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U.S. Atlantic and Gulf of Mexico Marine Mammal Stock Assessments 2023

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 2023

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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 2022 publication. The 2023 revisions consist primarily of updated abundance estimates and/or revised human-caused mortality and serious injury (M/SI) estimates. A total of 31 Atlantic and Gulf of Mexico stock assessment reports were updated for 2023. This year, the NEFSC revised 17 stock assessment reports, and the SEFSC revised 14 reports. The revisions consist primarily of updated abundance estimates and revised human-caused mortality and serious injury (M/SI) estimates. One stock changed in status from “non-strategic” to “strategic”, the short-finned pilot whale. This particular stock has oscillated between strategic and non-strategic over the years, depending on the latest abundance and bycatch estimates. The Barataria Bay Estuarine System Stock of common bottlenose dolphins was revised at the request of the Atlantic Scientific Review Group in order to incorporate recently published references regarding the mid-Barataria sediment diversion and effects of the DWH oil spill. This stock remains strategic in status.

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 2023 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 2022 SARs. Guidelines were scheduled for further review in 2021, and revised guidelines (GAMMS IV) were published in early 2023 (NMFS 2023). NMFS began partial implementation of the revised guidelines for the 2023 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 June 2024 date-stamp at the top right corner at the beginning of each report.

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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 2017–2021[a]. 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	372	0	367	0.04	0.1	0.73	14.8	10.8 [b]	Y	2022	2023		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	2.05	1.45	Y	2021	2021		NEC
4	Sei whale	Nova Scotia	Y	6,292	1.02	3,098	0.04	0.1	6.2	0.6	0.4	Y	2021	2021		NEC
5	Minke whale	Canadian East Coast	Y	21,968	0.31	17,002	0.04	0.5	170	9.4	8.6	N	2021	2021		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	Y	5,895	0.29	4,639	0.04	0.1	9.28	0.2	0	Y	2019	2021		NEC
8	Dwarf sperm whale	Western North Atlantic	Y	9,474	0.36	7,080	0.04	0.4	57	unk	0.8	N	2019	2021	Estimates for *Kogia spp.*	SEC
9	Pygmy sperm whale	Western North Atlantic	Y	9,474	0.36	7,080	0.04	0.4	57	unk	0.8	N	2019	2021	Estimates for *Kogia spp.*	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	Y	unk	unk	unk	0.04	0.5	unk	0	0	N	2019	2021		SEC
12	False killer whale	Western North Atlantic	Y	1,298	0.72	755	0.04	0.5	7.6	0	0	N	2019	2021		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	Y	4,260	0.24	3,817	0.04	0.5	24	0.2	0	N	2019	2021		NEC
15	Blainville's beaked whale	Western North Atlantic	Y	2,936	0.26	2.374	0.04	0.5	81	0.2	0	N	2019	2021		NEC
16	Gervais beaked whale	Western North Atlantic	Y	8,595	0.24	7,022	0.04	0.5	70	0	0	N	2019	2021		NEC
17	Sowerby's beaked whale	Western North Atlantic	Y	492	0.50	340	0.04	0.5	3.4	0	0	N	2019	2021		NEC
18	True's beaked whale	Western North Atlantic	Y	4,480	0.34	3, 391	0.04	0.5	34	0	0	N	2019	2021		NEC
19	Melon-headed whale	Western North Atlantic	Y	unk	unk	unk	0.04	0.5	unk	0	0	N	2019	2021		SEC
20	Risso's dolphin	Western North Atlantic	Y	44,067	0.19	30,662	0.04	0.5	307	18	18 (0.09)	N	2021	2021		NEC
21	Pilot whale, long-finned	Western North Atlantic	Y	39,215	0.30	30,627	0.04	0.5	306	5.7	5.5 (0.29)	N	2021	2021		NEC
22	Pilot whale, short-finned	Western North Atlantic	Y	18,726	0.33	14,292	0.04	0.5	143	218	218 (0.19)	Y	2021	2021		SEC
23	Atlantic white-sided dolphin	Western North Atlantic	Y	93,233	0.71	54,443	0.04	0.5	544	28	28(0.19)	N	2021	2021		NEC
24	White-beaked dolphin	Western North Atlantic	N	536,016	0.31	415,344	0.04	0.5	4,153	0	0	N	2019	2016		NEC
25	Common dolphin	Western North Atlantic	Y	93,100	0.56	59,897	0.04	0.5	1,452	414	414 (0.10)	N	2021	2021		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.
26	Atlantic spotted dolphin	Western North Atlantic	Y	31,506	0.28	25,042	0.04	0.5	250	0	0	N	2019	2021		SEC
27	Pantropical spotted dolphin	Western North Atlantic	Y	2,757	0.50	1,856	0.04	0.5	19	0	0	N	2019	2021		SEC
28	Striped dolphin	Western North Atlantic	Y	48,274	0.29	38,040	0.04	0.5	529	0	0	N	2019	2021		NEC
29	Fraser's dolphin	Western North Atlantic	Y	unk	unk	unk	0.04	0.5	unk	0	0	N	2019	2021		SEC
30	Rough-toothed dolphin	Western North Atlantic	Y	unk	unk	unk	0.04	0.5	undet	0	0	N	2018	2021		SEC
31	Clymene dolphin	Western North Atlantic	Y	21,778	0.72	12,622	0.04	0.5	126	0	0	N	2019	2021		SEC
32	Spinner dolphin	Western North Atlantic	Y	3,181	0.65	1,930	0.04	0.5	19	0	0	N	2019	2021		SEC
33	Common bottlenose dolphin	Western North Atlantic, Offshore	Y	64,587	0.24	52,801	0.04	0.48	507	28	28 (0.43)	N	2019	2021	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
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

ID	Species	Stock Area	Updated this Year	Nest	Nest CV	Nmin	Rmax	Fr	PBR	Total Annual M/SI	Annual Fish. M/SI (CV)	Strategic Status	SAR of Last Update	Last Survey Year	Comments	NMFS Ctr.
36	Common bottlenose dolphin	Western North Atlantic, S. Carolina, Georgia Coastal	N	6,027	0.34	4,569	0.04	0.5	46	1.4–1.6	1.0–1.2	Y	2017	2016		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	2016		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	2016		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	2013		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	2006		SEC
41	Common bottlenose dolphin	Northern South Carolina Estuarine System	N	453	0.28	359	0.04	0.5	3.6	0.5	0.3	N	2022	2016		SEC
42	Common bottlenose dolphin	Charleston Estuarine System	N	unk	unk	unk	0.04	0.5	undet	2.2	1.8	Y	2022	2005, 2006		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.
43	Common bottlenose dolphin	Northern Georgia, Southern South Carolina Estuarine System	N	unk	unk	unk	0.04	0.5	unk	1.5	1.3	Y	2022	n/a		SEC
44	Common bottlenose dolphin	Central Georgia Estuarine System	N	unk	unk	unk	0.04	0.5	undet	0.4	0.2	Y	2022	2008, 2009		SEC
45	Common bottlenose dolphin	Southern Georgia Estuarine System	N	unk	unk	unk	0.04	0.5	undet	0.1	0.1	Y	2022	2008, 2009		SEC
46	Common bottlenose dolphin	Jacksonville Estuarine System	N	unk	unk	unk	0.04	0.5	unk	2.0	2.0	Y	2022	n/a		SEC
47	Common bottlenose dolphin	Indian River Lagoon Estuarine System	N	1,032	0.03	1,004	0.04	0.5	10	5.7	3.0	Y	2022	2016, 2017		SEC
48	Common bottlenose dolphin	Biscayne Bay	N	unk	unk	unk	0.04	0.5	unk	0.8	0.6	Y	2022	n/a		SEC
49	Harbor porpoise	Gulf of Maine, Bay of Fundy	Y	85,765	0.53	56,420	0.046	0.5	649	145	145 (0.18)	N	2021	2021		NEC
50	Harbor seal	Western North Atlantic	N	61,336	0.08	57,637	0.12	0.5	1,729	339	334 (0.09)	N	2021	2018		NEC
51	Gray seal	Western North Atlantic	Y	27,911	0.20	23,624	0.128	1.0	1,512	4,570	1,348 (0.12)	N	2021	2021		NEC
52	Harp seal	Western North Atlantic	N	7.6M	unk	7.1M	0.12	1.0	426,000	178,573	86 (0.16)	N	2021	2019		NEC
53	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

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54	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
55	Rice's whale	Gulf of Mexico	N	51	0.5	34	0.04	0.1	0.1	0.5	0	Y	2022	2017, 2018	Total M/SI is a minimum estimate and does not include Fisheries M/SI.	SEC
56	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
57	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
58	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
59	Common bottlenose dolphin	Gulf of Mexico, Continental Shelf	N	63,280	0.11	57,917	0.04	0.48	556	65	64.6	N	2021	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
60	Common bottlenose dolphin	Gulf of Mexico, Eastern Coastal	N	16,407	0.17	14,199	0.04	0.4	114	9.2	8.8	N	2021	2017, 2018		SEC
61	Common bottlenose dolphin	Gulf of Mexico, Northern Coastal	N	11,543	0.19	9,881	0.04	0.45	89	28	7.9	N	2021	2017, 2018		SEC
62	Common bottlenose dolphin	Gulf of Mexico, Western Coastal	N	20,759	0.13	18,585	0.04	0.45	167	36	32.4	N	2021	2017, 2018		SEC
63	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
64	Common bottlenose dolphin	Laguna Madre	N	80	1.57	unk	0.04	0.4	undet	0.8	0.2	Y	2021	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

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65	Common bottlenose dolphin	Neuces Bay, Corpus Christi Bay	N	58	0.61	unk	0.04	0.4	undet	0.2	0	Y	2021	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	Copano Bay, Aransas Bay, San Antonio Bay, Redfish Bay, Espiritu Santo Bay	N	55	0.82	unk	0.04	0.4	undet	0.6	0	Y	2021	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	Matagorda Bay, Tres Palacios Bay, Lavaca Bay	N	61	0.45	unk	0.04	0.4	undet	0.4	0	Y	2021	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	West Bay	N	37	0.05	35	0.04	0.4	0.3	0	0	N	2021	2014, 2015		SEC
69	Common bottlenose dolphin	Galveston Bay, East Bay, Trinity Bay	N	842	0.08	787	0.04	0.4	6.3	1.0	0.4	N	2021	2016		SEC
70	Common bottlenose dolphin	Sabine Lake	N	122	0.19	104	0.04	0.45	0.9	0	0	N	2021	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

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71	Common bottlenose dolphin	Calcasieu Lake	N	0	-	-	0.04	0.45	undet	0.2	0.2	Y	2021	1992	Details for this stock are included in the collective report: Common bottlenose dolphin (*Tursiops truncatus truncatus*), Northern Gulf of Mexico Bay, Sound, and Estuary Stocks.	SEC
72	Common bottlenose dolphin	Vermilion Bay, West Cote Blanche Bay, Atchafalaya Bay	N	0	-	-	0.04	0.45	undet	0	0	Y	2021	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	Terrebonne, Timbalier Bay Estuarine System	N	3,870	0.15	3,426	0.04	0.4	27	0.2	0	N	2018	2016		SEC
74	Common bottlenose dolphin	Barataria Bay Estuarine System	Y	2,071	0.06	1,971	0.04	0.45	18	35	0.2	Y	2021	2019		SEC
75	Common bottlenose dolphin	Mississippi River Delta	N	1,446	0.19	1,238	0.04	0.4	11	9.2	0.2	N	2021	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
76	Common bottlenose dolphin	Mississippi Sound, Lake Borgne, Bay Boudreau	N	1,265	0.35	947	0.04	0.45	8.5	59	2.0	Y	2021	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.
77	Common bottlenose dolphin	Mobile Bay, Bonsecour Bay	N	122	0.34	unk	0.04	0.45	undet	16.0	1.0	Y	2021	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
78	Common bottlenose dolphin	Perdido Bay	N	0	-	-	0.04	0.4	undet	0.8	0.6	Y	2021	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	Pensacola Bay, East Bay	N	33	0.80	unk	0.04	0.4	undet	0.4	0.2	Y	2021	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	Choctawhatchee Bay	N	179	0.04	unk	0.04	0.5	undet	0.4	0	Y	2015	2007		SEC
81	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
82	Common bottlenose dolphin	St. Joseph Bay	N	142	0.17	123	0.04	0.4	1.0	unk	unk	N	2019	2011		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.
83	Common bottlenose dolphin	St. Vincent Sound, Apalachicola Bay, St. George Sound	N	439	0.14	unk	0.04	0.4	undet	0.2	0.2	Y	2021	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
84	Common bottlenose dolphin	Apalachee Bay	N	491	0.39	unk	0.04	0.4	undet	0	0	Y	2021	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
85	Common bottlenose dolphin	Waccasassa Bay, Withla-coochee Bay, Crystal Bay	N	unk	-	unk	0.04	0.4	undet	0.4	0.4	Y	2021	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
86	Common bottlenose dolphin	St. Joseph Sound, Clearwater Harbor	N	unk	-	unk	0.04	0.4	undet	0.8	0.4	Y	2021	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	Tampa Bay	N	unk	-	unk	0.04	0.4	undet	3.0	2.2	Y	2021	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	Sarasota Bay, Little Sarasota Bay	N	158	0.27	126	0.04	0.4	1.0	0.2	0.2	N	2021	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
89	Common bottlenose dolphin	Pine Island Sound, Charlotte Harbor, Gasparilla Sound, Lemon Bay	N	826	0.09	unk	0.04	0.4	undet	1.0	0.6	Y	2021	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
90	Common bottlenose dolphin	Caloosa-hatchee River	N	0	-	-	0.04	0.4	undet	0.4	0.2	Y	2021	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
91	Common bottlenose dolphin	Estero Bay	N	unk	-	unk	0.04	0.4	undet	0.4	0.2	Y	2021	n/a	Details for this stock are included in the collective report: Common bottlenose dolphin (*Tursiops truncatus truncatus*), Northern Gulf of Mexico Bay, Sound, and Estuary Stocks.	SEC
92	Common bottlenose dolphin	Chokoloskee Bay, Ten Thousand Islands, Gullivan Bay	N	unk	-	unk	0.04	0.4	undet	0.2	0.2	Y	2021	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.
93	Common bottlenose dolphin	Whitewater Bay	N	unk	-	unk	0.04	0.4	undet	0	0	Y	2021	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	Florida Bay	N	unk	unk	unk	0.04	0.5	unk	0.2	0.2	N	2022	2003		SEC
95	Common bottlenose dolphin	Florida Keys (Bahia Honda to Key West)	N	unk	-	unk	0.04	0.4	undet	0.2	0.2	Y	2021	n/a	Details for this stock are included in the collective report: Common bottlenose dolphin (*Tursiops truncatus truncatus*), Northern Gulf of Mexico Bay, Sound, and Estuary Stocks.	SEC
96	Atlantic spotted dolphin	Gulf of Mexico	N	21,506	0.26	17,339	0.04	0.48	166	36	36 (0.47)	N	2021	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
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

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.
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
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

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.
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

a. Period for North Atlantic right whales is 2018-2022.
b. Total annual average observed North Atlantic right whale mortality during the period 2018–2022 was 5.45 animals and annual average observed fishery mortality was 3.95 animals. Numbers presented in this table (14.8 total mortality and 10.8 fishery mortality) are 2018–2022 estimated annual means, accounting for undetected mortality and serious injury.

NORTH ATLANTIC RIGHT WHALE (*Eubalaena glacialis*)

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 19th-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; NMFS Southeast Regional Office unpublished data) and a lone calf documented off the Canary Islands in 2020 (North Atlantic Right Whale Catalog, unpublished data) represent individuals beyond the primary calving and wintering ground in the waters of the southeastern U.S. East Coast.

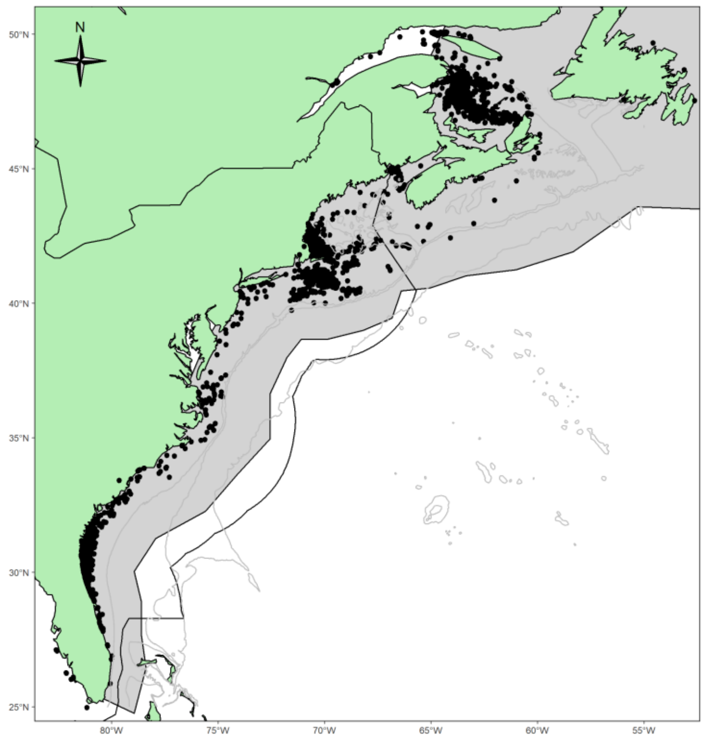


Figure 1. Approximate range (shaded area) and distribution of sightings (dots) of known North Atlantic right whales 2018-2023. Data from North Atlantic Right Whale Consortium database (https://www.narwc.org/narwc-databases.html, accessed 13 September, 2024) and NMFS unpublished data.

Although the location of much of the population is unknown during much of the year, passive acoustic studies have demonstrated year-round presence of right whales on the Scotian Shelf (Durette-Morin et al. 2022), in the Gulf of Maine (Morano et al. 2012; Bort et al. 2015), and off southern New England (Estabrook et al. 2022), New York (Murray et al. 2022), 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) pooled together acoustic detections from a large number of passive acoustic recorders and documented broad-scale use of the U.S. eastern seaboard during much of the year, with widespread right whale acoustic occurrence in winter months from Florida to the southern Scotian Shelf. Right whales occurred across the dataset (spanning 2004–2014) from Florida to southern Greenland. Since 2015, acoustic monitoring networks along the East Coast continue to show year round presence from Cape Hatteras, North Carolina to Massachusetts Bay, Massachusetts with a peak in detections south of New England in winter months (Passive Acoustic Cetacean Map (PACM; https://apps-nefsc.fisheries.noaa.gov/pacm/#/narw)). In Canada, large scale passive acoustic studies documented right whales in the Gulf of St. Lawrence (Simard et al. 2019) and Atlantic Canadian waters (Durette-Morin et al. 2022). Right whales were acoustically detected every year in the Gulf of St Lawrence from 2010–2018; the earliest seasonal detections were at the end of April, lasting until mid-January (Simard et al. 2019, Durette-Morin et al. 2022). Among the recorder locations in the Gulf of St. Lawrence, detections occurred in the southern Gulf to the Strait of Belle Isle, and daily detection rates quadrupled at listening stations off the Gaspé

Peninsula beginning in 2015 (Simard et al. 2019, Durette-Morin et al. 2022). Right whales were detected in Atlantic Canadian waters from the Bay of Fundy, to Cabot Strait, to Southern Newfoundland, but were not detected in the Labrador Sea and Newfoundland Shelf during extensive acoustic monitoring throughout the Atlantic Canadian continental shelf between 42°N and 58°N during 2015 through 2017 (Durette-Morin et al. 2022). Recently developed monthly habitat-based density estimates of right whales for U.S. waters and a portion of southern Canadian waters show strong correlation with acoustic detection rates (Roberts et al. 2024).

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 and along the US east coast, over short timeframes (Mate et al. 1997; Baumgartner and Mate 2005, Aschettino et al. 2022, 2023). The majority of right whale sightings off northeastern Florida and southeastern Georgia were within 90 km of the shoreline, as was most of the survey effort, however, one sighting occurred ~140 km offshore (NMFS unpub. data).

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 her maturation. An offshore survey in March 2010 observed the birth of a right whale in waters 75 km off Jacksonville, Florida (Foley et al. 2011). In 2016, the Southeastern U.S. Calving Area Critical Habitat was expanded north to Cape Fear, North Carolina (81 FR 4837, 26 February 2016). There is also at least one case of a calf apparently being born in the Gulf of Maine (Patrician et al. 2009) and another calf 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, Sorochan et al. 2021). 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), Baumgartner and Mate (2003), and Ross et al. (2023). 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).

