	na	na			na	na
1991	na	na	na	na	0	0	0	0	na	na	223	0.12	na	na
1992	na	na	na	na	0	0	0	0	na	na	227	0.09	0	0
1993	na	na	na	na	0	0	0	0	na	na	238	0.08	0	0
1994	na	na	0	0	0	0	0	0	na	na	163	0.02	0	0
1995	na	na	7.4	0.69	142	0.77	0	0	na	na	83	0	0	0
1996	na	na	43	0.79	0	0	63	1.39	na	na			0	0
1997	0	0	0	0	93	1.06	0	0	na	na			0	0
1998	0	0	0	0	0	0	0	0	na	na			0	0
1999	0	0	0	0	0	0	146	0.97	0	0			0	0
2000	0	0	0	0	27	0.29	0	0	0	0			0	0
2001	103	0.27	0	0	30	0.3	0	0	0	0			0	0
2002	87	0.27	0	0	26	0.29	0	0	0	0			0	0
2003	99	0.28	0	0	26	0.29	0	0	0	0			0	0
2004	159	0.3	0	0	26	0.29	0	0	0	0			0	0
2005	141	0.29	0	0	32	0.28	5	0.8	0	0			0	0
2006	131	0.28	0	0	25	0.28	20	1.05	0	0			0	0
2007	66	0.27	0	0	24	0.28	11	0.94	0	0			0	0
2008	23	1	0	0	6	0.99	34	0.77	0	0			0	0
2009	167	0.46	0	0	24	0.6	43	0.77	0	0			8.8	1
2010	21	0.96	30	0.48	114	0.32	42	0.81	1[a]	na			0	0
2011	271	0.25	29	0.53	72	0.37	64	0.71	0	0			0	0
2012	323	0.26	15	0.93	40	0.54	95	0.4	1[a]	0			61.8	.68
2013	269	0.29	62	0.67	17	0.54	104	0.46	0	0			0	0
2014	17	0.53	17	0.86	17	0.53	111	0.47	0	0			0	0
2015	250	0.32	30	0.55	22	0.45	55	0.54	0	0			9.1	1.0
2016	177	0.33	7	0.97	16	0.46	80	0.38	0	0			0	0
2017	380	0.23	22	0.71	0	0	133	0.28	0	0			4.92	1

	Mid-Atlantic Bottom Trawl		Mid-Atlantic Gillnet		North Atlantic Bottom Trawl		NE Sink Gillnet		Northeast Midwater Trawl		Pelagic Drift Gillnet		Pelagic Longline	
Year	**M/SI (est)**	**CV**	**M/SI (est)**	**CV**	**M/SI (est)**	**CV**	**M/SI (est)**	**CV**	**M/SI (est)**	**CV**	**M/SI (est)**	**CV**	**M/SI (est)**	**CV**
2018	205	0.54	98	0.91	28	0.54	93	0.45	0	0			1.44	1
2019	395	0.23	20	0.56	10	0.62	5	0.68	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

Harbor Seal

	Herring Purse Seine		Mid-Atlantic Bottom Trawl		Mid-Atlantic Gillnet		Mid-Atlantic Midwater Trawl		Northeast Bottom Trawl		NE Sink Gillnet		Northeast Midwater Trawl	
Year	**M/SI (est)**	**CV**	**M/SI (est)**	**CV**	**M/SI (est)**	**CV**	**M/SI (est)**	**CV**	**M/SI (est)**	**CV**	**M/SI (est)**	**CV**	**M/SI (est)**	**CV**
1990	na	na	na	na	na	na	na	na	0	0	602	0.68	na	na
1991	na	na	na	na	na	na	na	na	0	0	231	0.22	na	na
1992	na	na	na	na	na	na	na	na	0	0	373	0.23	na	na
1993	na	na	na	na	na	na	na	na	0	0	698	0.19	na	na
1994	na	na	na	na	na	na	na	na	0	0	1330	0.25	na	na
1995	na	na	na	na	0	0	na	na	0	0	1179	0.21	na	na
1996	na	na	na	na	0	0	na	na	0	0	911	0.27	na	na
1997	na	na	0	0	0	0	na	na	0	0	598	0.26	na	na
1998	na	na	0	0	11	0.77	na	na	0	0	332	0.33	na	na
1999	na	na	0	0	0	0	na	na	0	0	1446	0.34	0	0
2000	na	na	0	0	0	0	0	0	0	0	917	0.43	0	0
2001	na	na	0	0	0	0	0	0	0	0	1471	0.38	0	0
2002	na	na	0	0	0	0	0	0	0	0	787	0.32	0	0
2003	0	0	0	0	0	0	0	0	0	0	542	0.28	0	0
2004	0	0	0	0	15	0.86	0	0	0	0	792	0.34	0	0
2005	0	0	0	0	63	0.67	0	0	0	0	719	0.2	0	0
2006	na	na	0	0	26	0.98	0	0	0	0	87	0.58	0	0
2007	0	0	0	0	0	0	0	0	0	0	92	0.49	0	0
2008	0	0	0	0	88	0.74	0	0	0	0	242	0.41	0	0
2009	0	0	24	0.92	47	0.68	0	0	0	0	513	0.28	1.3	0.81
2010	0	0	11	1.1	89	0.39	1[a]	0	0	0	540	0.25	2	0
2011	1[a]	0	0	0	21	0.67	0	0	9	0.58	343	0.19	0	0
2012	0	0	23	1	0	0	0	0	3	1	252	0.26	1	0
2013	0	0	11	0.96	0	0	0	0	4	0.89	147	0.3	0	0
2014	0	0	10	0.95	19	1.06	0	0	11	0.63	390	0.39	na	ma
2015	0	0	7.4	1.0	48	0.52	0	0	0	0	474	0.17	2[a]	na
2016	0	0	0	0	18	0.95	0	0	0	0	245	0.29	1[a]	na
2017	0	0	0	0	3	0.62	0	0	0	0	298	0.18	0	0