Both visual and acoustic monitoring detected an important change in right whales' seasonal residency patterns beginning in 2010, with reduced right whale presence in the Bay of Fundy and Gulf of Maine (Davis et al. 2017; Davies et al. 2019, Meyer-Gutbrod et al. 2021). Between 2012 and 2016, visual surveys in the Great South Channel also saw a sharp decline in right whale sightings (Khan et al. 2018), while the number of individuals using Cape Cod Bay in spring increased (Mayo et al. 2018; Ganley et al. 2019, Meyer-Gutbrod et al. 2023). Right whale aggregations in the central Gulf of Maine in winter (Cole et al. 2013) have also not been detected since 2011 (NMFS unpublished data), although the species is detected acoustically every year in the Gulf of Maine (Davis and Van Parijs 2023; PACM 2024). Additionally, large numbers of right whales have been documented feeding and socializing south of Martha's Vineyard and Nantucket Islands (Leiter et al. 2017; Stone et al. 2017; Quintana-Rizzo et al. 2021; O'Brien et al. 2022), an area outside of the 2016 Northeastern U.S. Foraging Area Critical Habitat. Right whale presence in this area is nearly year round, including in summer months. The highest sighting rates in this area are between December and May, when close to a quarter of the population may be present at any given time. The age and sex of the whales using this area did not vary significantly from that of the population (Quintana-Rizzo et al. 2021). 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; Simard et al. 2019; DFO 2020). 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). Individuals utilizing the Gulf of St. Lawrence foraging habitat exhibit site fidelity (Crowe et al. 2021), and individual variation in the use of this habitat is partially explained by maternal lineage (Bishop et al. 2022).

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 the understanding of genetic variability, the number of reproductively active individuals, reproductive fitness, parentage, and relatedness of individuals (Frasier et al. 2007, 2009). It has also helped fill gaps in our understanding of the species' age structure, calf development, calf survival, and weaning (Hamilton et al. 2023). Because the callosity patterns used to identify individual right whales take months to develop after a whale's birth, obtaining biopsy samples from calves on the calving grounds provides a means of genetically identifying calves later in life or after death. 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 most reliable way to establish parentage, if the calf is not sampled when associated with its mother early on, information such as age and familial relationships may be 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). Hamilton et al. (2022) reported that of the 470 calves observed between 1998 and 2018, 370 (78.7%) were biopsied, 293 as calves and 77 later in life, their identification linked by photographs. Of the 100 calves not biopsied during this period, 32 were sufficiently photographed to allow subsequent identification and aging, but 68 had yet to be identified other than as a unique calf.

Frasier (2007b) genetically examined the paternity of 87 calves born between 1980 and 2001. Although genetic profiles were available for 69% of all potential fathers in the population, paternity was assigned to only 51% of the calves, and all the sampled males were excluded as fathers of the remaining calves. The findings suggested that either the unsampled males were particularly successful or that the population of males, and the population as a whole, was larger than suggested by the photo-identification data (Frasier 2007b). However, a 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

Estimation of the western North Atlantic right whale stock size is based on a state-space model of the sighting histories of individual whales identified using photo-identification techniques (Pace et al. 2017; Pace 2021) with an accommodation of potential recruits based on observed calves (Linden 2024a,b). Population size was estimated using sighting histories constructed from the central photo-ID recapture database (curated at the New England Aquarium) as it existed on 13 September 2024, and included photographic information from all dedicated survey teams in the US and Canada up through 30 November 2023. Using a hierarchical, state-space Bayesian open population model of these histories (Linden 2024b) produced a median abundance value (Nest) in 2023 of 372 individuals (95%CI: 360-383; Table 1). Typically this model has relied on individual animals being photographically identifiable from their callosity patterns to be recruited into the population, which are typically not stable until animals are greater than 1 year old. However, a recent model development has directly addressed this challenge and individuals less than 1 year old are now included in the abundance estimate (Linden 2024a). 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 may better characterize that uncertainty (as opposed to a CV that may appear in other stock assessment reports).

*Table 1. Best and minimum abundance estimates in 2023 for western North Atlantic right whales (***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
372	360–383	367–377	367	0.1	0.04	0.73

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 right whale 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.

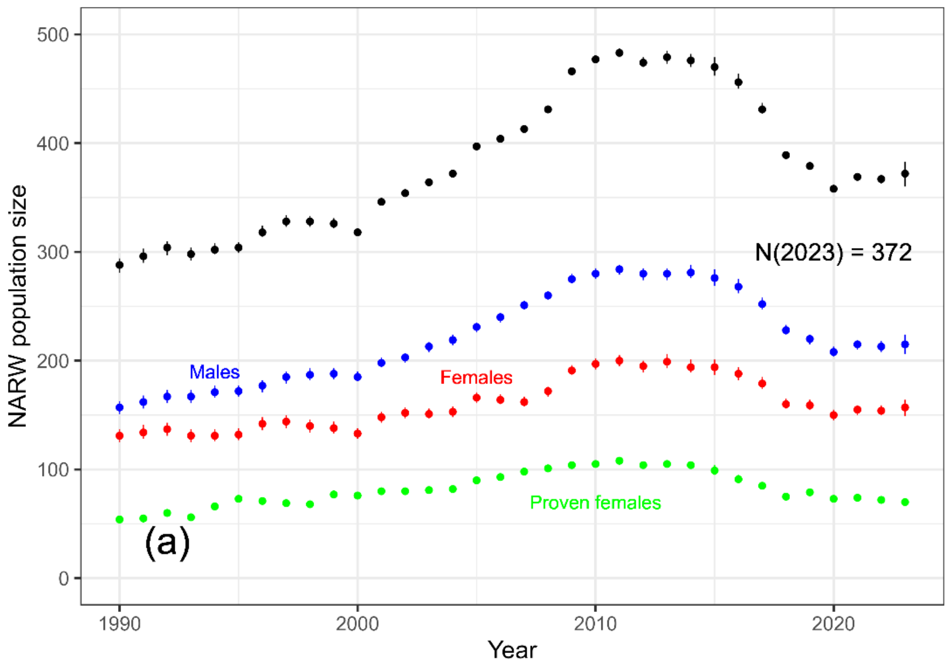
Minimum Population Estimate

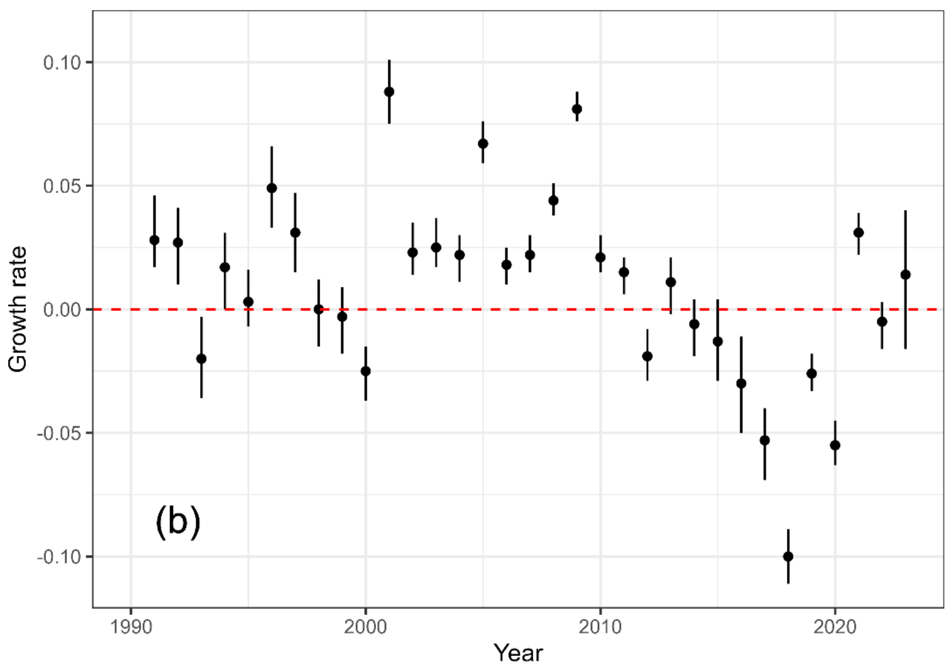
The minimum population estimate is the lower limit of the two-tailed 60% credible interval about the median of the posterior abundance estimates using the methods of Pace et al. (2017) and refinements of Pace (2021) and Linden (2024a). This is roughly equivalent to the 20th percentile of the log-normal distribution as specified by Wade and Angliss (1997). The median estimate of abundance for adult and subadult western North Atlantic right whales is 372, and the minimum population estimate is 367 individuals (based on photographic information collected through 30 November 2023; Table 1).

Current Population Trend

The population growth rate reported for the period of 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 the 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 2.6% per annum from posterior median point estimates of 288 individuals in 1990 to 483 in 2011. There was a 100% chance that abundance declined from 2011 to 2020 when the final estimate was 358 individuals. The overall abundance decline between 2011 and 2020 was 26% (derived from 2011 and 2020 median point estimates). 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. This pattern persists in Linden (2024b), though survival has increased since 2020 (Figure 2c). The 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 in the mid-2010s (Figure 3). However, the methods in Pace et al. (2017) and Linden (2024b) account for changes in capture probability.


NARW population size
N(2023) = 372
Males
Females
Proven females
(a)
Year


Growth rate
(b)
Year

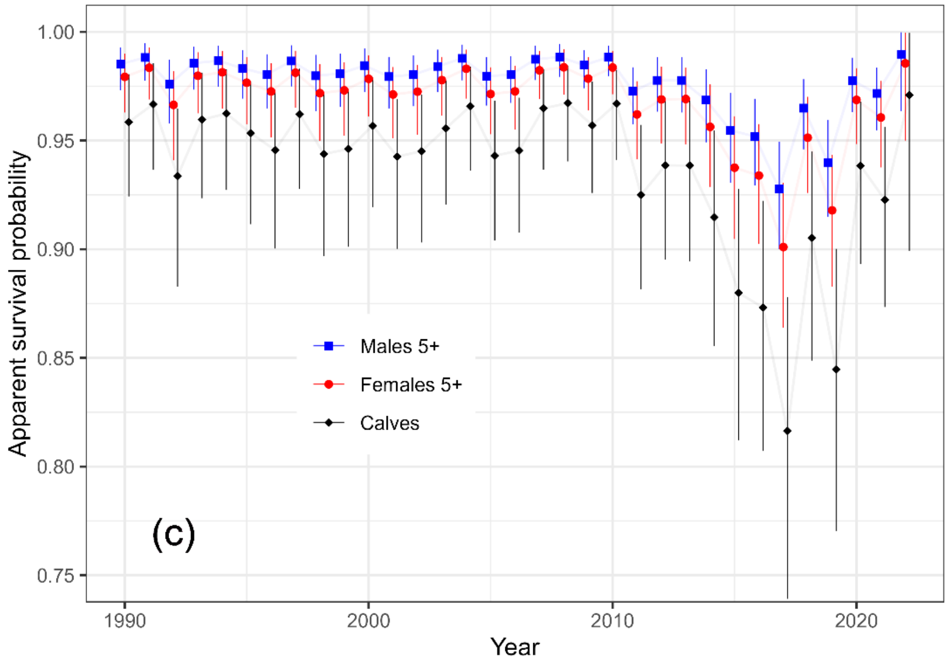


Figure 2. (a) Abundance estimates for North Atlantic right whales. Estimates are the median values of a posterior distribution from modeled capture histories. Also shown are sex-specific abundance estimates, including estimates for both adult females and females of all ages. (b) Annual population growth rates from the abundance values. (c) Sex-specific survival rate estimates. All graphs show associated 95% credible intervals.

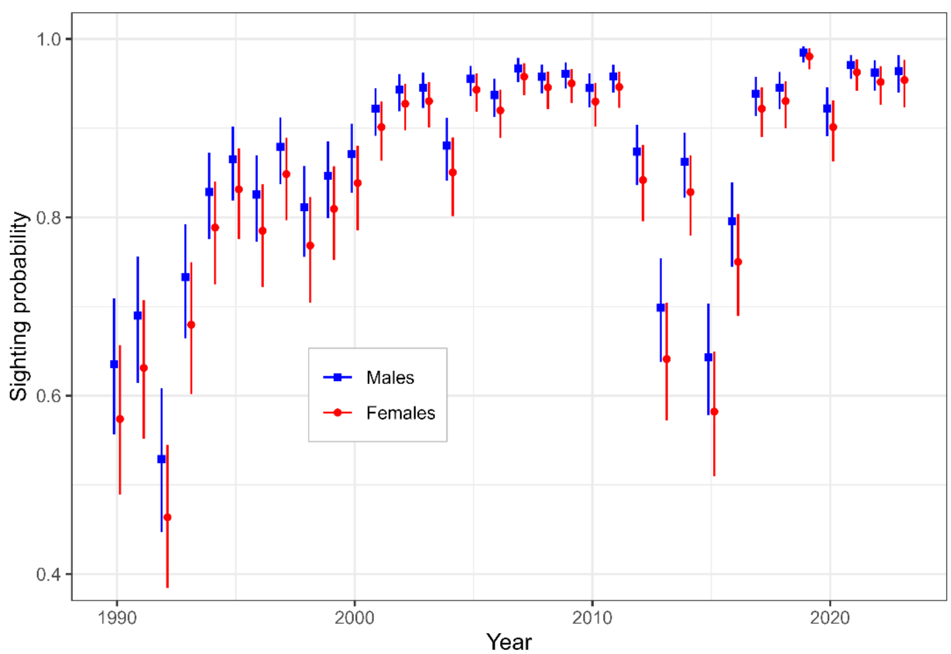


Figure 3. Estimated recapture probability and associated 95% credible intervals of North Atlantic right whales 1990–2023 based on a Bayesian mark-resight/recapture model allowing random fluctuation among years for survival rates, treating capture rates as fixed effects over time, and using both observed and known states as data (from Linden 2024b). Males are shown in blue with squares; females are shown in red with circles.

CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

Knowlton et al. (1994) reported that during 1980–1992, 145 calves were born to 65 identified females (not including six documented neonate mortalities), and the number of calves born annually ranged from 5 to 17, with a mean of 11.2. The mean calving interval, based on 86 records from 1976-1992, 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). While the pool of reproductively active females climbed from 1980 to 1986 as photographic effort captured mothers new to the study, it became static at approximately 51 individuals from 1987–1992 (Knowlton et al. 1994). Since 1993, calf production has been more variable than a simple stochastic model would predict.

During 1990–2023, at least 518 calves were born into the population (including neonate mortalities). The number of calves born annually ranged from 0 to 39 with a mean of 14.9 (SD=8.7).

Population productivity is indexed by dividing the number of detected calves by the estimated abundance each year (Apparent Productivity Index [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 rate 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%; Corkeron et al. 2018). Based on the most recent population estimate, the number of females known to have calved that are likely still alive is 70 [95% CI: 62, 78] (Linden 2024b).

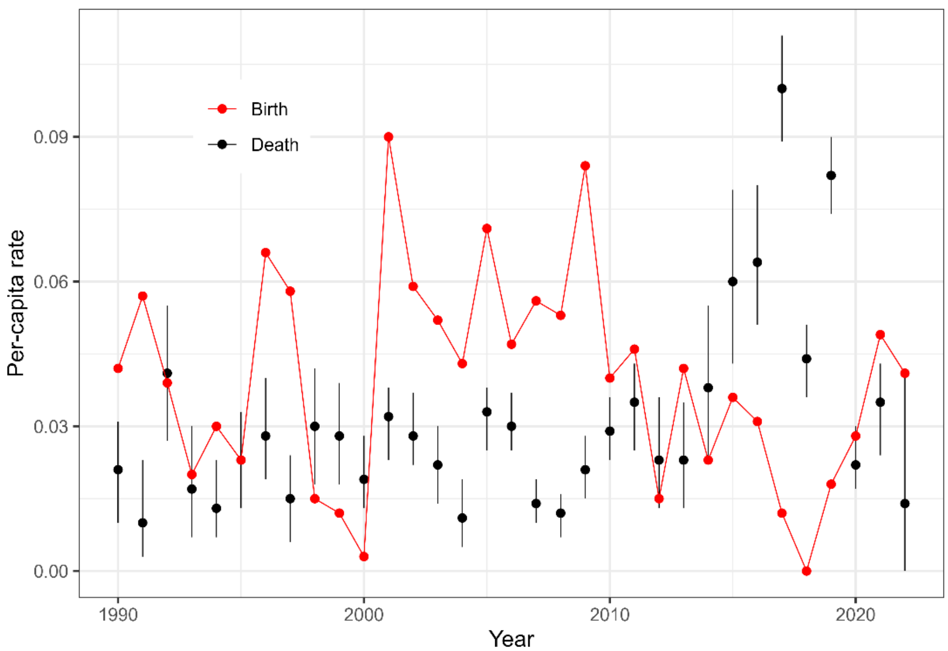


Figure 4. North Atlantic right whale per capita death rate and birth rate (red line, closed circles) with associated 95% credible intervals, 1990–2022.

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; Knowlton et al. 2022). There is also clear evidence that North Atlantic right whales are growing to shorter adult lengths than in earlier decades (Stewart et al. 2021) and are in poor body condition compared to southern right whales (Christiansen et al. 2020, Miller et al. 2011), as well as compared to the population’s body condition in the past (Knowlton et al. 2022). Stewart et al. (2022) found that smaller females have longer inter-birth intervals than larger females. All these changes may result from a combination of documented regime shifts in primary feeding habitats (Meyer-Gutbrod and Greene 2014; Meyer-Gutbrod et al. 2021; Meyer-Gutbrod et al. 2022; Record et al. 2019) and increased energy expenditures related to non-lethal entanglements (Rolland et al. 2016; Pettis et al. 2017; van der Hoop et al. 2017). Despite management actions, overall entanglement rates as measured by the rate at which scars are acquired by living North Atlantic right whales (Hamilton et al. 2020; Figure 5) remain high.

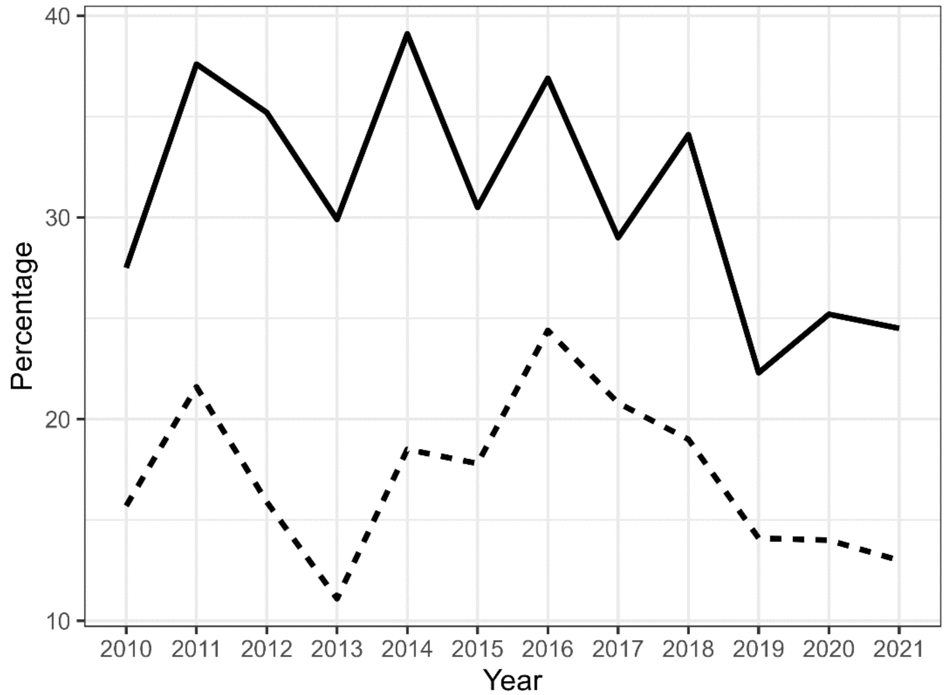


Figure 5. North Atlantic right whale entanglement rates estimated by monitoring scars on living whales. The crude entanglement rate (dashed line) is the proportion of whales seen with newly discovered entanglement scars; the year the scar was detected may not represent the year the entanglement occurred. The annual entanglement rate (solid line) is the minimum rate of entanglement, derived from the proportion of whales with new scars that were adequately photographed in both years of sequential combinations (e.g., 2017/2018; data from Hamilton et al. 2023).

An analysis of the age structure of this population found that it contained a smaller proportion of juvenile whales than expected, only 26–31%, which may reflect lowered recruitment and/or high juvenile mortality (Hamilton et al. 1998; IWC 2001). By 2022, only 14.5% of the whales presumed alive were confirmed juveniles (Hamilton et al. 2023). Calf and perinatal mortality was estimated by Browning et al. (2010) to be between 17 and 45 animals during the period 1989 and 2003. It is possible that the apparently low reproductive rate for this species 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, or threatened stocks, or stocks of unknown status relative to OSP (MMPA Sec. 3. 16 U.S.C. 1362; Wade and Angliss 1997). The recovery factor for right whales is 0.1 because this species is listed as endangered under the Endangered Species Act (ESA). The minimum population size is 367. 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 0.73 (Table 1).

ANNUAL HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

For the period 2018 through 2022, the annual detected (i.e., observed) human-caused mortality and serious injury to right whales averaged 5.45 individuals per year (Table 2). This is derived from two components: 1) incidental fishery entanglement records at 3.95 per year and 2) vessel strike records averaging 1.5 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. 2024). 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 for adults and juveniles; the estimates are exclusive to those individuals old enough to enter the sightings catalog (>0.5 years of age). The estimated rate of total non-calf mortality using this modeling approach is 14.8 animals (non-calves) per year, or 74 animals total, for the period 2018–2022 (Linden 2024b). 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 6 shows the estimates of total mortality for 1990–2022 using the state-space model. The model's estimated 14.8 total mortality rate for the 5-year period 2018–2022 is 2.7 times higher than the 5.45 *detected* mortality and serious injury value reported for the same period. An analysis of right whale mortalities between 2003 and 2018 found that of the 33 examined non-calf carcasses for which cause of death could be determined, all mortality was human-caused (Sharp et al. 2019). Based on these findings, 100% of the estimated mortality of 14.8 animals (non-calves) per year is assumed to be human-caused. Sharp et al. (2019) found that 5 of 10 (50%) calf mortalities were from natural causes.

There is currently insufficient information to apportion the estimated total right whale mortality to that occurring in U.S. waters. To apportion the estimated total right whale mortality by cause, e.g., entanglement versus vessel collision, we used the proportion of observed mortalities and serious injuries from entanglement compared to those from vessel collision for the period 2018–2022. During this period, 72% of the observed mortality and serious injury was the result of entanglement and 28% was from vessel collisions. Applying these proportions to the estimated total mortality of adults and juveniles provides an estimate of 54 total entanglement deaths and 20 total vessel collision deaths during 2018–2022 (Table 2). These estimates may be biased if there is significant bias in the detection of entanglement versus vessel collision 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 the 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 Sharp et al. (2019) theorized 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. However, floating carcasses of whales, which move only by wind and currents, may not be carried into areas where detection is likely, whereas entangled whales may continue to swim and carry gear for days to years (see van der Hoop et al. 2017) and move into areas patrolled by survey teams. Based on records of mortalities and serious injuries maintained by the NMFS Greater Atlantic and Southeast Regional Offices between 2001–2020, 59% of all right whale serious injuries were first documented by survey teams, whereas only 19% of right whale carcasses were first discovered by survey teams. The visibility of some entanglements may add to the likelihood of serious injury detection, whereas blunt trauma from a vessel collision may not be externally detectable. 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. Annual estimated and observed human-caused mortality and serious injury for the North Atlantic right whale (*Eubalaena glacialis*). Estimated total mortality is derived from annual population estimates for adults and juveniles from 2018–2022 (Linden 2024b). Observed values are from confirmed interactions from 2018–2022.

Years	Source	Total	Annual Average
2018–2022	**Estimated total adult and juvenile mortality**	**74**	**14.8**
	Estimated adult and juvenile incidental fishery-related mortality	**54**	**10.8**
	Estimated adult and juvenile vessel collision mortality	**20**	**4.0**
2018–2022	Detected total human-caused M/SI[a]	27.25	5.45
	Detected incidental fishery-related M/SI[a,b]	19.75	3.95
	Detected vessel collision M/SI[a]	7.5	1.5
	Fishery-related SI prevented[c]	2	0.4

a. Observed serious injury events with decimal values were counted as 1 for this comparison.
b. The observed incidental fishery interaction count does not include fishery-related serious injuries that were prevented by disentanglement.
c. Fishery-related serious injuries prevented are a result of successful disentanglement efforts.

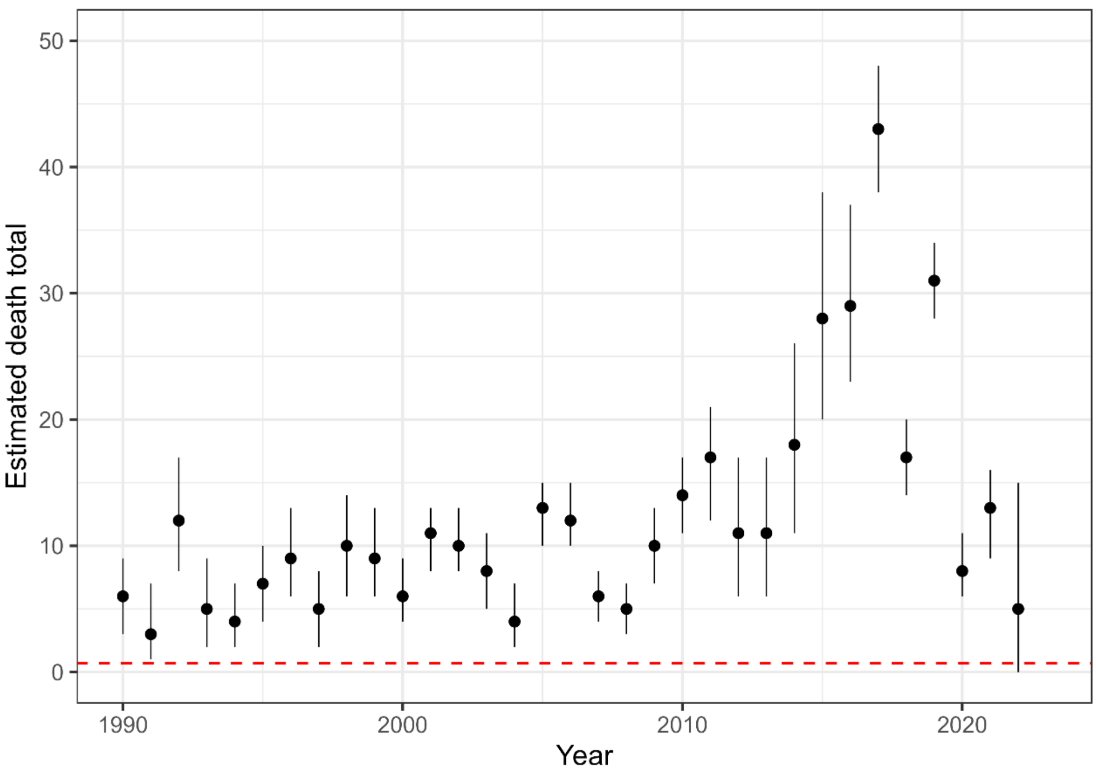


Figure 6. Time series of estimated total right whale mortalities, 1990–2022. Red dashed line indicates Potential Biological Removal (PBR) level.

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 whale species (Corkeron et al. 2018). The principal factors preventing growth and recovery of the population are entanglement and vessel strikes. Between 1970 and 2018, 124 right whale mortalities were recorded (Knowlton and Kraus 2001; Moore et al. 2005; Sharp et al. 2019). Of these, 18 (14.5%) were calves 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.

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 (e.g., 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 travel great distances before being detected (see van der Hoop et al. 2017; Morin et al. 2020), and recovered gear is often not adequately marked, it is difficult to assign most entanglements to the country of origin. It is not known how the disruption of survey efforts by COVID-19 virus precautions may have impacted the detection of serious injuries or mortalities in 2020 and 2021.

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; Christiansen et al. 2020; Stewart et al. 2021). 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. 2024). 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. These 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 the 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. Knowlton et al. (2012) reviewed scarring on identified individual right whales over a period of 30 years (1980–2009), documenting 1,032 definite, unique entanglement events on the 626 individual whales. 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. Moore et al. (2021) reported that between 1980 and 2017, 86.1% (642 of 746) individual whales identified had evidence of entanglement interactions. Analysis 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).

Analyses of entanglement trends indicate that mitigation measures implemented prior to 2010 had not been effective at reducing large whale mortality. Knowlton et al. (2012) concluded from their analysis of right whale entanglement scarring rates from 1980–2009 that management efforts of the prior decade had not reduced right whale encounters with gear, and that the rate of serious entanglements (whales bearing gear or with a cut deeper than 8cm) had increased. Using observed mortalities of eight large whale species from 1970–2009, van der Hoop et al. (2013) found an increasing trend in entanglement mortality despite regulatory efforts. Pace et al. (2014), analyzing entanglement rates and serious injuries due to entanglement of four large whale species during 1999–2009, found an increase in annual entanglement rates but no significant trend in entanglement-related mortality, indicating that mitigation measures implemented prior to 2009 had not been effective at reducing large whale mortality due to commercial fishing. Since 2009, new entanglement mitigation measures (72 FR 193, 05 October 2007; 79 FR 124, 27 June 2014; 86 FR 51970, 17 September 2021; 87 FR 11590, 02 March 2022) have been implemented as part of the Atlantic Large Whale Take Reduction Plan, but their effectiveness has yet to be formally 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 of vessel strike mortality and serious injury to right whales from 2017 through 2021 are summarized in Table 3. Researchers have identified increasing vessel speed as a factor in lethal vessel strike events involving whales (Vanderlaan and Taggart 2007) and inferred a strong relationship between vessel speed and the likelihood of interactions (Conn and Silber, 2013). Using simple biophysical models, Kelley et al. (2020) determined that whales can be seriously injured or killed by vessels of all sizes and that a collision with a 50-ton vessel transiting at seven knots has a probability of lethality greater than 50%.

In 2008, NOAA Fisheries implemented the North Atlantic right whale vessel speed regulations (50 CFR 224.105) in an effort to reduce vessel strike mortality. Since this rule was established, there have been several evaluations of vessel compliance with the rule and its effectiveness at reducing vessel strikes of right whales (Silber and Bettridge 2012, Laist et al. 2014, van der Hoop et al. 2015, Hayes et al., 2018). Most recently, NMFS (2020) found that vessel compliance with the speed rule varied across Seasonal Management Areas with apparent compliance during the 2018-2019 season reaching 81% coastwide. In August 2022, NMFS proposed substantial changes to the speed rule to further reduce ongoing lethal vessel strikes of right whales in U.S. waters, which was supported by a coast wide vessel strike risk assessment (Garrison et al. 2022).

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. There were 34 dead whales documented through December 2022: 11 vessel strike, 9 entanglement, 1 perinatal, 3 unknown, and 10 unexamined. Additionally, 30 free-swimming whales were documented as being seriously injured (2 vessel strike, 27 entanglement, and 1 abandoned dependent calf) and 45 morbidity cases were documented with sublethal injuries and/or illness (3 vessel strike, 35 entanglement, 2 unknown injury, and 5 poor body condition). The UME entanglement serious injury tallies include events where serious injury was averted by disentanglement and abandoned dependent calves Therefore, some of the serious injuries listed in the UME are not captured in Table 3. The latest UME updates are available at (https://www.fisheries.noaa.gov/national/marine-life-distress/2017-2024-north-atlantic-right-whale-unusual-mortality-event).

Table 3. Observed human-caused mortality and serious injury records of right whales: 2018–2022[a]

Date[b]	Fate	ID	Location[b]	Assigned Cause	Value against PBR[c]	Country[d]	Gear Type[e]	Description
01/22/2018	Mortality	3893	off 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. Gear consistent with a portion of a Canadian snow crab set.
02/15/2018	Serious Injury	3296	off 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	Gulf of St Lawrence, QC	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. Entangled in Canadian gear of unknown origin.

Date[b]	Fate	ID	Location[b]	Assigned Cause	Value against PBR[c]	Country[d]	Gear Type[e]	Description
07/30/2018	Prorated Injury	3843	off Grand Manan, NB	EN	0.75	XC	GU	Free-swimming with buoy trailing 70 ft 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	off Nantucket, MA	EN	1	XU	NP	No gear present, but evidence of constricting wraps across ventral surface and at pectorals. COD - acute, severe entanglement.
12/1/2018	Serious Injury	3208	off Nantucket, MA	EN	1	XU	NP	No gear present. Evidence of new, healed, constricting body wrap. Health decline evident - gray, lesions, thin. Previously reported as 24Dec2018
12/20/2018	Prorated Injury	2310	off 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
6/4/2019	Mortality	4023	Gulf of St Lawrence, QC	VS	1	CN	-	Abrasion, blubber hemorrhage, and muscle contusion caudal to blowholes consistent with pre-mortem vessel strike
6/20/2019	Mortality	1281	Gulf of St Lawrence, QC	VS	1	CN	-	Sharp trauma penetrating body cavity consistent with vessel strike. Vessel >65 ft based on laceration dimensions.
6/25/2019	Mortality	1514	Gulf of St Lawrence, QC	VS	1	CN	-	Fractured ear bones, skull hemorrhaging, and jaw contusion consistent with blunt trauma from vessel strike.
6/27/2019	Mortality	3450	Gulf of St Lawrence, QC	VS	1	CN	-	Hemothorax consistent with blunt force trauma.
7/4/2019	Serious Injury	3125	Gulf of St Lawrence, 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: 200-300 ft of line removed. Embedded rostrum and blowhole wraps remain, but now able to open mouth. Significant health decline. No resights. Gear consistent with a portion of a Canadian snow crab set.

Date[b]	Fate	ID	Location[b]	Assigned Cause	Value against PBR[c]	Country[d]	Gear Type[e]	Description
8/6/2019	Mortality	1226	Gulf of St Lawrence, QC	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. Entangled in line of Canadian origin.
1/8/2020	Serious Injury	5010	off Altamaha Sound, GA	VS	1	US	-	Dependent calf with deep lacerations to head and lips, exposing bone. No resights post 15Jan2020.
2/24/2020	Serious Injury	3180	off Nantucket, MA	EN	1	XU	NR	Free-swimming with bullet buoy lodged in right mouthline, far forward. Line seen exiting left gape. No trailing gear visible. Poor condition - emaciated with heavy cyamid load. No resights.
3/16/2020	Prorated Injury	-	Georges Bank, US EEZ	EN	0.75	XU	NR	Free-swimming with 2 polyballs trailing approximately 30 ft aft of flukes. Attachment point(s) and full configuration unknown. No resights
6/24/2020	Mortality	5060	off Elberon, NJ	VS	1	US	-	Dependent calf with deep lacerations along head and peduncle from 2 separate vessel strikes. Head lacerations were chronic and debilitating while the laceration to peduncle was acutely fatal. Proximate COD - sharp and blunt vessel trauma. Ultimate COD - hemorrhage and paralysis.
10/11/2020	Serious Injury	4680	2.7 nm E off Sea Bright, NJ	EN	1	XU	NR	Free-swimming with 2 lines embedded in rostrum, remaining configuration unknown. Extremely poor condition - emaciated with greying skin. Large, open lesion on left side of head. No resights.
10/19/2020	Mortality	3920	off Nantucket, MA	EN	1	CN	PT	Free-swimming with deeply embedded rostrum wrap. Partial disentanglement - removed 100 ft of trailing line and attached telemetry. Health deteriorated over subsequent sightings - emaciation, increased cyamid load, sloughing skin. Carcass documented on 27Feb2021 off Florida. No necropsy conducted but COD from chronic entanglement most parsimonious. Gear consistent with Canadian snow crab.
1/11/2021	Serious Injury	1803	off Fernandina Beach, FL	EN	1	XU	NR	Free-swimming with constricting wraps at peduncle and fluke insertion and around left fluke blade. No resights post 12Jan2021.
2/12/2021	Mortality	5130	St. Augustine, FL	VS	1	US	-	Dependent calf. 54 ft vessel traveling at 21 kts self-reported strike. Calf stranded on 13Feb2021. Deep lacerations across back and head with associated fractured ribs and skull.

Date[b]	Fate	ID	Location[b]	Assigned Cause	Value against PBR[c]	Country[d]	Gear Type[e]	Description
2/12/2021	Prorated Injury	3230	St. Augustine, FL	VS	.52	US	-	54 ft vessel traveling at 21 kts self-reported strike. Lactating female with lacerations of unknown depth resighted on 16Feb2021. Dependent calf died from injuries received (see 12Feb2021 mortality event). No additional resights.
3/10/2021	Serious Injury	3560	off Sandwich, MA	EN	1	CN	GU	Free-swimming with constricting rostrum wrap and trailing gear 300 ft. Partial disentanglements on 3 separate occasions removed sections of trailing gear. Successful calving event in Dec2021. Stable health until 23Jul2022 when appeared thinner, increased lesions & cyamids, and discolored rostrum. Dependent calf (see 02Dec2021 event) last sighted on 26Apr2022, not present at 23Jul2022 or 22Sep2022 sightings. (Carrying new entanglement and in significant health decline at 22Sep2022 sighting.) No additional resights. Gear consistent with rope from Canadian trap gear.
7/13/2021	Serious Injury	4615	Gulf of St Lawrence, QC	EN	1	CN	NR	Recent (within hours) entanglement - Line through mouth and over rostrum leading down towards right flipper and back towards flukes. Constricting line over peduncle and down to weighted gear. Resighted on 14Jul2021 with rostrum wrap leading down to weighted gear, no gear on fluke or peduncle area. No additional resights. Entangled in rope of Canadian origin.
12/2/2021	Serious Injury	2022 calf of 3560	off Cumberland Island, GA	EN	1	CN	-	Dependent calf of seriously injured lactating female (see 10Mar2021 event). No resights post 26Apr2022.
5/19/2022	Prorated Injury	3823	Gulf of St. Lawrence, QC	EN	0.75	XC	NR	Free-swimming with line through mouth, possible bridling under chin, and trailing with one line ending hundreds of feet and the other descends to depth. Full configuration unknown. No resights.
6/30/2022	Serious Injury	1403	Gulf of St Lawrence, QC	EN	1	XC	NR	Free-swimming with tight body wrap. Attachment point(s) unknown. No resights post 07Jul2022.
8/20/2022	Mortality	5120	Gulf of St Lawrence, QC	EN	1	US	PT	Free-swimming with at least 4 constricting peduncle wraps and 1 fluke blade wrap. Resights through 12Jun2023 show deterioration in health - skin sloughing and increased cyamid load. Carcass found on 28Jan2024. Embedded in peduncle with associated hemorrhaging, thin body condition consistent with chronic entanglement. Entangled in a buoy line with markings consistent with Maine state waters trap gear.

Date[b]	Fate	ID	Location[b]	Assigned Cause	Value against PBR[c]	Country[d]	Gear Type[e]	Description
9/21/2022	Serious Injury	3560	off Nantucket, MA	EN	1[f]	XU	NR	Free-swimming with line through mouth and trailing to weighted gear. Possible flipper involvement. Heavy cyamid load. Still carrying constricting rostrum wrap from previous entanglement (see 10Mar2021 event). No resights

Assigned Cause	Observed five-year mean (US/CN/XU/XC)
Vessel strike	1.5 (0.7/0.8/0/0)
Entanglement	3.95 (0.2/1.55/1.7/0. 50)

a. For more details on events, see Henry et al. 2024. For full gear analysis, see https://www.fisheries.noaa.gov/new-england-mid-atlantic/marine-mammal-protection/atlantic-large-whale-take-reduction-plan.
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.
f. Individual with 2 separate serious injury entanglement events. Only one is counted against PBR.

STATUS OF STOCK

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 listed as an endangered species under the ESA. The size of this stock is extremely low relative to OSP and, until recently, had been declining (Figure 2a). 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; IUCN 2020). The observed (and clearly biased low, Pace et al. 2021) human-caused mortality and serious injury was 7.1 right whales per year from 2018 through 2022. Using the refined methods in Linden (2024b), the estimated annual rate of total mortality of adults and juveniles for the period 2018–2022 was 14.8, which is 2.7 times larger than the 5.45 total derived from reported mortality and serious injury for the same period. Given that PBR has been calculated as 0.73, human-caused mortality or serious injury for this stock must be considered significant.

OTHER FACTORS THAT MAY BE AFFECTING THE STOCK

Habitat Issues

Beyond human-caused mortality and serious injury, there are other factors that may be causing a decline or impeding right whale recovery, or may become factors in the future. These include potential effects of climate change and impacts of emerging industries such as offshore wind energy and aquaculture development.

Baumgartner et al. (2017) discussed 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. Ocean warming in the Gulf of Maine has 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; Meyer-Gutbrod et al. 2021) and an increase in sightings in Cape Cod Bay (Mayo et al. 2018; Ganley et al. 2019).

Climate change is also affecting the seasonal timing of the whales' presence in traditional habitats, leading to a mismatch with static management measures designed to reduce anthropogenic threats (Ganley et al. 2022; Pendleton et al. 2022). The Gulf of St. Lawrence has become an important habitat for a large portion of the population since at least 2015 (Simard et al. 2019; Crowe et al. 2021; Durette-Morin et al. 2022), which resulted in a substantial increase in anthropogenic mortality before management measures could be implemented (Davies and Brillant 2019). An Unusual Mortality Event was declared for the species as a result (https://www.fisheries.noaa.gov/national/marine-life-distress/2017-2024-north-atlantic-right-whale-unusual-mortality-event). Gavrilchuk et al. (2021) suggested 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 search of dense *Calanus* patches.