	Herring Purse Seine		Mid-Atlantic Bottom Trawl		Mid-Atlantic Gillnet		Mid-Atlantic Midwater Trawl		Northeast Bottom Trawl		NE Sink Gillnet		Northeast Midwater Trawl	
Year	**M/SI (est)**	**CV**	**M/SI (est)**	**CV**	**M/SI (est)**	**CV**	**M/SI (est)**	**CV**	**M/SI (est)**	**CV**	**M/SI (est)**	**CV**	**M/SI (est)**	**CV**
2018	0	0	6	0.94	26	0.52	0	0	0	0	188	0.36	0	0
2019	0	0	7	0.93	17	0.35	0	0	5	0.88	316	0.15	0	0

Note: This table only includes observed bycatch.
[a] Unextrapolated mortalities
na=not applicable; unk= observer coverage was absent or too low to detect bycatch, or no estimate generated; tbd= to be determined

Gray Seal

	Herring Purse Seine		Mid-Atlantic Bottom Trawl		Mid-Atlantic Gillnet		Mid-Atlantic Midwater Trawl		Northeast Bottom Trawl		NE Sink Gillnet		Northeast Midwater Trawl	
Year	**M/SI**	**CV**	**M/SI**	**CV**	**M/SI**	**CV**	**M/SI**	**CV**	**M/SI**	**CV**	**M/SI**	**CV**	**M/SI**	**CV**
1994	na	na	na	na	0	0	0	0	0	0	19	0.95	0	0
1995	na	na	na	na	0	0	0	0	0	0	117	0.42	0	0
1996	na	na	na	na	0	0	0	0	0	0	49	0.49	0	0
1997	na	na	0	0	0	0	0	0	0	0	131	0.5	0	0
1998	na	na	0	0	0	0	0	0	0	0	61	0.98	0	0
1999	na	na	0	0	0	0	0	0	0	0	155	0.51	0	0
2000	na	na	0	0	0	0	0	0	0	0	193	0.55	0	0
2001	na	na	0	0	0	0	0	0	0	0	117	0.59	0	0
2002	na	na	0	0	0	0	0	0	0	0	0	0	0	0
2003	0	0	0	0	0	0	0	0	0	0	242	0.47	0	0
2004	0	0	0	0	69	0.92	0	0	0	0	504	0.34	0	0
2005	0	0	0	0	0	0	0	0	unk	unk	574	0.44	0	0
2006	na	na	0	0	0	0	0	0	0	0	248	0.47	0	0
2007	0	0	0	0	0	0	0	0	unk	unk	886	0.24	0	0
2008	0	0	0	0	0	0	0	0	16	0.52	618	0.23	0	0
2009	0	0	38	0.7	0	0	0	0	22	0.46	1063	0.26	0	0
2010	0	0	0	0	267	0.75	1[a]	0	30	0.34	1155	0.28	0	0
2011	0	0	25	0.57	19	0.6	0	0	58	0.25	1491	0.22	0	0
2012	0	0	30	1.1	14	0.98	0	0	37	0.49	542	0.19	1[a]	na
2013	0	0	29	0.67	0	0	0	0	20	0.37	1127	0.2	1[a]	na
2014	0	0	7	0.96	22	1.09	0	0	19	0.45	917	0.14	0	0
2015	0	0	0	0	15	1.04	0	0	23	0.46	1021	0.25	0	0
2016	0	0	26	0.57	7	0.93	0	0	0	0	498	0.33	0	0
2017	0	0	26	0.40	0	0	0	0	16	0.24	930	0.16	0	0
2018	0	0	56	0.58	0	0	0	0	32	0.42	1113	0.32	1[a]	na
2019	0	0	22	0.53	18	0.40	0	0	30	0.37	2014	0.17	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 Gillnet	
Year	**M/SI**	**CV**	**M/SI**	**CV**	**M/SI**	**CV**
1994	0	0	0	0	861	0.58
1995	0	0	0	0	694	0.27
1996	0	0	0	0	89	0.55
1997	0	0	0	0	269	0.5
1998	17	1.02	0	0	78	0.48
1999	0	0	0	0	81	0.78
2000	0	0	0	0	24	1.57
2001	0	0	49	1.1	26	1.04
2002	0	0	0	0	0	0
2003	0	0	*	*	0	0
2004	0	0	0	0	303	0.3
2005	0	0	0	0	35	0.68
2006	0	0	0	0	65	0.66
2007	38	0.9	0	0	119	0.35
2008	176	0.74	0	0	238	0.38
2009	0	0	5	1.02	415	0.27
2010	0	0	0	0	253	0.61
2011	0	0	3	1.02	14	0.46
2012	0	0	0	0	0	0
2013	0	0	0	0	22	0.75
2014	0	0	0	0	57	0.42
2015	0	0	0	0	119	0.34
2016	0	0	0	0	85	0.50
2017	0	0	0	0	44	0.37
2018	0	0	0	0	14	0.80
2019	0	0	5.4	0.89	162	0.19