Food limitation may contribute to a decline in the population's health and reproduction. Meyer-Gutbrod et al. (2022) found that the right whales' increased use of the Gulf of St. Lawrence over the last 10 years was driven by a decline in prey in the Gulf of Maine, and not an increase in prey in Canada. Knowlton et al. (2022) found that the apparent health of all whales in the population had declined significantly since the 1980s, including those not documented as injured.

Declining body sizes are a potential contributor to low birth rates over the past decade. Stewart et al. (2022) found that larger whales had shorter inter-birth intervals and produced more calves per potential reproductive year. A whale born in 2019 is now expected to reach a body length 1 m shorter than a whale born in 1981. Smaller whales may be the result of poor nutrition or sublethal injury, either to the whale or to their mother (Stewart et al. 2021). Reed et al. (2022) show that it is both the failure of the pre-breeding females to transition to reproducing females, as smaller whales have less capacity to gain sufficient condition to calve than larger females (Christiansen et al. 2020), as well as the mortality of reproducing females, that has contributed to the recent right whale population decline.

Offshore wind energy development along the east coast of the U.S. will introduce additional stressors to North Atlantic right whales and their habitat, such as noise and/or pressure, entanglement hazards, vessel traffic, and changes in oceanographic conditions. Potential impacts to North Atlantic right whales, depending on the stressors, include: hearing impairment; behavioral disturbance; avoidance of wind areas; injury and mortality (i.e., from entanglement or vessel strike); and changes in quality and availability of prey that may lead to reduced fitness (decreased survival and reproduction, Bailey et al. 2014; Barkaszi et al. 2021; Carpenter et al. 2016; Dorrell et al. 2022; Leiter et al. 2017; Maxwell et al. 2022; Quintana-Rizzo et al. 2021). While only a few projects in U.S. water are currently fully approved and under development, should the proposed development go forward as planned, the extensive overlap with their range would mean that in the future, any individual right whale may be exposed to multiple projects.

Expansions to the aquaculture industry, both inshore and offshore, may also affect North Atlantic right whales. Lines in the water for various types of aquaculture increase the potential for entanglement, both directly through whale interactions with aquaculture gear or secondarily through the entanglement of trailing gear on a whale with fixed aquaculture gear (Price et al. 2017). Increased vessel traffic in and around aquaculture farms will increase ambient noise levels and the risk of vessel strikes (Price et al. 2017). There may also be oceanographic changes to areas used for aquaculture that could affect the physical environment or create changes to prey availability.

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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. Off the eastern United States, Nova Scotia, and the southeastern coast of Newfoundland, fin whales 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).

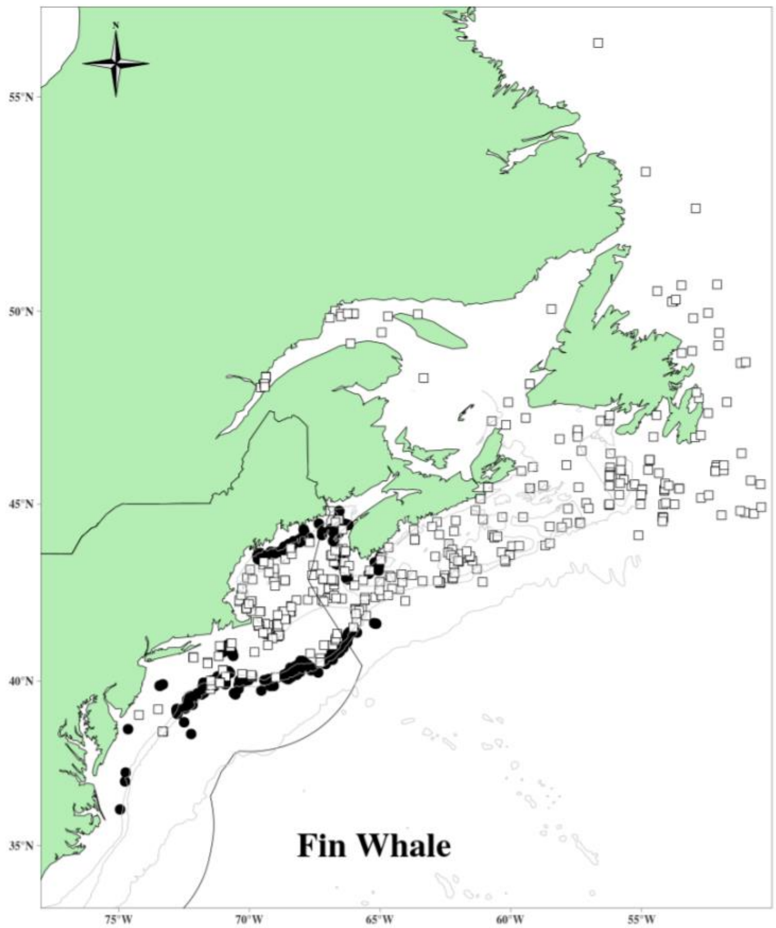

Figure 1. Distribution of fin whale sightings from NEFSC and SEFSC shipboard and aerial surveys during the summers of 1995, 1998, 1999, 2002, 2004, 2006, 2007, 2008, 2010, 2011, 2016, and 2021 and DFO's 2007 TNASS and 2016 NAISS surveys. Isobaths are the 100-m, 1,000-m and 4000-m depth contours. Circle 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). 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 between Cape Hatteras and Nova Scotia during 1978–1982 (CETAP 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 grounds for fin whales. There is evidence of site fidelity by females, and perhaps some segregation by sexual, maturational, or reproductive class in the feeding area (Agler et al. 1993; Schleimer et al. 2019). 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. Based on an analysis of neonate stranding data, Hain et al. (1992) 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), the 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. Estimates generated from the 2021 surveys are more recent and focus on U.S. waters, although more of the stock range was covered in the 2016 survey.

Earlier Abundance Estimates

Please see Appendix IV for earlier abundance estimates.

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.

An abundance estimate of 2,240 (CV=0.39) fin whales was generated from vessel surveys conducted in U.S. waters of the western North Atlantic during the summer of 2021 (Table 1; Garrison and Dias 2023; Palka 2023). One survey was conducted from 16 June to 23 August in waters north of 36ºN latitude and consisted of 5,871 km of on-effort trackline along the shelf break and offshore to the outer edge of the U.S. EEZ (NEFSC and SEFSC 2022). The second vessel survey covered waters from central Florida (25ºN latitude) to approximately 38ºN latitude between the 200-m isobaths and the outer edge of the U.S. EEZ during 12 June–31 August. A total of 5,659 km of trackline was covered on effort (NEFSC and SEFSC 2022). 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. No fin whales were seen in the SE portion of the survey.

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**
Jun–Aug 2021	New Jersey to lower Bay of Fundy	2,240	0.39
Jun–Aug 2021	Central Florida to New Jersey	0	-
Jun–Aug 2021	Central Florida to lower Bay of Fundy (COMBINED)	2,240	0.39

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 the fullstock. The statistical power to detect a trend in abundance for this stock is poor due to the relatively imprecise abundance estimates and variable survey design. For example, the power to detect a precipitous decline in abundance (i.e., 50% decrease in 15 years) with estimates of low precision (e.g., CV>0.30) remains below 80% (alpha=0.30) unless surveys are conducted on an annual basis (Taylor et al. 2007). There is current work to standardize the stratum-specific previous abundance estimates to consistently represent the same regions and include appropriate corrections for perception and availability bias. These standardized abundance estimates will be used in state-space trend models that incorporate environmental factors that could potentially influence the process and observational errors for each stratum.

CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

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

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

POTENTIAL BIOLOGICAL REMOVAL

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

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

N_{est}	CV	N_{min}	F_r	R_{max}	PBR
6,802	0.24	5,573	0.1	4	11

ANNUAL HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

For the period 2017 through 2021, the annual detected (i.e., observed) human-caused mortality and serious injury to fin whales averaged 2.05 individuals per year (Table 3). This is derived from two components: 1) incidental fishery entanglement records at 1.45 per year and 2) vessel strike records averaging 0.60 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. 2023). 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 4.

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.
2017−2021	Fishery entanglements	1.45
2017−2021	Vessel strikes	0.60
TOTAL		2.05

Fishery-Related Serious Injury and Mortality

United States

U.S. fishery interaction records for large whales are sourced from dedicated fishery observer data, and opportunistic reports compiled in the Greater Atlantic Regional Fisheries Office (GARFO)/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 with substantial evidence of fishery interactions causing serious injury or mortality are presented in Table 4 (Henry et al. 2023). These records likely underestimate entanglements for the stock.

Canada

Confirmed mortalities and serious injuries from the current reporting period that were likely a result of an interaction with Canadian fisheries are included in Table 4.

Table 4. Confirmed human-caused mortality and serious injury records of fin whales (*Balaenoptera physalus*) attributed to entanglement (EN) or vessel strike (VS): 2017–2021[a].

Date[b]	Fate	ID	Location[b]	Assigned Cause	Value against PBR[c]	Country[d]	Gear Type[e]	Description
30May17	Mortality	-	Port Newark, NJ	VS	1	US	-	Fresh carcass on bow of 656 ft vessel. Speed at strike unknown.

Date[b]	Fate	ID	Location[b]	Assigned Cause	Value against PBR[c]	Country[d]	Gear Type[e]	Description
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	-	off 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	-	Off 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.
7Jul20	Prorated Injury	-	off of MacKenzie Point, Pleasant Bay, NS	EN	0.75	XC	NR	Free-swimming. Line crossing from left mouth/head over back and down right side. Attachment point(s) and full configuration unknown. No resights.
Assigned Cause					**5-Year mean (US/CN/XU/XC)**			
Vessel Strike					0.60 (0.60/0/0/0)			
Entanglement					1.45 (0/0.8/0.15/0.5)			

a. For more details on events please see Henry et al. 2023.
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, Nav=Navigational buoy.

Other Mortality

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

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, mortality and serious injury in commercial fisheries cannot be considered insignificant and approaching a zero mortality and serious injury rate. The status of this stock relative to its Optimum Sustainable Population (OSP) is unknown. There are insufficient data to determine the population trend for fin whales though there is evidence for a decline of the subpopulation in the northern Gulf of St. Lawrence (Schleimer et al. 2019). 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.

OTHER FACTORS THAT MAY BE AFFECTING THE STOCK

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.

Phenological changes were documented for fin whales in the Gulf of St. Lawrence by Ramp et al. (2015). Their study documented earlier shifts in the timing of arrival and departure of fin whales in the Gulf of St. Lawrence from 1984–2010. They estimated an arrival date shift of >1 day per year earlier, and a departure date shift of 0.4 day per year earlier in the Jacques Cartier Passage of the Gulf of St. Lawrence. Further, their study found significant relationships between fin whale arrival/departure dates, the first ice-free week in the Gulf of St. Lawrence, and January sea surface temperatures in Cabot Strait. Another study (Pendleton et al. 2022) estimated the date of peak habitat use for fin whales from 1998–2018 in Cape Cod Bay, located in the southwestern Gulf of Maine. This study found a significant positive relationship between the date of peak occupancy of fin whales in Cape Cod Bay and the thermal spring transition date (Friedland et al. 2015) in the eastern Gulf of Maine. These studies suggest that fin whales are adapting to long-term changes in temperature, although the mechanisms behind these relationships and effects on the population are not known at this time.

Chavez-Rosales et al. (2022) documented an overall 178 km northeastward spatial distribution shift of the seasonal core habitat of Northwest Atlantic cetaceans that was related to changing habitat/climatic factors. Results varied by season and species. This study used sightings data collected during seasonal aerial and shipboard line transect abundance surveys during 2010 to 2017. During this time frame, the weighted centroid of the fin whale core habitat moved farthest during fall (223 km towards the northeast) and least during winter (33 km). There is uncertainty in how, if at all, the changes in distribution and population size of cetacean species may interact with changes in distribution of prey species and how the ecological shifts will affect human impacts to the species.

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

STOCK DEFINITION AND GEOGRAPHIC RANGE

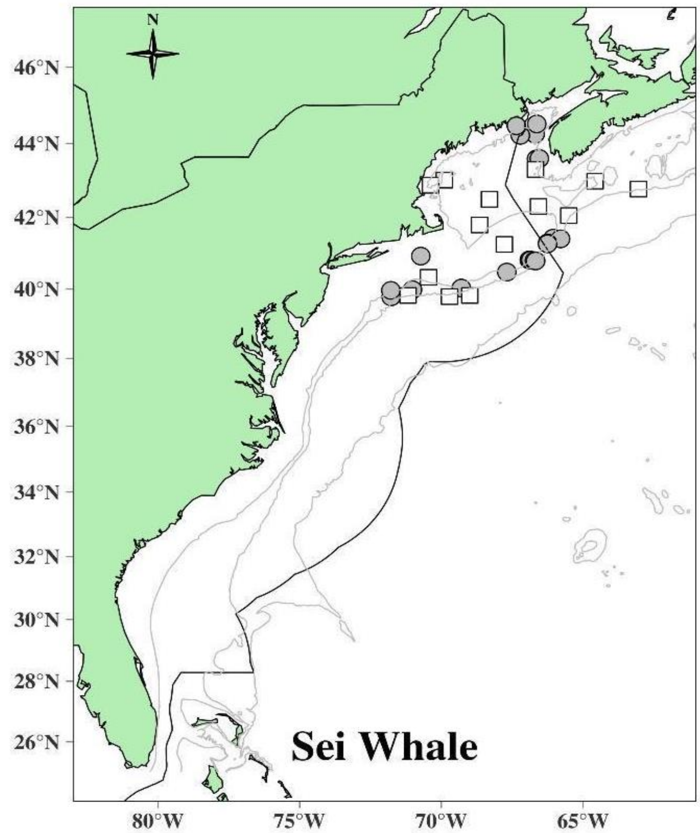

Figure 1. Distribution of sei whale sightings from NEFSC and SEFSC shipboard and aerial surveys during the summers of 1995, 1998, 1999, 2002, 2004, 2006, 2007, 2008, 2010, 2011, 2016 and 2021 and DFO's 2007 TNASS and 2016 NAISS surveys. Isobaths are the 100-m, 200-m, 1000-m and 4000-m depth contours.

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 based on both mitochondrial DNA and microsatellite analyses, did not reveal stock structure in the North Atlantic 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 as a stock for the purposes of management under the MMPA. The IWC considered the boundaries of this stock to be 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.

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 of 825 sei whales taken between 1965 and 1972 at the Blandford, Nova Scotia whaling station, 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. Indeed, the greatest abundance of sei whales in U.S. waters occurs during spring, 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 from 2004–2014, detected sei whale 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. ,North Atlantic sei whales are largely planktivorous, feeding primarily on euphausiids and copepods (Flinn et al. 2002), although they are known to eat fish in other oceans (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 sei whale stomachs from sei whales showed a clear preference for copepods between June and October, while euphausiids were taken only in May and November (Mitchell 1975). During some years sei whales were reported in more inshore locations, such as the Great South Channel (in 1987 and 1989) and Stellwagen Bank (in 1986) (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 the best available for the Nova Scotia stock of sei whales. This estimate is considered the best 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 (spring), and used aerial survey data corrected for availability bias. However, this estimate must be considered uncertain for the following reasons: 1) the entire known range of this stock was not surveyed 2)uncertainties exist regarding population structure and whale movements between surveyed and unsurveyed areas, 3) data collection includes ambiguous identification between fin and sei whales and 4) analytical challenges exist, such as how best to account for the ambiguous sightings and low encounter rates, and how to define the most appropriate species-specific availability bias correction factor.

Recent Surveys and Abundance Estimates

The springtime (March–May) average abundance estimate generated from spatially- and temporally-explicit density models was 6,292 (CV=1.02) sei whales. This was 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 Scotian shelf outside of U.S. waters. Surveys included over 25,000 km of shipboard and over 99,000 km of aerial visual line-transect data collected in all seasons in Atlantic waters from Florida to Nova Scotia. These data were divided into 10x10 km spatial grid cells and 8-day temporal 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 also accounted for platform- and species-specific availability bias, with 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) spanning 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 visual 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 34 (CV=0.99) sei whales was generated from vessel surveys conducted in U.S. waters of the western North Atlantic during the summer of 2021 (Table 1; Garrison and Dias 2023; Palka 2023). One survey was conducted from 16 June to 23 August in waters north of 36ºN latitude and consisted of 5,871 km of on-effort trackline along the shelf break and offshore to the outer edge of the U.S. EEZ (NEFSC and SEFSC 2022). The second vessel survey covered waters from central Florida (25ºN latitude) to approximately 38ºN latitude between the 200-m isobaths and the outer edge of the U.S. EEZ during 12 June–31 August. A total of 5,659 km of trackline was covered on effort (NEFSC and SEFSC 2022). 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. Additionally aerial surveys were concurrently conducted from Nova Scotia to Florida from the coast to the shelf break and did not record any sei whales.

Comprehensive summer 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, 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).

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). The estimate considered best is in bold font.

Month/Year	Area	N_{est}	CV
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
Jun–Aug 2021	New Jersey to southern Nova Scotia	34	0.99
Jun–Aug 2021	Central Florida to New Jersey	0	0
Jun–Aug 2021	Central Florida to southern Nova Scotia (COMBINED)	34	0.99

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

There are insufficient data to determine population trends for this species. 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.

Although not a formal trend analysis, during all seasons, the seasonal average habitat-based abundance estimates generated by Palka et al. (2021) resulted in lower recent abundance estimates (2014–2017) as compared to those from the past (2010–2013), where the center of the distribution moved southwesterly (Chavez-Rosales et al. 2022).

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

For the period 2017 through 2021, the annual detected (i.e., observed) human-caused mortality and serious injury to Nova Scotia sei whales averaged 0.60 individuals per year (Table 3). This is derived from two components: 1) incidental fishery entanglement records at 0.40 per year, and 2) other human caused mortality averaging 0.20 per year.

Injury determinations are made based upon the best available data; these determinations may change with the availability of new information (Henry et al. 2023). 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 4.

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.
2017– 2017	Fishery entanglement	0.40
2017– 2021	Vessel strikes	0
2017– 2021	Other human-caused mortality	0.20
TOTAL		0.60

Fishery-Related Mortality and Serious Injury

No confirmed fishery-related mortalities or serious injuries of sei whales have been reported in the NMFS Sea Sampling bycatch database. Records of stranded, floating, or injured sei whales for the period 2017 through 2021 on file at NMFS indicate two sei whales with substantial evidence of fishery interaction that caused mortality or serious injury (Table 4), suggesting an annual mortality and serious injury rate of 0.4 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): 2017–2021 [a].***

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

Date[b]	Injury Determination	ID	Location[b]	Assigned Cause	Value against PBR[c]	Country[d]	Gear Type[e]	Description
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 (0/0/0/0)
Entanglement	0.40 (0/0/0.40/0)

a. For more details on events please see Henry et al. 2023.
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

No sei whale vessel collisions were recorded during 2017-2021. One sei whale in 2019 was reported with cause of death as starvation due to plastic ingestion (see Table 3 - other mortality).

STATUS OF STOCK

This is a strategic stock because the sei whale is listed as an endangered species under the ESA. The total 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) is unknown. There are insufficient data to determine population trends for sei whales.

OTHER FACTORS THAT MAY BE AFFECTING THE STOCK

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.

Chavez-Rosales et al. (2022) documented an overall 178 km northeastward spatial distribution shift of the seasonal core habitat of Northwest Atlantic cetaceans that was related to changing habitat/climatic factors. Results varied by season and species. This study used sighting data collected during seasonal aerial and shipboard line transect abundance surveys during 2010 to 2017. During this time frame, the weighted centroid of the sei whale core habitat moved farthest during winter (179 km towards the southwest) and least during spring (70 km). There is uncertainty in how, if at all, the changes in distribution and population size of cetacean species may interact with changes in distribution of prey species and how the ecological shifts will affect human impacts to the species.

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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., This 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).