Note: This table only includes observed bycatch.
na=not applicable; unk= observer coverage was absent or too low to detect bycatch, or no estimate generated; tbd= to be determined

Appendix VI: Table C. Estimates of Human-caused Mortality Resulting from the *Deepwater Horizon* Oil Spill

Estimates of human-caused mortality are a result of a population model developed to estimate the injury and time to recovery for stocks affected by the *Deepwater Horizon* (DWH) oil spill, taking into account long-term impacts resulting from mortality, reproductive failure, reduced survival rates, and the proportion of the stock exposed to DWH oil (DWH MMIQT 2015).

	2010	2011	2012	2013	2014	2015	2016	2017	2018	2019	2020	2021	2022	2023	2024
Beaked Whales[a]	15.96	13.49	11.42	9.68	8.21	6.28	4.81	3.68	2.79	2.09	1.52	1.05	0.65	0.31	0
Common Bottlenose Dolphin, Oceanic Stock	96.55	81.93	69.71	59.39	50.63	38.86	29.86	22.88	17.40	13.03	9.48	6.54	4.06	1.91	0
Bryde's Whale	1.44	1.22	1.03	0.88	0.74	0.57	0.44	0.33	0.25	0.19	0.14	0.09	0.06	0.03	0
Clymene Dolphin	26.23	22.12	18.71	15.86	13.45	10.28	7.86	6.00	4.55	3.40	2.46	1.70	1.05	0.49	0
False Killer Whale	6.67	5.64	4.78	4.05	3.44	2.63	2.01	1.54	1.17	0.87	0.63	0.44	0.27	0.13	0
***Kogia* spp.**	111.92	91.48	75.08	61.80	50.98	37.92	28.27	21.04	15.56	11.33	8.03	5.40	3.27	1.50	0
Melon-headed Whale	29.33	24.83	21.04	17.84	15.13	11.56	8.85	6.76	5.13	3.83	2.78	1.92	1.19	0.56	0
Pantropical Spotted Dolphin	748.73	631.49	534.21	452.68	384.00	293.38	224.47	171.38	129.89	96.96	70.37	48.47	30.04	14.12	0
Pygmy Killer Whale	4.94	4.19	3.56	3.03	2.57	1.97	1.51	1.16	0.88	0.66	0.48	0.33	0.21	0.10	0
Risso's Dolphin	16.18	13.73	11.68	9.95	8.48	6.51	5.00	3.83	2.92	2.18	1.59	1.10	0.68	0.32	0
Rough-toothed Dolphin	113.72	96.50	82.11	69.96	59.64	45.78	35.18	26.96	20.50	15.35	11.17	7.72	4.79	2.26	0
Shelf Dolphins[b]	912.14	774.01	658.54	561.05	478.31	367.12	282.07	216.17	164.39	123.07	89.55	61.82	38.38	18.07	0
Short-finned Pilot Whale	10.79	9.13	7.73	6.56	5.56	4.25	3.25	2.49	1.88	1.41	1.02	0.71	0.44	0.21	0
Sperm Whale	29.82	25.12	21.20	17.90	15.14	11.53	8.79	6.70	5.07	3.78	2.74	1.89	1.17	0.55	0
Spinner Dolphin	352.31	297.15	251.37	213.01	180.70	138.05	105.63	80.65	61.13	45.63	33.12	22.82	14.14	6.65	0
Striped Dolphin	39.30	33.15	28.04	23.76	20.16	15.40	11.78	9.00	6.82	5.09	3.69	2.54	1.58	0.74	0

a. Beaked whales include Blainville's beaked whales, Gervais' beaked whales, and Cuvier's beaked whales
b. Shelf dolphins include common bottlenose dolphins and Atlantic spotted dolphins

DWH MMIQT [*Deepwater Horizon* Marine Mammal Injury Quantification Team]. 2015. Models and analyses for the quantification of injury to Gulf of Mexico cetaceans from the *Deepwater Horizon* Oil Spill, MM_TR.01_Schwacke_Quantification.of.Injury.to.GOM.Cetaceans. Southeast Fisheries Science Center, Protected Resources and Biodiversity Division, 75 Virginia Beach Dr., Miami, Florida 33140. PRBD Contribution #: PRBD-2020-02.