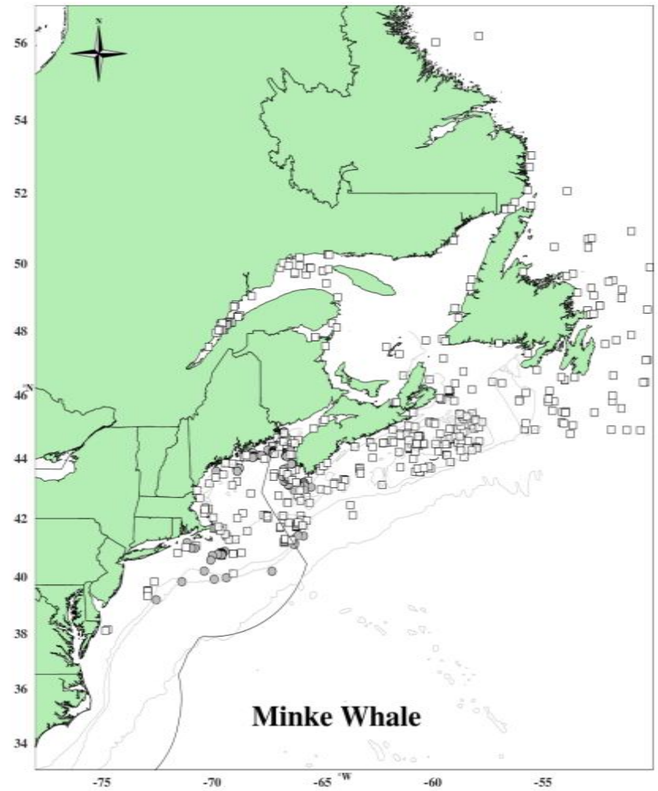

Figure 1. Distribution of minke whale sightings from NEFSC and SEFSC shipboard and aerial surveys during the summers of 1995, 1998, 1999, 2002, 2004, 2006, 2007, 2008, 2010, 2011, 2016 and 2021 and DFO's 2007 TNASS and 2016 NAISS surveys. Isobaths are the 100-m, 200-mm 1000-m and 4000-m depth contours. 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. However, using individual genotypes and likelihood assignment methods, they identified 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 derived from surveys covering more of this species habitat: from Newfoundland to Florida, in contrast to the most recent estimate from 2021 that only covered from Nova Scotia to Florida. 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 earlier abundance estimates.

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 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 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 visual 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 resulting in an abundance estimate and estimate of perception bias. 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.

A more recent abundance estimate of 5,630 (CV=0.58) minke whales was generated from vessel surveys conducted in U.S. waters of the western North Atlantic during the summer of 2021 (Table 1; Garrison and Dias 2023; Palka 2023). One survey was conducted from 16 June to 23 August in waters north of 36ºN latitude and consisted of 5,871 km of on-effort trackline along the shelf break and offshore to the outer edge of the U.S. EEZ (NEFSC and SEFSC 2022). The second vessel survey covered waters from central Florida (25ºN latitude) to approximately 38ºN latitude between the 200-m isobaths and the outer edge of the U.S. EEZ during 12 June–31 August. A total of 5,659 km of trackline was covered on effort (NEFSC and SEFSC 2022). 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.

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 survey, and resulting 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**
Jun–Aug 2021	New Jersey to lower Bay of Fundy	5,630	0.58

Month/Year	Area	N_{est}	CV
Jun–Aug 2021	Central Florida to New Jersey	0	0
Jun–Aug 2021	Central Florida to lower Bay of Fundy (COMBINED)	5,630	0.58

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. However, they can be estimated according to life history parameters, such as the age of female sexual maturity (6-8 years) and pregnancy rates (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. 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

For the period 2017 through 2021, the annual detected (i.e., observed) human-caused mortality and serious injury to common minke whales averaged 9.40 individuals per year (Table 3. This is derived from two components: 1) incidental fishery entanglement records at 8.60 per year, and 2) vessel strikes averaging 0.80 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. 2023). 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 4.

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.
2017−2021	Fishery entanglement non-observed	8.60
2017−2021	U.S. fisheries using observer data	0
2017−2021	Vessel strikes	0.8
TOTAL		9.40

Fishery-Related Serious Injury and Mortality

United States

U.S. fishery interaction records for large whales are sourced from dedicated fishery observer data and opportunistic reports compiled in the Greater Atlantic Regional Fisheries Office (GARFO)/NMFS entanglement/stranding database. No confirmed fishery-related mortalities or serious injuries of minke whales have been reported in the NMFS Sea Sampling bycatch database (fishery observers) during this reporting period. Records of stranded, floating, or injured minke whales with substantial evidence of fishery interactions causing serious injury or mortality for the reporting period, are presented in Table 4 (Henry et al. 2023). These records likely underestimate entanglements for the stock.

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 4. Most cases in which gear was recovered and identified involved gillnet or pot/trap gear.

Canada

Read (1994) reported common minke whale interactions with gillnets in Newfoundland and Labrador, with cod traps in Newfoundland, and with herring weirs in the Bay of Fundy. Hooker et al. (1997) summarized bycatch data from a Canadian fisheries observer program whereby observers were deployed 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. From 1991 -1996, no observed common minke whale bycatch interactions were reported. More current observer data are not available. Wimmer and Maclean (2021) reported that 34% of the live entanglements documented in Eastern Canada between 2004 and 2019 involved minke whales.

Mortalities and serious injuries that were likely a result of an interaction with Canadian fisheries are detailed in Table 4.

Table 4. Confirmed human-caused mortality and serious injury records of common minke whales (*Balaenoptera acutorostrata acutorostrata*): 2017–2021.

Date[b]	Injury determination	ID	Location[b]	Assigned Cause[c]	Value against PBR[d]	Country[e]	Gear Type[f]	Description
24Apr17	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 cause of death (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	No gear present. Evidence of multiple constricting wraps on lower jaw and ventral pleats with associated hemorrhaging.

Date[b]	Injury determination	ID	Location[b]	Assigned Cause[c]	Value against PBR[d]	Country[e]	Gear Type[f]	Description
09Aug17	Mortality	-	off Plymouth, MA	EN	1	US	NP	No gear present. 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.
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.
04Sep17	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.
06Sep17	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.
17Sep17	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 it is unlikely to have drifted into gear post-mortem.
26Sep17	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.
27Sep17	Mortality	-	off 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.

Date[b]	Injury determination	ID	Location[b]	Assigned Cause[c]	Value against PBR[d]	Country[e]	Gear Type[f]	Description
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	-	off 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	-	off 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	-	off 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	-	off 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.

Date[b]	Injury determination	ID	Location[b]	Assigned Cause[c]	Value against PBR[d]	Country[e]	Gear Type[f]	Description
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	-	off 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	-	off 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	-	off Matinicus 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.
14Jan20	Mortality	-	off Port Mouton Island, NS	EN	1	CN	PT	Fresh carcass anchored in gear.
03Feb20	Mortality	-	off Chesconessex, VA	EN	1	US	GN	Gear around pectorals and peduncle and through mouth. COD = Peracute underwater entrapment.
17Jun20	Mortality	-	off Newport, RI	EN	1	US	NE	Fresh carcass anchored in gear.
26Aug20	Mortality	-	off Wood Island, ME	EN	1	US	PT	Limited necropsy - fresh, scavenged carcass with flipper wraps and evidence of constricting peduncle wrap. Stomach full of partially digested fish indicating acute mortality. COD from entanglement most parsimonious.
04Sep20	Serious Injury	-	off North Truro, MA	EN	1	XU	NE	Free-swimming with netting in mouth, embedded in rostrum, and trailing alongside each flank. Poor condition - emaciated, pocked skin, and deep, open wounds. Disentanglement response unsuccessful. No resights.
06Oct20	Mortality	-	Dennis Harbor, MA	EN	1	US	GU	Closed bridle of line through mouth with associated hemorrhaging at corners of the mouth.
06Jun21	Prorated Injury	-	off North Truro, MA	EN	0.75	XU	NR	Free-swimming with a large jumble of gear on its back, forward of dorsal. Attachment point(s) and full configuration unknown.
04Jul21	Mortality	-	off Point Judith, RI	EN	1	XU	NE	Constricting rostrum wraps with associated hemorrhaging. Propeller lacerations on ventral jaw acquired post mortem. COD consistent with entanglement.
11Jul21	Prorated Injury	-	Mary’s Harbour, NL	EN	0.75	CN	GN	Fisher self-reported animal entangled in gear in unknown configuration. Partially disentangled - released carrying part of net and poly float from unknown attachment point(s) and in unknown configuration.

Date[b]	Injury determination	ID	Location[b]	Assigned Cause[c]	Value against PBR[d]	Country[e]	Gear Type[f]	Description
17Jul21	Mortality	-	off Manomet Point, Plymouth, MA	VS	1	US	-	Fractured rostrum and mandibles with associated tissue damage consistent with blunt force trauma. Robust body condition and partially digested fish in forestomach indicates acute mortality.
01Aug21	Mortality	-	off Sequin Island, ME	EN	1	US	PT	Anchored in gear with constricting wraps on left pectoral and at fluke insertion. Limited necropsy but COD from entanglement most parsimonious. Robust condition and unlikely to have drifted into gear post mortem and become so extensively wrapped.

Assigned Cause	5-Year mean (US/CN/XU/XC)
Vessel strike	0.8 (0.6/ 0.2/0/0)
Entanglement	8.60 (3.15/2.40/2.35/0.70)

a. For more details on events please see Henry et al. 2023.
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.

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 4. 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 2021, 123 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 2017–2021 as part of this UME are included in Table 4.

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 the Canadian Maritime Provinces were recorded by the Marine Animal Response Society (MARS) (Wimmer and Maclean 2021).

The Whale Release and Strandings program reports common minke whale stranding mortalities in Newfoundland and Labrador (Ledwell and Huntington 2018, 2019, 2020, 2021a, 2021b). Those that have been determined to be human-caused serious injury or mortality are included in Table 4.

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 because the estimated average annual human-related mortality does not exceed PBR. The total fishery-related mortality and serious injury for this stock (8.6) 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 is unknown.

It is expected that the uncertainties described above will have little effect on the designation of the status of the entire stock. The estimate of human-caused mortality and serious injury in this assessment (9.4 animals annually, Table 3) is negatively biased due to reliance on strandings and entanglement data as the primary data sources.Human-caused mortality is 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.

OTHER FACTORS THAT MAY BE AFFECTING THE STOCK

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).

Chavez-Rosales et al. (2022) documented an overall 178 km northeastward spatial distribution shift of the seasonal core habitat of Northwest Atlantic cetaceans that was related to changing habitat/climatic factors. Results varied by season and species. This study used sightings data collected during seasonal aerial and shipboard line transect abundance surveys during 2010 to 2017. During this time frame, the weighted centroid of the minke whale core habitat moved farthest during winter (133 km towards the northeast) and least during fall (10 km towards the west). There is uncertainty in how, if at all, the changes in distribution and population size of cetacean species may interact with changes in distribution of prey species and how the ecological shifts will affect human impacts to the species.

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

STOCK DEFINITION AND GEOGRAPHIC RANGE

The distribution of the sperm whale in the U.S. Exclusive Economic Zone (EEZ) occurs on the continental shelf edge, over the continental slope, and into mid-ocean regions (Figure 1). Waring et al. (1993, 2001) suggested that this offshore distribution is more commonly associated with the Gulf Stream edge and other features. However, the sperm whales that occur in the eastern U.S. Atlantic EEZ likely represent only a fraction of the total stock. The nature of linkages of the U.S. habitat with those to the south, north, and offshore is unknown. Historical whaling records compiled by Schmidly (1981) suggest an offshore distribution off the southeast U.S., over the Blake Plateau, and into deep ocean waters. In the southeast Caribbean, both large and small adults, as well as calves and juveniles of different sizes are reported (Watkins et al. 1985). Whether the northwestern Atlantic population is discrete from the northeastern Atlantic is currently unresolved. The International Whaling Commission recognizes one stock for the North Atlantic. Based on reviews of many types of stock studies (i.e., tagging, genetics, catch data, mark-recapture, biochemical markers, etc.), Reeves and Whitehead (1997), and Dufault et al. (1999) suggested that sperm whale populations have no clear geographic structure. Ocean-wide genetic studies (Lyrholm and Gyllensten 1998; Lyrholm et al. 1999) indicated low genetic diversity but strong differentiation between potential social (matrilineally related) groups. Further, Englehaupt et al. (2009) found no differentiation between mtDNA samples from the western North Atlantic and from the North Sea, but significant differentiation between samples from the Gulf of Mexico and from the Atlantic Ocean just outside the Gulf of Mexico. These ocean-wide findings, combined with observations from other studies, indicate stable social groups, site fidelity, and latitudinal range limitations in groups of females and juveniles (Whitehead 2002). In contrast, males migrate to polar regions to feed and move among populations to breed (Whitehead 2002, Englehaupt 2009). There exists one tag return of a male tagged off Browns Bank (Nova Scotia) in 1966 and returned from Spain in 1973 (Mitchell 1975). Another male taken off northern Denmark in August 1981 had been wounded the previous summer by whalers off the Azores (Reeves and Whitehead 1997). Steiner et al. (2012) reported on resightings of photographed individual male sperm whales between the Azores and Norway. In U.S. Atlantic EEZ waters, there appears to be a distinct seasonal cycle (CETAP 1982; Scott and Sadove 1997). In winter, sperm whales are concentrated east and northeast of Cape Hatteras. In spring, the center of distribution shifts northward to east of Delaware and Virginia and is widespread throughout the central portion of the mid-Atlantic Bight and the southern portion of Georges Bank. This is supported by acoustic studies in which detection of sperm whale vocalizations had a winter peak off Cape Hatteras, with the peak shifting farther north in the spring (Stanistreet et al. 2018). In summer, the distribution is similar but now also includes the area east and north of Georges Bank and into the Northeast Channel region, as well as the continental shelf (inshore of the 100-m isobath) south of New England. In the fall, sperm whales occur south of New England in relatively high numbers, and are alsopresent along the

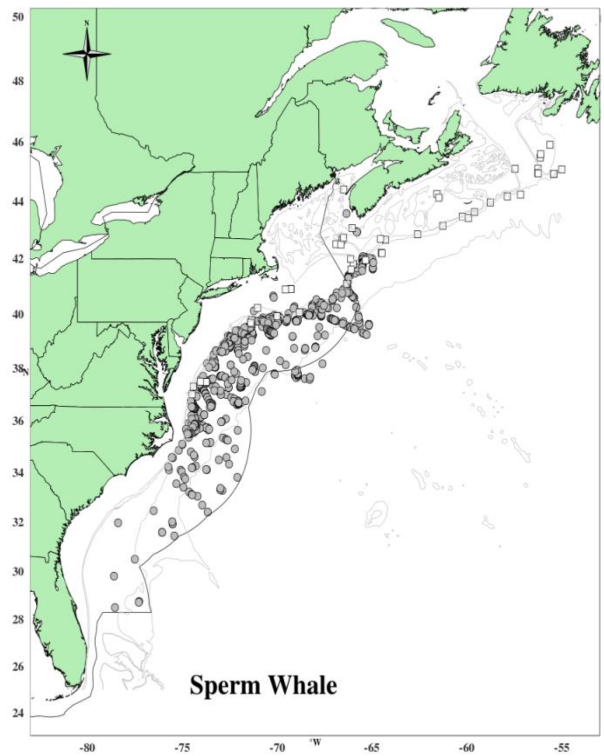

Figure 1. Distribution of sperm whale sightings from NEFSC and SEFSC shipboard and aerial surveys during the summer in 1998, 1999, 2002, 2004, 2006, 2011, 2016 and 2021 and Department of Fisheries and Oceans Canada 2007 TNASS and 2016 NAISS surveys. Isobaths are the 100m, 1,000m, and 4,000m depth contours. Circle symbols represent shipboard sightings and squares are aerial sightings.

continental shelf edge in the mid-Atlantic Bight. Similar inshore (<200 m) observations have been made on the southwestern (R.D. Kenney, pers. comm.) and eastern Scotian Shelf, particularly in the region of “the Gully” (Whitehead et al. 1991).

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

POPULATION SIZE

Several estimates from selected regions of sperm whale habitat exist for select time periods; however, at present there is no reliable estimate of total sperm whale abundance for the entire North Atlantic. Sightings have been almost exclusively in the continental shelf edge and continental slope areas (Figure 1); however, there has been little or no survey effort beyond the slope. The best recent abundance estimate for sperm whales is the sum of the 2021 surveys described below—5,895 (CV=0.29).

Recent Surveys and Abundance Estimates

Abundance estimates of 3,321 (CV=0.35), and 1,028 (CV=0.35) sperm whales were generated from vessel surveys conducted in U.S. waters of the western North Atlantic during the summer of 2016 (Table 1; Garrison 2020; Palka 2020). One survey was conducted from 27 June to 25 August in waters north of 38ºN latitude (Central Virginia) and consisted of 5,354 km of on-effort trackline along the shelf break and offshore to the outer limit of the U.S. EEZ (NEFSC and SEFSC 2018). The second vessel survey covered waters from Central Florida to approximately 38ºN latitude between the 100-m isobath and the outer limit of the U.S. EEZ during 30 June–19 August. A total of 4,399 km of trackline was covered on effort (NEFSC and SEFSC 2018). Both surveys utilized two visual teams and an independent observer approach to estimate detection probability on the trackline (Laake and Borchers 2004). Mark-recapture distance sampling was used to estimate abundance. Estimates from the two surveys were combined and CVs pooled to produce a species abundance estimate for the stock area.

More recent abundance estimates of 3,789 (CV=0.38) and 2,106 (CV=0.44) sperm whales were generated from vessel surveys conducted in U.S. waters of the western North Atlantic during the summer of 2021 (Table 1; Garrison and Aichinger-Dias 2023; Palka 2023). One survey was conducted from 16 June to 23 August in waters north of 36ºN latitude and consisted of 5,871 km of on-effort trackline along the shelf break and offshore to the outer edge of the U.S. EEZ (NEFSC and SEFSC 2022). The second vessel survey covered waters from central Florida (25ºN latitude) to approximately 38ºN latitude between the 200-m isobaths and the outer edge of the U.S. EEZ during 12 June–31 August. A total of 5,659 km of trackline was covered on effort (NEFSC and SEFSC 2022). Both surveys utilized two visual teams and an independent observer approach to estimate detection probability on the trackline (Laake and Borchers 2004). Mark-recapture distance sampling was used to estimate abundance. Estimates from the two surveys were combined and CVs pooled to produce a species abundance estimate for the stock area.

Table 1. Summary of abundance estimates for the western North Atlantic sperm whale (*Physeter macrocephalus*). Month, year, and area covered during each abundance survey, and resulting abundance estimate (N_{best}) and coefficient of variation (CV). The estimate considered best is in bold font.

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

Month/Year	Area	N_{best}	CV
Jun–Aug 2021	New Jersey to lower Bay of Fundy	3,789	0.38
Jun–Aug 2021	Central Florida to New Jersey	2,106	0.44
Jun–Aug 2021	**Central Florida to lower Bay of Fundy (COMBINED)**	**5,895**	**0.29**

Minimum Population Estimate

The minimum population estimate is the lower limit of the two-tailed 60% confidence interval of the log-normally distributed best abundance estimate. This is equivalent to the 20th percentile of the log-normal distribution as specified by Wade and Angliss (1997). The best estimate of abundance for sperm whales is 5,895 (CV=0.29). The minimum population estimate for the western North Atlantic sperm whale is 4,639.

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

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

POTENTIAL BIOLOGICAL REMOVAL

Potential Biological Removal (PBR) is the product of minimum population size, one-half the maximum productivity rate, and a recovery factor (MMPA Sec. 3. 16 U.S.C. 1362; Wade and Angliss 1997). The minimum population size is 4,639 (Table 2). The maximum productivity rate is 0.04, the default value for cetaceans. The recovery factor, which accounts for endangered, depleted, threatened stocks, or stocks of unknown status relative to optimum sustainable population (OSP) is assumed to be 0.10 because the sperm whale is listed as endangered under the Endangered Species Act (ESA). PBR for the western North Atlantic sperm whale is 9.28.

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

N_{est}	CV	N_{min}	F_r	R_{max}	PBR
5,895	0.29	4,639	0.1	0.04	9.28

ANNUAL HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

There are no documented reports of fishery-related mortality or serious injury to this stock during 2017–2021, though one stranding mortality in Florida in 2021 was attributed to ingestion of plastics including fishing net.

Table 3. The total annual observed average human-caused mortality and serious injury for the western North Atlantic sperm whale (Physeter macrocephalus).

Years	Source	Annual Avg.
2017−2021	Fishery entanglements	0
2017−2021	Vessel strikes	0
2017−2021	Other (plastic ingestion, see Table 4)	0.2
TOTAL		0.2

Fishery Information

Detailed fishery information is reported in Appendix III.

Other Mortality

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

STATUS OF STOCK

This is a strategic stock because the species is listed as endangered under the ESA. Total fishery-related mortality and serious injury for this stock is less than 10% of the calculated PBR, and therefore can be considered to be insignificant and approaching a zero mortality and serious injury rate. The status of this stock relative to OSPis unknown. There are insufficient data to determine population trends. The current stock abundance estimate was based upon a small portion of the known stock range. A Recovery Plan for sperm whales was finalized in 2010 (NMFS 2010).

OTHER FACTORS THAT MAY BE AFFECTING THE STOCK

Strandings

During 2017–2021, 10 sperm whale strandings were documented along the U.S. Atlantic coast within the EEZ (NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 15 October 2022; Table 4). Two of these strandings were classified as human interactions, both due to plastic ingestion (Table 3); however, in only one case was the plastic clearly the cause of death.

Table 4. Sperm whale (**Physeter macrocephalus*****) reported strandings along the U.S. and Canada Atlantic coast 2017–2021.***

Stranding State or Province	2017	2018	2019	2020	2021	Total
Newfoundland/Labrador[a]	01	1	2	1	3	7
Nova Scotia[b]	1	0	0	2	1	3
Massachusetts	0	1	0	0	0	2
New York	0	1	0	0	0	1
Maryland**c**	0	0	1	0	0	1
Virginia	0	0	0	0	0	0
North Carolina	1	0	0	0	0	1
South Carolina	0	1	0	0	0	1
Florida[d]	1	1	0	1	1	4

Stranding State or Province	2017	2018	2019	2020	2021	Total
TOTAL U.S.	**2**	**4**	**3**	**1**	**1**	**11**

a. Data provided by Whale Release and Strandings, Tangly Whales Inc. Newfoundland, Canada (Ledwell et al. 2018, Ledwell et al. 2021a, 2021b).
b. Data supplied by Nova Scotia Marine Animal Response Society (pers. comm.).
c. Maryland 2019 animal coded as HI due to plastic ingestion, although not clearly the cause of death.
d. Florida 2021 animal coded as HI due to ingestion of fishing net and other plastic as well as FI entanglement.

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

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 North Atlantic stock of sperm whales is lacking.

Anthropogenic sound in the world's oceans has been shown to affect marine mammals, with vessel traffic, seismic surveys, and active naval sonars being the main anthropogenic contributors to low- and mid-frequency noise in oceanic waters (e.g., Nowacek et al. 2015; Gomez et al. 2016; NMFS 2018). More recent studies have documented changes in dive patterns and acoustic behavior of sperm whales in response to anthropogenic noise (Farmer et al. 2018; Isojunno et al. 2020; Stanistreet et al. 2022). 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.

Chavez-Rosales et al. (2022) documented an overall 178 km northeastward spatial distribution shift of the seasonal core habitat of Northwest Atlantic cetaceans that was related to changing habitat/climatic factors. Results varied by season and species. This study used sightings data collected during seasonal aerial and shipboard line transect abundance surveys during 2010 to 2017. During this time, the weighted centroid of sperm whale core habitat moved farthest during fall (255 km towards the northeast) and least during winter (71 km). There is uncertainty in how, if at all, the changes in distribution and population size of cetacean species may interact with changes in distribution of prey species and how the ecological shifts will affect human impacts to the species.

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

STOCK DEFINITION AND GEOGRAPHIC RANGE

The dwarf sperm whale (*Kogia sima*) is distributed worldwide in temperate to tropical waters (Caldwell and Caldwell 1989; McAlpine 2009). Pygmy sperm whales and dwarf sperm whales (*K. sima*) are difficult to differentiate at sea (Caldwell and Caldwell 1989; Bloodworth and Odell 2008; McAlpine 2009), and sightings of either species are often categorized as *Kogia* sp. Sightings of *Kogia* whales in the western North Atlantic occur in oceanic waters along the continental shelf break and slope from Canada to Florida (Figure 1; Mullin and Fulling 2003; Roberts et al. 2015). In addition, stranding records for *Kogia* spp. are common from Canada to Florida (Bloodworth and Odell 2008; Berini et al. 2015). Based on the results of passive acoustic monitoring, Hodge et al. (2018) reported that *Kogia* are common in the western North Atlantic in continental shelf break and slope waters between Virginia and Florida, and more common than suggested by visual surveys. Because there are confirmed sightings within waters of Canada and the Bahamas, this is likely a transboundary stock (e.g., Halpin et al. 2009; Dunn 2013; Figure 1).

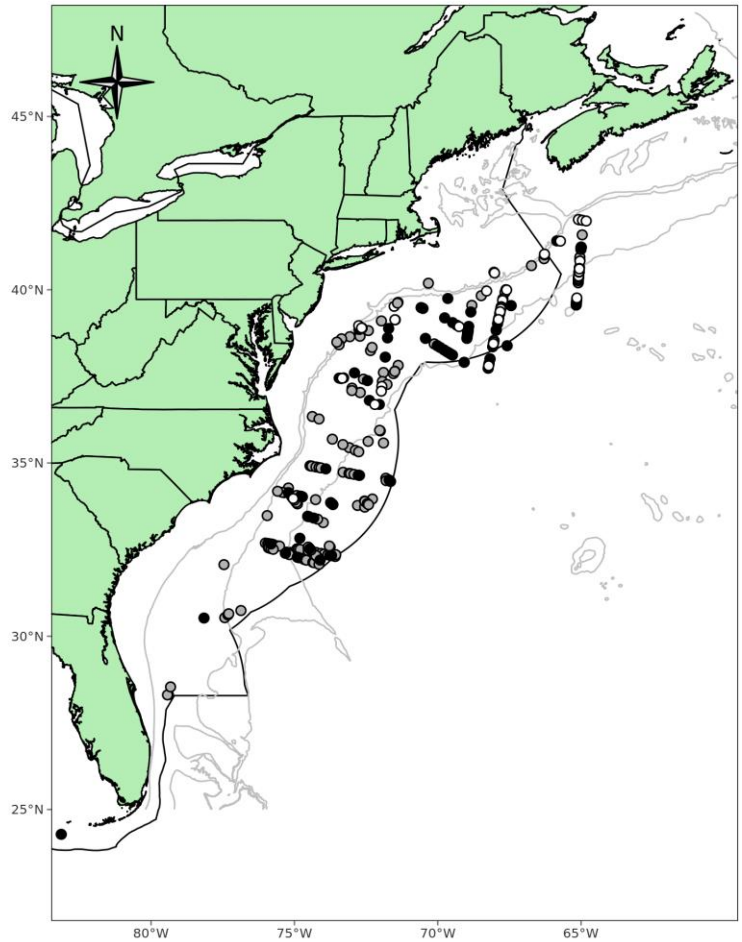

Figure 1. Distribution of Kogia spp. sightings from NEFSC and SEFSC shipboard and aerial surveys during 1995, 1998, 1999, 2002, 2004, 2006, 2007, 2008, 2010, 2011, 2016 and 2021. Black circles represent sightings of dwarf sperm whales; white circles represent sightings of pygmy sperm whales; and gray circles represent sightings of unidentified Kogia. Isobaths are the 200-m, 1,000-m, and 4,000-m depth contours. The darker line indicates the U.S. EEZ.

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

Across its geographic range, including the western North Atlantic, the population biology of dwarf sperm whales is inadequately known (Staudinger et al. 2014). Dwarf sperm whales in the western North Atlantic are managed separately from those in the northern Gulf of Mexico. Although there have been no directed studies of the degree of demographic independence between the two areas, this management structure is consistent with the fact that the western North Atlantic and Gulf of Mexico belong to distinct marine ecoregions (Spalding et al. 2007; Moore and Merrick 2011). Within the western North Atlantic, the range of *Kogia* sightings traverses multiple marine ecoregions (Spalding et al. 2007) and crosses Cape Hatteras, a known biogeographic break for other marine species, so it is possible that multiple demographically independent populations exist within the western North Atlantic stock. Additional morphological, acoustic, genetic, and/or behavioral data are needed to further delineate population structure within the western North Atlantic and across the broader geographic area.

POPULATION SIZE

Total numbers of dwarf sperm whales off the U.S. Atlantic coast are unknown. Because *K. sima* and *K. breviceps* are difficult to differentiate at sea, the reported abundance estimates are for both species of *Kogia* combined. The best estimate for *Kogia* spp. in the western North Atlantic is 9,474 (CV=0.36; Table 1; Garrison and Dias 2023; Palka 2023). This estimate is from summer 2021 surveys covering waters from central Florida to the lower Bay of Fundy. This estimate is almost certainly negatively biased. One component of line transect estimates is *g(0)*, the probability of seeing an animal on the transect line. Estimating *g(0)* is difficult because it consists of accounting for both perception bias (i.e., at the surface but missed) and availability bias (i.e., below the surface while in range of the observers), and many uncertainties (e.g., group size and diving behavior) can confound both (Marsh and Sinclair 1989; Barlow 1999). The long dive times of *Kogia* spp. contribute to a lower probability that animals will be available at the surface and therefore more negative bias. Data on dive-surface behaviors for Kogia spp. were used to estimate and correct for availability bias (Palka et al. 2017), and a two-team approach was used to estimate perception bias (see below). However, Kogia spp. are very difficult to see when at the surface in even moderate sea states, so it is probable that some unquantified negative bias remains in the best abundance estimates.

Earlier Abundance Estimates

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

Recent Surveys and Abundance Estimates

Abundance estimates of 4,548 (CV=0.49) and 3,202 (CV=0.59) *Kogia* spp. were generated from two non-overlapping vessel surveys conducted in the western North Atlantic during the summer of 2016 (Table 1; Garrison 2020; Palka 2020). One survey was conducted from 27 June to 25 August in waters north of 38ºN latitude and consisted of 5,354 km of on-effort trackline along the shelf break and offshore to the U.S. EEZ (NEFSC and SEFSC 2018). The second vessel survey covered waters from Central Florida to approximately 38ºN latitude between the 100-m isobaths and the U.S. EEZ during 30 June–19 August. A total of 4,399 km of trackline was covered on effort (NEFSC and SEFSC 2018). Both surveys utilized two visual teams and an independent observer approach to estimate detection probability on the trackline (Laake and Borchers 2004). Mark-recapture distance sampling was used to estimate abundance. Estimates from the two surveys were combined and CVs pooled to produce a species abundance estimate for the stock area.

More recent abundance estimates of 4,012 (CV=0.54) and 5,462 (CV=0.47) Kogia spp. were generated from vessel surveys conducted in U.S. waters of the western North Atlantic during the summer of 2021 (Table 1; Garrison and Dias 2023; Palka 2023). One survey was conducted from 16 June to 23 August in waters north of 36ºN latitude and consisted of 5,871 km of on-effort trackline along the shelf break and offshore to the outer edge of the U.S. EEZ (NEFSC and SEFSC 2022). The second vessel survey covered waters from central Florida (25ºN latitude) to approximately 38ºN latitude between the 200-m isobaths and the outer edge of the U.S. EEZ during 12 June–31 August. A total of 5,659 km of trackline was covered on effort (NEFSC and SEFSC 2022). 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. For both surveys, a correction was applied (probability at surface = 0.539 [CV=0.307]; Palka et al. 2017) to account for availability bias. Estimates from the two surveys were combined and CVs pooled to produce a species abundance estimate for the stock area.

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

Month/Year	Area	N_{best}	CV
Jun–Aug 2016	New Jersey to lower Bay of Fundy	4,548	0.49

Month/Year	Area	N_{best}	CV
Jun–Aug 2016	central Florida to New Jersey	3,202	0.59
Jun–Aug 2016	central Florida to lower Bay of Fundy (COMBINED)	7,750	0.38
Jun–Aug 2021	New Jersey to lower Bay of Fundy	4,012	0.54
Jun–Aug 2021	central Florida to New Jersey	5,462	0.47
Jun–Aug 2021	**central Florida to lower Bay of Fundy (COMBINED)**	**9,474**	**0.36**

Minimum Population Estimate

The minimum population estimate is the lower limit of the two-tailed 60% confidence interval of the log- normally distributed best abundance estimate. This is equivalent to the 20th percentile of the log-normal distribution as specified by Wade and Angliss (1997). The best estimate of abundance for *Kogia* spp. is 9,474 (CV=0.36). The minimum population estimate for *Kogia* spp. is 7,080 animals (Table 2).

Current Population Trend

There are three available coastwide abundance estimates for *Kogia* spp. from the summers of 2011, 2016, and 2021. 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. An availability bias correction factor (0.539, CV=0.307; Palka et al. 2017) was applied to the 2021 estimate, and in order to do an appropriate trend analysis, this correction was also applied to previous estimates. The resulting estimates were 7,022 (CV=0.25) in 2011; 14,378 (CV=0.20) in 2016; and 9,474 (CV=0.36) in 2021 (Garrison and Dias 2023). (A generalized linear model did not indicate a statistically significant (p=0.728) trend in these estimates. The high level of uncertainty in these estimates limits the ability to detect a statistically significant trend. In addition, interpretation of trends is complicated by two methodological factors. First, the ability to detect *Kogia* spp. visually is highly dependent upon weather and visibility conditions which may contribute to differences between estimates. Second, during 2016 and 2021 the surveys did not use scientific echosounders during some survey periods. Changing the use of echosounders may affect the surfacing/diving patterns of the animals and thus have an influence on the availability of animals to the visual survey teams. Finally, a key uncertainty in this assessment of trend is that interannual variation in abundance may be caused by either changes in spatial distribution associated with environmental variability or changes in the population size of the stock.

CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

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

POTENTIAL BIOLOGICAL REMOVAL

Potential Biological Removal (PBR) is the product of minimum population size, one-half the maximum productivity rate, and a recovery factor (MMPA Sec. 3. 16 U.S.C. 1362; Wade and Angliss 1997). The minimum population size for *Kogia* spp. is 7,080. The maximum productivity rate is 0.04, the default value for cetaceans. The recovery factor is 0.4 because the CV of the average mortality estimate is greater than 0.8 (Wade and Angliss 1997). PBR for western North Atlantic *Kogia* spp. is 57 (Table 2).

Table 2. Best and minimum abundance estimates for western North Atlantic* Kogia *spp. with Maximum Productivity Rate (R_{max}), Recovery Factor (F_r) and PBR.

N_{est}	CV N_{est}	N_{min}	F_r	R_{max}	PBR
9,474	0.36	7,080	0.4	0.04	57

ANNUAL HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

Total annual estimated fishery-related mortality and serious injury to dwarf and pygmy sperm whales combined in the Western North Atlantic during 2017–2021 was 0.8 due to interactions with the large pelagics longline commercial fishery (Table 3). Mean annual mortality and serious injury during 2017–2021 for dwarf sperm whales due to other human-caused actions was presumed to be 0. The minimum total mean annual human-caused mortality and serious injury for dwarf sperm whales is unknown because the estimate of fishery-related mortality and serious injury includes both dwarf and pygmy sperm whales and does not include any estimate for dwarf sperm whales alone. Recorded takes of dwarf and pygmy sperm whales in fisheries in the western North Atlantic are rare. However, observer coverage in the fisheries is relatively low. Furthermore, the likelihood is low that a whale killed at sea due to a fishery interaction or vessel-strike will be recovered (Williams et al. 2011). These factors introduce some uncertainty into estimating the true level of human-caused mortality and serious injury for this stock.

Fishery Information

There are two commercial fisheries that interact, or that could potentially interact, with this stock in the Atlantic Ocean. They are the Category I Atlantic Highly Migratory Species longline and the Atlantic Ocean, Caribbean, Gulf of Mexico large pelagics longline fisheries (Appendix III). Percent observer coverage (percentage of sets observed) for these longline fisheries in the Atlantic for each year during 2017–2021 was 11, 10, 10, 9, and 8, respectively (Table 3).

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

The large pelagics longline fishery operates in the U.S. Atlantic (including Caribbean) and Gulf of Mexico EEZ, 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 2017 to 2021 was 0.8 Kogia spp. (CV=1; Table 3; Garrison and Stokes 2020a; 2020b; 2021; 2023a; 2023b).

Table 3. Summary of the incidental mortality and serious injury of* Kogia *spp. by the U.S. commercial large pelagics longline 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 mortality and serious injury using on-board observer data, the annual estimated mortality and serious injury, the combined annual estimates of mortality and serious injury (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]	Observer Coverage[c]	Observed Serious Injury	Observed Mortality	Estimated Serious Injury	Estimated Mortality	Estimated Combined Mortality	Est. CVs	Mean Annual Mortality
Pelagic Longline	2017	65	Obs. Data, Logbook	11	0	0	0	0	0	-	0.8 (1.00)
	2018	57		10	0	0	0	0	0	-	
	2019	50		10	0	0	0	0	0	-	
	2020	50		9	1	0	4	0	4	1	
	2021	49		8	0	0	0	0	0	-	

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

STATUS OF STOCK

Dwarf sperm whales are not listed as threatened or endangered under the Endangered Species Act, and the western North Atlantic stock is not considered strategic under the Marine Mammal Protection Act. While there is some uncertainty in estimating fishery-related mortality and serious injury for this stock alone, it is believed that U.S. fishery-related mortality and serious injury of *Kogia* spp. is less than 10% of the calculated PBR of *Kogia* spp. and, therefore, can be considered to be insignificant and approaching the zero mortality and serious injury rate. The status of dwarf sperm whales relative to optimum sustainable population is unknown. No statistically significant trend in abundance was detected for *Kogia* spp. over the years 2011–2021; however, there are key methodological issues and uncertainty that limit the ability to evaluate trend.

OTHER FACTORS THAT MAY BE AFFECTING THE STOCK

Strandings

During 2017–2021, 37 dwarf sperm whales were reported stranded along the U.S. East Coast (Table 4; NOAA

National Marine Mammal Health and Stranding Response Database unpublished data, accessed 13 October 2022 (Southeast Region [SER]) and 18 September 2022 (Northeast Region [NER])). Evidence of human interaction was detected for four of the strandings (all were pushed out to sea by members of the public). No evidence of human interaction was detected for 17 strandings, and for the remaining 16 strandings, it could not be determined if there was evidence of human interaction. In addition, there were 16 records of unidentified stranded *Kogia*. Evidence of human interaction was detected for four of the strandings (all were pushed out to sea by members of the public). For the remaining 12 strandings, it could not be determined whether 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.

Stranding data underestimate the extent of human and fishery-related mortality and serious injury because not all of the marine mammals that die or are seriously injured in human interactions wash ashore, or, if they do, they are not all recovered (Peltier et al. 2012; Wells et al. 2015; 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.

Table 4. Dwarf and pygmy sperm whale (**Kogia sima*** ***(Ks),*** **Kogia breviceps** ***(Kb) and*** **Kogia** ***sp. (Sp)) strandings along the U.S. Atlantic coast, 2017–2021. Data are from the NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 13 October 2022 (SER) and 18 September 2022 (NER). Strandings that were not reported to species have been reported as*** **Kogia** ***sp. The level of technical expertise among stranding network personnel varies, and given the potential difficulty in correctly identifying stranded*** **Kogia** ***whales to species, reports to specific species should be viewed with caution.***

STATE	2017			2018			2019			2020			2021			TOTALS		
	Ks	Kb	Sp	Ks	Kb	Sp	Ks	Kb	Sp	Ks	Kb	Sp	Ks	Kb	Sp	Ks	Kb	Sp
Maine	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	1	0	0
Massachusetts	0	0	0	0	2	0	0	0	0	0	0	0	0	0	0	0	2	0
Rhode Island	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	1	0
Connecticut	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	1	0
New York	2	1	0	0	0	0	0	0	1	0	1	0	0	0	0	2	2	1
New Jersey	0	3	0	0	0	0	0	0	0	0	0	0	0	2	0	0	5	0
Delaware	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	1	0
Maryland	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	1	0
Virginia	0	2	1	1	0	0	1	0	0	0	0	0	1	0	0	3	2	1
North Carolina	0	2	1	1	2	2	5	5	0	2	2	0	3	5	1	11	16	4
South Carolina	1	3	0	2	4	0	1	3	1	0	2	1	0	3	2	4	15	4
Georgia	0	2	0	2	1	0	1	0	0	1	0	1	0	2	0	4	5	1
Florida	3	7	1	4	4	0	2	2	2	0	3	2	3	5	0	12	21	5
TOTALS	**6**	**20**	**3**	**10**	**15**	**2**	**10**	**11**	**4**	**4**	**9**	**4**	**7**	**17**	**3**	**37**	**72**	**16**

Habitat Issues

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

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

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

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

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

STOCK DEFINITION AND GEOGRAPHIC RANGE

The pygmy sperm whale (*Kogia breviceps*) is distributed worldwide in temperate and tropical waters (Caldwell and Caldwell 1989; McAlpine 2009). Pygmy sperm whales and dwarf sperm whales (*K. sima*) are difficult to differentiate at sea (Caldwell and Caldwell 1989; Bloodworth and Odell 2008; McAlpine 2009), and sightings of either species are often categorized as *Kogia* sp. Sightings of the two *Kogia* species in the western North Atlantic occur in oceanic waters along the continental shelf break and slope from Canada to Florida (Figure 1; Mullin and Fulling 2003; Roberts et al. 2015). In addition, stranding records for *Kogia* spp. are common from Canada to Florida (Bloodworth and Odell 2008; Berini et al. 2015). Based on the results of passive acoustic monitoring, Hodge et al. (2018) reported that *Kogia* are common in the western North Atlantic in continental shelf break and slope waters between Virginia and Florida, and more common than suggested by visual surveys. Because there are confirmed sightings within waters of Canada and the Bahamas, this is likely a transboundary stock (e.g., Halpin et al. 2009; Lawson and Gosselin 2009; Dunn 2013; Figure 1).

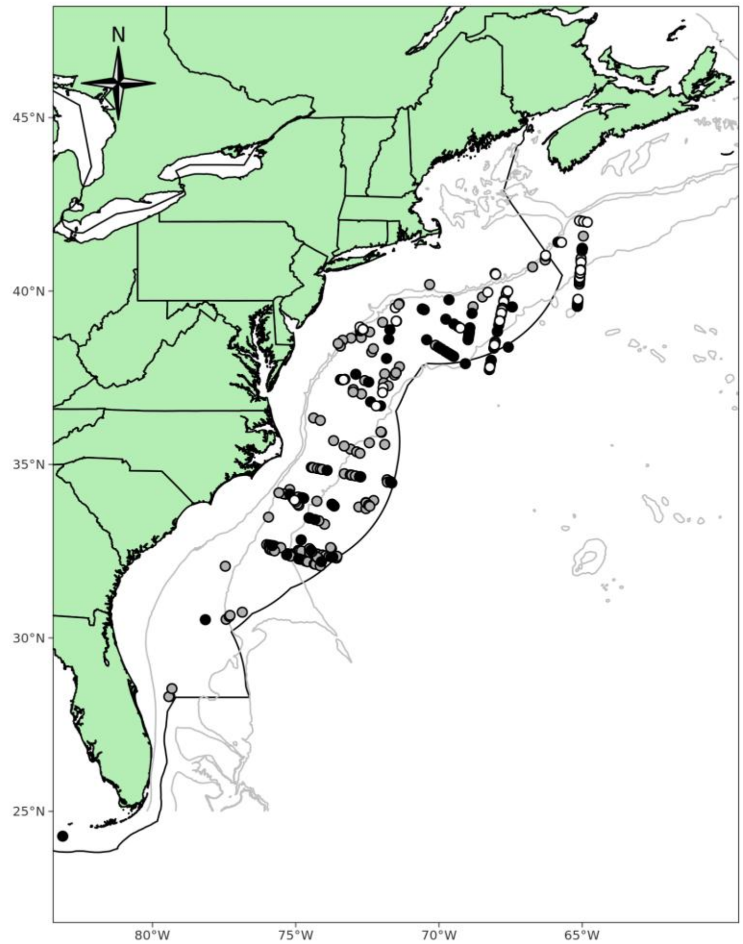

Figure 1. Distribution of Kogia spp. sightings from NEFSC and SEFSC shipboard and aerial surveys during 1995, 1998, 1999, 2002, 2004, 2006, 2007, 2008, 2010, 2011, 2016 and 2021. Black circles represent sightings of dwarf sperm whales; white circles represent sightings of pygmy sperm whales; and gray circles represent sightings of unidentified Kogia. Isobaths are the 200-m, 1,000-m, and 4,000-m depth contours. The darker line indicates the U.S. EEZ.

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

Across its geographic range, including the western North Atlantic, the population biology of pygmy sperm whales is inadequately known (Staudinger et al. 2014). Pygmy sperm whales in the western North Atlantic Ocean are managed separately from those in the northern Gulf of Mexico. Although there have been no directed studies of the degree of demographic independence between the two areas, this management structure is consistent with the fact that the western North Atlantic and Gulf of Mexico belong to distinct marine ecoregions (Spalding et al. 2007; Moore and Merrick 2011). Within the western North Atlantic, the range of *Kogia* sightings traverses multiple marine ecoregions (Spalding et al. 2007) and crosses Cape Hatteras, a known biogeographic break for other marine species, so it is possible that multiple demographically independent populations exist within the western North Atlantic stock. Additional morphological, acoustic, genetic, and/or behavioral data are needed to further delineate population structure within the western North Atlantic and across the broader geographic area.

POPULATION SIZE

Total numbers of pygmy sperm whales off the U.S. Atlantic coast are unknown. Because *K. breviceps* and *K. sima* are difficult to differentiate at sea, the reported abundance estimates are for both species of *Kogia* combined. The best abundance estimate for *Kogia* spp. in the western North Atlantic is 9,474 (CV=0.36; Table 1; Garrison and Dias 2023; Palka 2023). This estimate is from summer 2021 surveys covering waters from central Florida to the lower Bay of Fundy. This estimate is almost certainly negatively biased. One component of line transect estimates is *g(0)*, the probability of seeing an animal on the transect line. Estimating *g(0)* is difficult because it consists of accounting for both perception bias (i.e., at the surface but missed) and availability bias (i.e., below the surface while in range of the observers), and many uncertainties (e.g., group size and diving behavior) can confound both (Marsh and Sinclair 1989; Barlow 1999). The long dive times of *Kogia* spp. contribute to a lower probability that animals will be available at the surface and therefore more negative bias. Data on dive-surface behaviors for Kogia spp. were used to estimate and correct for availability bias (Palka et al. 2017), and a two-team approach was used to estimate perception bias (see below). However, Kogia spp. are very difficult to see when at the surface in even moderate sea states, so it is probable that some unquantified negative bias remains in the best abundance estimates.

Earlier Abundance Estimates

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

Recent Surveys and Abundance Estimates

Abundance estimates of 4,548 (CV=0.49) and 3,202 (CV=0.59) *Kogia* spp. were generated from two non-overlapping vessel surveys conducted in U.S. waters of the western North Atlantic during the summer of 2016 (Table 1; Garrison 2020; Palka 2020). One survey was conducted from 27 June to 25 August in waters north of 38ºN latitude and included 5,354 km of on-effort trackline along the shelf break and offshore to the outer edge of the U.S. EEZ (NEFSC and SEFSC 2018). The second vessel survey covered waters from Central Florida to approximately 38ºN latitude between the 100-m isobaths and the outer edge of the U.S. EEZ from 30 June to 19 August. A total of 4,399 km of trackline was covered on effort (NEFSC and SEFSC 2018). Both surveys utilized two visual teams and an independent observer approach to estimate detection probability on the trackline (Laake and Borchers 2004). Mark-recapture distance sampling was used to estimate abundance (Thomas et al. 2009). Estimates from the two surveys were combined and CVs pooled to produce a species abundance estimate for the stock area.

More recent abundance estimates of 4,012 (CV=0.54) and 5,462 (CV=0.47) Kogia spp. were generated from vessel surveys conducted in U.S. waters of the western North Atlantic during the summer of 2021 (Table 1; Garrison and Dias 2023.; Palka 2023). One survey was conducted from 16 June to 23 August in waters north of 36ºN latitude and consisted of 5,871 km of on-effort trackline along the shelf break and offshore to the outer edge of the U.S. EEZ (NEFSC and SEFSC 2022). The second vessel survey covered waters from central Florida (25ºN latitude) to approximately 38ºN latitude between the 200-m isobaths and the outer edge of the U.S. EEZ during 12 June–31 August. A total of 5,659 km of trackline was covered on effort (NEFSC and SEFSC 2022). 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. For both surveys, a correction was applied (probability at surface = 0.539 [CV=0.307]; Palka et al. 2017) to account for availability bias. Estimates from the two surveys were combined and CVs pooled to produce a species abundance estimate for the stock area.

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

Month/Year	Area	N_{best}	CV
Jun–Aug 2016	New Jersey to lower Bay of Fundy	4,548	0.49
Jun–Aug 2016	Central Florida to New Jersey	3,202	0.59
Jun–Aug 2016	Central Florida to lower Bay of Fundy (COMBINED)	7,750	0.38
Jun–Aug 2021	New Jersey to lower Bay of Fundy	4,012	0.54
Jun–Aug 2021	Central Florida to New Jersey	5,462	0.47
Jun–Aug 2021	**Central Florida to lower Bay of Fundy (COMBINED)**	**9,474**	**0.36**

Minimum Population Estimate

The minimum population estimate is the lower limit of the two-tailed 60% confidence interval of the log- normally distributed best abundance estimate. This is equivalent to the 20th percentile of the log-normal distribution as specified by Wade and Angliss (1997). The best estimate of abundance for *Kogia* spp. is 9,474 (CV=0.36). The minimum population estimate for *Kogia* spp. is 7,080 animals (Table 2).

Current Population Trend

There are three available coastwide abundance estimates for *Kogia* spp. from the summers of 2011, 2016, and 2021. 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. An availability bias correction factor (0.539, CV=0.307; Palka et al. 2017) was applied to the 2021 estimate, and in order to do an appropriate trend analysis, this correction was also applied to previous estimates. The resulting estimates were 7,022 (CV=0.25) in 2011; 14,378 (CV=0.20) in 2016; and 9,474 (CV=0.36) in 2021 (Garrison and Dias 2023). A generalized linear model did not indicate a statistically significant (p=0.728) trend in these estimates. The high level of uncertainty in these estimates limits the ability to detect a statistically significant trend. In addition, interpretation of trends is complicated by two methodological factors. First, the ability to detect *Kogia* spp. visually is highly dependent upon weather and visibility conditions which may contribute to differences between estimates. Second, during 2016 and 2021 the surveys did not use scientific echosounders during some survey periods. Changing the use of echosounders may affect the surfacing/diving patterns of the animals and thus have an influence on the availability of animals to the visual survey teams. Finally, a key uncertainty in this assessment of trend is that interannual variation in abundance may be caused by either changes in spatial distribution associated with environmental variability or changes in the population size of the stock.

CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

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

POTENTIAL BIOLOGICAL REMOVAL

Potential Biological Removal (PBR) is the product of minimum population size, one-half the maximum productivity rate, and a recovery factor (MMPA Sec. 3. 16 U.S.C. 1362; Wade and Angliss 1997). The minimum population size for *Kogia* spp. is 7,080. The maximum productivity rate is 0.04, the default value for cetaceans. The recovery factor is 0.4 because the CV of the average mortality estimate is greater than 0.8 (Wade and Angliss 1997). PBR for western North Atlantic *Kogia* spp. is 57 (Table 2).

Table 2. Best and minimum abundance estimates for western North Atlantic **Kogia** *spp. with Maximum Productivity Rate (R_{max}), Recovery Factor (F_r) and PBR.*

N_{est}	CV N_{est}	N_{min}	F_r	R_{max}	PBR
9,474	0.36	7,080	0.4	0.04	57

ANNUAL HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

Total annual estimated fishery-related mortality and serious injury to dwarf and pygmy sperm whales combined in the Western North Atlantic during 2017–2021 was 0.8 due to interactions with the large pelagics longline commercial fishery (Table 3). Additional mean annual mortality and serious injury for pygmy sperm whales during 2017–2021 due to other human-caused sources was 0.2 (ingestion of debris, see Other Mortality section). The minimum total mean annual human-caused mortality and serious injury for pygmy sperm whales during 2017–2021 was therefore 0.2. This is considered a minimum because the estimate of fishery-related mortality and serious injury includes both dwarf and pygmy sperm whales and does not include any estimate for pygmy sperm whales alone. Recorded takes of dwarf and pygmy sperm whales in fisheries in the western North Atlantic are rare. However, observer coverage in the fisheries is relatively low. Furthermore, the likelihood is low that a whale killed at sea due to a fishery interaction or vessel-strike will be recovered (Williams et al. 2011). These factors introduce some uncertainty into estimating the true level of human-caused mortality and serious injury for this stock.

Fishery Information

There are two commercial fisheries that interact, or that could potentially interact, with this stock in the Atlantic Ocean. They are the Category I Atlantic Highly Migratory Species longline and the Atlantic Ocean, Caribbean, Gulf of Mexico large pelagics longline fisheries (Appendix III). Percent observer coverage (percentage of sets observed) for these longline fisheries in the Atlantic for each year during 2017–2021 was 11, 10, 10, 9, and 8, respectively (Table 3).

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

The large pelagics longline fishery operates in the U.S. Atlantic (including the Caribbean) and Gulf of Mexico EEZ. 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 2017 to 2021 was 0.8 Kogia spp. (CV=1.00; Table 3; Garrison and Stokes 2020a; 2020b; 2021; 2023a; 2023b).

Table 3. Summary of the incidental mortality and serious injury of* Kogia *spp. by the U.S. commercial large pelagics longline 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 mortality and serious injury using on-board observer data, the annual estimated mortality and serious injury, the combined annual estimates of mortality and serious injury (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]	Observer Coverage[c]	Observed Serious Injury	Observed Mortality	Estimated Serious Injury	Estimated Mortality	Estimated Combined Mortality	Est. CVs	Mean Annual Mortality
Pelagic Longline	2017	65	Obs. Data, Logbook	11	0	0	0	0	0	-	0.8 (1.00)
	2018	57		10	0	0	0	0	0	-	
	2019	50		10	0	0	0	0	0	-	
	2020	50		9	1	0	4	0	4	1	
	2021	49		8	0	0	0	0	0	-	

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

Other Mortality

One pygmy sperm whale stranded during 2021 in New Jersey with evidence of human interaction in the form of ingested debris (cloth/fabric). This human interaction was believed to contribute to the stranding and death of the animal (Northeast Regional Marine Mammal Stranding Network; NOAA National Marine Mammal Health and

Stranding Response Database unpublished data, accessed 18 September 2022). Therefore, this mortality was included within the annual human-caused mortality and serious injury total for this stock.

STATUS OF STOCK

Pygmy sperm whales are not listed as threatened or endangered under the Endangered Species Act, and the western North Atlantic stock is not considered strategic under the Marine Mammal Protection Act. While there is some uncertainty in estimating fishery-related mortality and serious injury for this stock alone, it is believed that U.S. fishery-related mortality and serious injury of Kogia spp. is less than 10% of the calculated PBR of *Kogia* spp. and, therefore, can be considered to be insignificant and approaching the zero mortality and serious injury rate. The status of pygmy sperm whales in the U.S. Atlantic EEZ relative to optimum sustainable population is unknown. No statistically significant trend in abundance was detected for Kogia spp. over the years 2011–2021; however, there are key methodological issues and uncertainty that limit the ability to evaluate trend.

OTHER FACTORS THAT MAY BE AFFECTING THE STOCK

Strandings

During 2017–2021, 72 pygmy sperm whales were reported stranded along the U.S. East coast (Table 4; NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 13 October 2022 (Southeast Region [SER]) and 18 September 2022 (Northeast Region [NER])). Evidence of human interaction was detected for eight of the strandings, three of which were pushed out to sea by members of the public and five had ingested plastic or other debris. For one of the cases of ingested debris, this interaction was believed to contribute to the stranding and death of the animal (see Annual Human-Caused Mortality and Serious Injury and Other Mortality sections). No evidence of human interaction was detected for 25 strandings, and for the remaining 39 strandings, it could not be determined if there was evidence of human interaction. In addition, there were 16 records of unidentified *Kogia*. Evidence of human interaction was detected for four of the strandings (all were pushed out to sea by members of the public). For the remaining 12 strandings, it could not be determined whether 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.

Stranding data underestimate the extent of human and fishery-related mortality and serious injury because not all of the marine mammals that die or are seriously injured in human interactions wash ashore, or, if they do, they are not all recovered (Peltier et al. 2012; Wells et al. 2015; 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.

Table 4. Dwarf and pygmy sperm whale (**Kogia sima** ***(Ks),*** **Kogia breviceps** ***(Kb) and*** **Kogia** ***sp. (Sp)) strandings along the U.S. Atlantic coast, 2017–2021. Data are from the NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 13 October 2022 (SER) and 18 September 2022 (NER). Strandings that were not reported to species have been reported as*** **Kogia** ***sp. The level of technical expertise among stranding network personnel varies, and given the potential difficulty in correctly identifying stranded*** **Kogia** ***whales to species, reports to specific species should be viewed with caution.***

STATE	2017			2018			2019			2020			2021			TOTALS		
	Ks	Kb	Sp	Ks	Kb	Sp	Ks	Kb	Sp	Ks	Kb	Sp	Ks	Kb	Sp	Ks	Kb	Sp
Maine	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	1	0	0
Massachusetts	0	0	0	0	2	0	0	0	0	0	0	0	0	0	0	0	2	0
Rhode Island	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	1	0
Connecticut	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	1	0
New York	2	1	0	0	0	0	0	0	1	0	1	0	0	0	0	2	2	1
New Jersey	0	3	0	0	0	0	0	0	0	0	0	0	0	2	0	0	5	0
Delaware	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	1	0

STATE	2017			2018			2019			2020			2021			TOTALS		
Maryland	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	1	0
Virginia	0	2	1	1	0	0	1	0	0	0	0	0	1	0	0	3	2	1
North Carolina	0	2	1	1	2	2	5	5	0	2	2	0	3	5	1	11	16	4
South Carolina	1	3	0	2	4	0	1	3	1	0	2	1	0	3	2	4	15	4
Georgia	0	2	0	2	1	0	1	0	0	1	0	1	0	2	0	4	5	1
Florida	3	7	1	4	4	0	2	2	2	0	3	2	3	5	0	12	21	5
TOTALS	**6**	**20**	**3**	**10**	**15**	**2**	**10**	**11**	**4**	**4**	**9**	**4**	**7**	**17**	**3**	**37**	**72**	**16**

Habitat Issues

The chronic impacts of contaminants (polychlorinated biphenyls [PCBs] and chlorinated pesticides [DDT, DDE, dieldrin, etc.]) on marine mammal reproduction and health are of concern (e.g., Schwacke et al. 2002; Jepson et al. 2016; Hall et al. 2018). Bryan et al. (2012) examined liver and kidney samples from stranded pygmy sperm whales from the U.S. Atlantic and Gulf of Mexico and found that all samples contained mercury concentrations in excess of the USEPA action limits, potentially levels hazardous to the health of whales and putting them at greater risk of disease.

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

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

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

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

STOCK DEFINITION AND GEOGRAPHIC RANGE

The pygmy killer whale is distributed worldwide in tropical and subtropical waters (Jefferson et al. 1994). However, sightings of this species in the western North Atlantic are extremely rare and stranding records are also sparse, probably due to the natural rarity of the species (Baird 2018; Braulik 2018). In the western North Atlantic, strandings are recorded from primarily South Carolina and Georgia, with two from North Carolina and one from Massachusetts, and there have been two sightings during NMFS vessel surveys from 1992 to 2016. In the Hawaiian Islands, there is evidence for limited movement of individuals and for island-associated populations (Baird 2018), and the author suggested it is likely that there is population structure within the species elsewhere. Pygmy killer whales in the western North Atlantic are managed separately from those in the northern Gulf of Mexico. Although there have been no directed studies of the degree of demographic independence between the two areas, this management structure is consistent with evidence for population structure in other areas (Baird 2018) and is further supported because the two stocks occupy distinct marine ecoregions (Spalding et al. 2007; Moore and Merrick 2011). Due to the paucity of sightings in the western North Atlantic, there are insufficient data to determine whether the western North Atlantic stock comprises multiple demographically independent populations. Additional morphological, acoustic, genetic, and/or behavioral data are needed to further delineate population structure within the western North Atlantic and across the broader geographic area. Because there are confirmed sightings within waters of Canada and the Bahamas, this is likely a transboundary stock (e.g., Halpin et al. 2009; Dunn 2013; Harris 2015; Figure 1).

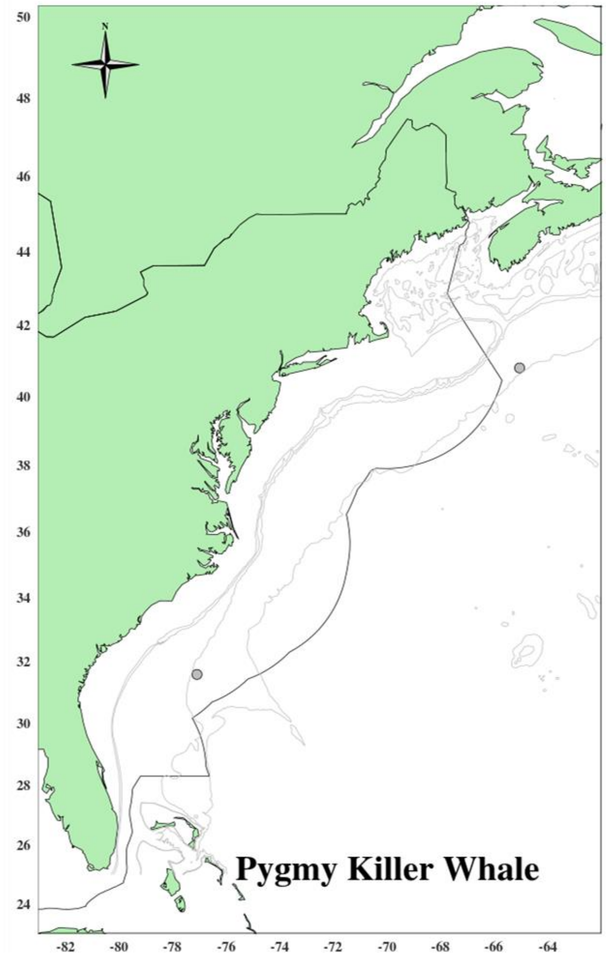

Figure 1. Distribution of pygmy killer whale sightings from NEFSC and SEFSC shipboard (circles) and aerial (squares) surveys during 1992, 1995, 1998, 1999, 2002, 2004, 2006, 2007, 2008, 2010, 2011, 2013, 2016, and 2021. Isobaths are the 100-m, 200-m, 1,000-m and 4,000-m depth contours. The darker line indicates the U.S. EEZ.

POPULATION SIZE

The number of pygmy killer whales off the U.S. Atlantic coast is unknown since they were rarely seen in any surveys. A single group of six pygmy killer whales was sighted in waters ~1500 m deep off Georgia during a 1992 NMFS winter vessel survey (Hansen et al. 1994), and a single pygmy killer whale was sighted in waters ~4000 m deep far offshore of Long Island, New York, during a 2013 NMFS summer vessel survey (NEFSC and SEFSC 2013). Abundances have not been estimated from these single sightings. However, there has been at least one additional sighting of pygmy killer whales off Massachusetts (Halpin et al. 2009; Kenney 2013). Several cruises—a winter 2002 cruise (NMFS 2002), a summer 2005 cruise (NMFS 2005), a summer 2016 cruise (NEFSC and SEFSC 2016), and a summer 2021 cruise (NEFSC and SEFSC 2022)—each had one or two sightings of pygmy killer or melon-headed whales (identity was not confirmed), and these groups were recorded off Cape Hatteras or off the North

Carolina/South Carolina border.

Minimum Population Estimate

Present data are insufficient to calculate a minimum population estimate for this stock (Table 1).

Current Population Trend

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

CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

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

POTENTIAL BIOLOGICAL REMOVAL

Potential Biological Removal level (PBR) is the product of the minimum population size, one-half the maximum productivity rate, and a recovery factor (MMPA Sec. 3.16 U.S.C. 1362; Wade and Angliss 1997). The minimum population size is unknown. The maximum productivity rate is 0.04, the default value for cetaceans. The "recovery" factor, which accounts for endangered, depleted, threatened stocks, or stocks of unknown status relative to optimum sustainable population (OSP), is assumed to be 0.5 because this stock is of unknown status. PBR for the western North Atlantic stock of pygmy killer whales is unknown (Table 1).

Table 1. Best and minimum abundance estimates for the western North Atlantic pygmy killer whale (**Feresa attenuata*****) with Maximum Productivity Rate (R_{max}), Recovery Factor (F_r) and PBR.***

N_{est}	CV N_{est}	N_{min}	F_r	R_{max}	PBR
Unknown	-	Unknown	0.5	0.04	Unknown

ANNUAL HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

Total annual estimated human-caused mortality and serious injury to this stock during 2017–2021 was presumed to be zero, as there were no reports of mortalities or serious injuries to pygmy killer whales in the western North Atlantic. This species is rare and as a result the likelihood of observing a take is very low. Survey effort and observer effort are insufficient to effectively estimate takes for this species.

Fishery Information

There is one commercial fishery that could potentially interact with this stock in the Atlantic Ocean, the Category I Atlantic Ocean, Caribbean, Gulf of Mexico large pelagics longline fishery (Appendix III). Pelagic swordfish, tunas and billfish are the target species of the longline fishery. Percent observer coverage (percentage of sets observed) for this fishery in the Atlantic for each year during 2017–2021 was 11, 10, 10, 9, and 8, respectively. There were no observed mortalities or serious injuries to pygmy killer whales by this fishery in the Atlantic Ocean during 2017–2021 (Garrison and Stokes 2020a; 2020b; 2021; 2023a; 2023b). Detailed fishery information is reported in Appendix III.

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

STATUS OF STOCK

Pygmy killer whales are not listed as threatened or endangered under the Endangered Species Act, and the Western North Atlantic stock is not considered strategic under the Marine Mammal Protection Act. No fishery-related mortality or serious injury has been observed during recent years; however, because this stock is rare, it is unknown whether total fishery-related mortality and serious injury can be considered insignificant and approaching the zero mortality and serious injury rate. The status of pygmy killer whales in the western U.S. Atlantic EEZ relative to optimum sustainable population is unknown. There are insufficient data to determine the population trends for this species.

OTHER FACTORS THAT MAY BE AFFECTING THE STOCK

Strandings

During 2017–2021, four pygmy killer whales were reported stranded along the U.S. East Coast (NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 13 October 2022 (Southeast Region) and 18 September 2022 (Northeast Region)). One stranding occurred in Georgia in 2018, and the remaining three occurred in South Carolina in 2020. Evidence of human interaction was detected for one of the strandings (pushed out to sea by members of the public), and for the remaining three strandings, it could not be determined if there was evidence of human interaction.

Stranding data underestimate the extent of human and fishery-related mortality and serious injury because not all of the marine mammals that die or are seriously injured in human interactions wash ashore, or, if they do, they are not all recovered (Peltier et al. 2012; Wells et al. 2015; 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

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

The chronic impacts of contaminants (polychlorinated biphenyls [PCBs] and chlorinated pesticides [DDT, DDE, dieldrin, etc.]) on marine mammal reproduction and health are of concern (e.g., Schwacke et al. 2002; Jepson et al. 2016; Hall et al. 2018), but research on contaminant levels for this stock is lacking.

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

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

STOCK DEFINITION AND GEOGRAPHIC RANGE

The false killer whale is distributed worldwide throughout warm temperate and tropical oceans (Jefferson et al. 2008). This species is usually sighted in offshore waters but in some cases inhabits waters closer to shore, particularly around oceanic islands (e.g., Hawaii, Baird et al. 2013). While sightings from the U.S. western North Atlantic have been uncommon (Figure 1), the combination of sighting, stranding and bycatch records indicates that this species routinely occurs in the western North Atlantic. False killer whales have been sighted in U.S. Atlantic waters from southern Florida to Maine (Schmidly 1981). There are periodic records (primarily stranding) from southern Florida to Cape Hatteras dating back to 1920 (Schmidly 1981). Most of the records are from the southern half of Florida and include a mass stranding in 1970 that may have numbered as many as 175 individuals (Caldwell et al. 1970; Schmidly 1981). Because there are confirmed sightings within waters of Canada and the Bahamas, this is likely a transboundary stock (e.g., Halpin et al. 2009; Dunn 2013; DFO 2017; Emery 2020; Figure 1).

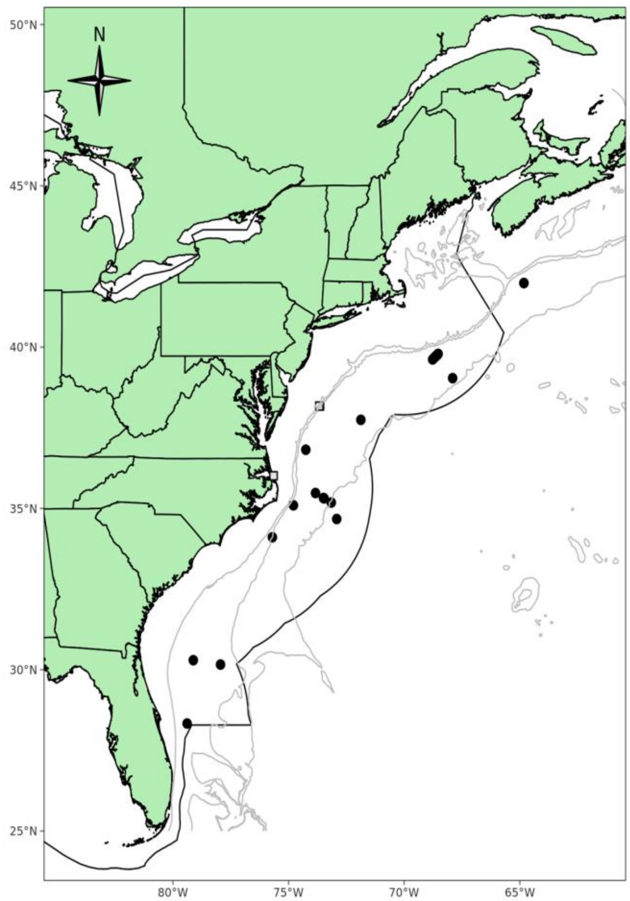

Figure 1. Distribution of false killer whale sightings from NEFSC and SEFSC shipboard (circles) and aerial (squares) surveys during 1995, 1998, 1999, 2002, 2004, 2006, 2007, 2008, 2010, 2011, 2016, and 2021. Isobaths are the 200-m, 1,000-m, and 4,000-m depth contours. The darker line indicates the U.S. EEZ.

Genetic analyses (Chivers et al. 2007; Martien et al. 2014) indicate false killer whales exhibit significant population structuring in the Pacific, with restricted gene flow among whales sampled near the main Hawaiian Islands, the Northwestern Hawaiian Islands, and pelagic waters of the eastern and the central North Pacific. Martien et al. (2014) also found their two Atlantic samples to be genetically divergent from those in the Pacific. False killer whales in the western North Atlantic are managed separately from those in the northern Gulf of Mexico. Although there have been no directed studies of the degree of demographic independence between the two areas, this management structure is consistent with evidence for strong population structuring in other areas (Martien et al. 2014) and further supported because the two stocks occupy distinct marine ecoregions (Spalding et al. 2007; Moore and Merrick 2011). Given the paucity of sightings, there are insufficient data to determine whether the western North Atlantic stock comprises multiple demographically independent populations. Additional morphological, acoustic, genetic, and/or behavioral data are needed to further delineate population structure within the western North Atlantic and across the broader geographic area.

POPULATION SIZE

The best available abundance estimate for western North Atlantic false killer whales is 1,298 (CV=0.72; Table 1; Garrison and Dias 2023; Palka 2023). This estimate is from summer 2021 surveys covering waters from central Florida to the lower Bay of Fundy.

Earlier Abundance Estimates

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

Recent Surveys and Abundance Estimates

Abundance estimates of 1,182 (CV=0.63) and 609 (CV=1.08) false killer whales were generated from vessel surveys conducted in U.S. waters of the western North Atlantic during the summer of 2016 (Table 1; Garrison 2020; Palka 2020). One survey was conducted from 27 June to 25 August in waters north of 38ºN latitude and consisted of 5,354 km of on-effort trackline along the shelf break and offshore to the U.S. EEZ (NEFSC and SEFSC 2018). The second vessel survey covered waters from Central Florida to approximately 38ºN latitude between the 100-m isobaths and the U.S. EEZ during 30 June–19 August. A total of 4,399 km of trackline was covered on effort (NEFSC and SEFSC 2018). Both surveys utilized two visual teams and an independent observer approach to estimate detection probability on the trackline (Laake and Borchers 2004). Mark-recapture distance sampling was used to estimate abundance. It should be noted that the abundance estimate from the second vessel survey was based on a single sighting and therefore has a very high uncertainty.

More recent abundance estimates of 753 (CV=1.13) and 545 (CV=0.68) false killer whales were generated from vessel surveys conducted in U.S. waters of the western North Atlantic during the summer of 2021 (Table 1; Garrison and Dias 2023; Palka 2023). One survey was conducted from 16 June to 23 August in waters north of 36ºN latitude and consisted of 5,871 km of on-effort trackline along the shelf break and offshore to the outer edge of the U.S. EEZ (NEFSC and SEFSC 2022). The second vessel survey covered waters from central Florida (25ºN latitude) to approximately 38ºN latitude between the 200-m isobaths and the outer edge of the U.S. EEZ during 12 June–31 August. A total of 5,659 km of trackline was covered on effort (NEFSC and SEFSC 2022). Both surveys utilized two visual teams and an independent observer approach to estimate detection probability on the trackline (Laake and Borchers 2004). Mark-recapture distance sampling was used to estimate abundance. Estimates from the two surveys were combined and CVs pooled to produce a species abundance estimate for the stock area.

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

Month/Year	Area	N_{best}	CV
Jun–Aug 2016	New Jersey to lower Bay of Fundy	1,182	0.63
Jun–Aug 2016	Central Florida to New Jersey	609	1.08
Jun–Aug 2016	Central Florida to lower Bay of Fundy (COMBINED)	1,791	0.56
Jun–Aug 2021	New Jersey to lower Bay of Fundy	753	1.13
Jun–Aug 2021	Central Florida to New Jersey	545	0.68
Jun–Aug 2021	**Central Florida to lower Bay of Fundy (COMBINED)**	1,298	0.72

Minimum Population Estimate

The minimum population estimate is the lower limit of the two-tailed 60% confidence interval of the log-normally distributed best abundance estimate. This is equivalent to the 20th percentile of the log-normal distribution as specified by Wade and Angliss (1997). The best estimate of abundance for false killer whales is 1,298 (CV=0.72). The minimum population estimate for false killer whales is 755 (Table 2).

Current Population Trend

There are three available coastwide abundance estimates for false killer whales from the summers of 2011, 2016, and 2021. Each of these is derived from surveys with similar survey designs and all three used the two-team independent observer approach to estimate abundance. The resulting estimates were 442 (CV=1.06) in 2011; 1,791 (CV=0.56) in 2016; and 1,298 (CV=0.72) in 2021 (Garrison 2020; Garrison and Dias 2023; Palka 2020; Palka 2023). A generalized linear model did not indicate a statistically significant (p=0.786) trend in these estimates. The high level of uncertainty in these estimates limits the ability to detect a statistically significant trend. A key uncertainty in this assessment of trend is that interannual variation in abundance may be caused by either changes in spatial distribution associated with environmental variability or changes in the population size of the stock.

CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

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

POTENTIAL BIOLOGICAL REMOVAL

Potential Biological Removal (PBR) is the product of minimum population size, one-half the maximum net productivity rate, and a recovery factor (MMPA Sec. 3. 16 U.S.C. 1362; Wade and Angliss 1997). The minimum population size is 755. The maximum productivity rate is 0.04, the default value for cetaceans. The "recovery" factor, which accounts for endangered, depleted, threatened stocks, or stocks of unknown status relative to optimum sustainable population (OSP) is assumed to be 0.5 because this stock is of unknown status. PBR for the western North Atlantic false killer whale stock is 7.6 (Table 2).

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

N_{est}	CV N_{est}	N_{min}	F_r	R_{max}	PBR
1,298	0.72	755	0.5	0.04	7.6

ANNUAL HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

Total annual estimated human-caused mortality and serious injury to this stock during 2017–2021 was presumed to be zero, as there were no reports of mortalities or serious injuries to false killer whales in the western North Atlantic. Recorded takes of false killer whales in fisheries in the western North Atlantic are extremely rare. However, observer coverage in the fisheries is relatively low. Furthermore, the likelihood is low that a dolphin killed at sea due to a fishery interaction or vessel strike will be recovered (Williams et al. 2011). These factors introduce some uncertainty into estimating the true level of human-caused mortality and serious injury for this stock.

Fishery Information

There are two commercial fisheries that interact, or that could potentially interact, with this stock in the Atlantic Ocean. These are the Category I Atlantic Highly Migratory Species longline and the Atlantic Ocean, Caribbean, Gulf of Mexico large pelagics longline fisheries (Appendix III). Percent observer coverage (percentage of sets observed) for these longline fisheries in the Atlantic for each year during 2017–2021 was 11, 10, 10, 9, and 8, respectively.

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

The Atlantic Ocean, Caribbean, Gulf of Mexico large pelagics longline fishery operates in the U.S. Atlantic (including Caribbean) and Gulf of Mexico EEZ, and pelagic swordfish, tunas and billfish are the target species. There were no observed mortalities or serious injuries to false killer whales by this fishery in the Atlantic Ocean during 2017–2021 (Garrison and Stokes 2020a; 2020b; 2021; 2023a; 2023b).

STATUS OF STOCK

False killer whales are not listed as threatened or endangered under the Endangered Species Act and the western North Atlantic stock is not considered strategic under the Marine Mammal Protection Act. No fishery-related mortality or serious injury has been observed in recent years; therefore, total fishery-related mortality and serious injury can be considered insignificant and approaching the zero mortality and serious injury rate. The status of false killer whales in the U.S. EEZ relative to optimum sustainable population is unknown. There was no statistically significant trend in population size for this species; however, the high level of uncertainty in the estimates limits the ability to detect a statistically significant trend.

OTHER FACTORS THAT MAY BE AFFECTING THE STOCK

Strandings

Historically, there have been intermittent false killer whale strandings along the U.S. East Coast, however, during 2017–2021, none were reported (NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 13 October 2022 (Southeast Region) and 18 September 2022 (Northeast Region)).

Habitat Issues

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

The chronic impacts of contaminants (polychlorinated biphenyls [PCBs] and chlorinated pesticides [DDT, DDE, dieldrin, etc.]) on marine mammal reproduction and health are of concern (e.g., Schwacke et al. 2002; Jepson et al. 2016; Hall et al. 2018), but research on contaminant levels for this stock is lacking.

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

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

STOCK DEFINITION AND GEOGRAPHIC RANGE

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

Cuvier's beaked whale sightings have occurred principally along the continental shelf edge in the Mid-Atlantic region off the northeast U.S. coast (CETAP 1982; Waring et al. 1992; Waring et al. 2001; Hamazaki 2002; Palka 2006) (Figure 1). Monthly aerial surveys conducted off Cape Hatteras between 2011 and 2015 recorded Cuvier's beaked whales sighted during every month of the year (McLellan et al. 2018) and acoustic recordings confirm consistent year-round presence (Stanistreet et al. 2017).

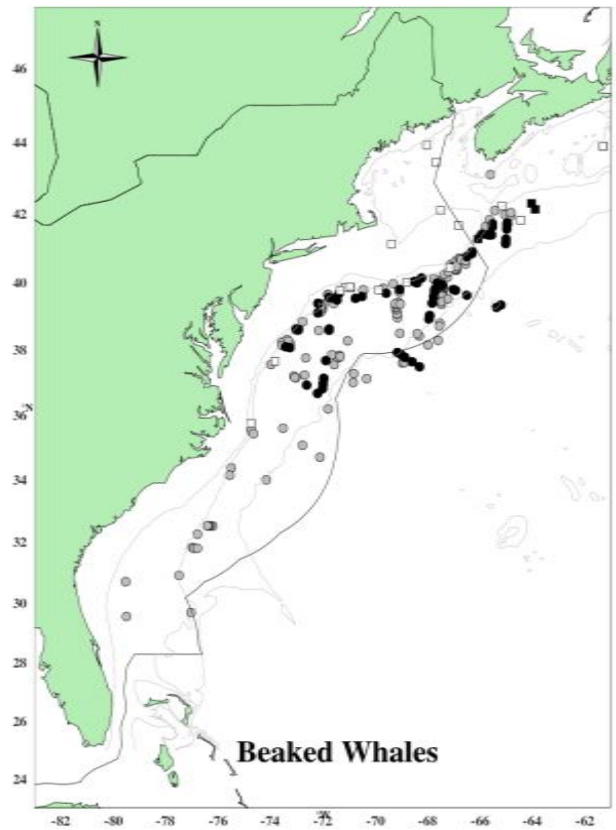

Figure 1. Distribution of beaked whale sightings (includes Ziphius and Mesoplodon spp.) from NEFSC and SEFSC shipboard and aerial surveys during the summers of 1995, 1998, 1999, 2002, 2004, 2006, 2007, 2008, 2010, 2011 and 2016, 2021 and Depatment of Fisheries and Oceans Canada 2007 TNASS and 2016 NAISS surveys. Isobaths are the 200-m, 1000-m and 4000-m depth contours. Circle symbols represent shipboard sightings and squares are aerial sightings. Black symbols are sightings identified as Cuvier's beaked whales.

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

POPULATION SIZE

The best abundance estimate for Cuvier's beaked whales is the northeast 2021 survey described below—4,670 (CV=0.24). This estimate, derived from shipboard surveys, covers most of this stock's known range.

Recent Surveys and Abundance Estimates

Abundance estimates of 3,897 (CV=0.47) and 1,847 (CV=0.49) Cuvier's beaked whales (not including Mesoplodon spp.) were generated from vessel surveys conducted in U.S. waters of the western North Atlantic during the summer of 2016 (Table 1; Garrison 2020; Palka 2020). One survey was conducted from 27 June to 25 August in waters north of 38ºN latitude and consisted of 5,354 km of on-effort trackline along the shelf break and offshore to the outer limit of the U.S. EEZ (NEFSC and SEFSC 2018). The second vessel survey covered waters from Central Florida to approximately 38ºN latitude between the 100-m isobath and the outer limit of the U.S. EEZ during 30 June–19 August. A total of 4,399 km of trackline was covered on effort (NEFSC and SEFSC 2018). Both surveys utilized two visual teams and an independent observer approach to estimate detection probability on the trackline (Laake and Borchers 2004). Mark-recapture distance sampling was used to estimate abundance. Estimates from the two surveys were combined and CVs pooled to produce an abundance estimate for the stock area, yielding a combined total of 5,744 Cuvier's beaked whales (CV=0.36). These estimates are known to be biased low due to the fact that unidentified Ziphiidae abundance was estimated at 3,755 (CV=0.42) in the NE and at 2,812 (CV=0.43) in the SE, and these numbers likely include an unknown number of Cuvier's beaked whales.

More recent abundance estimates of 1,742 (CV=0.39) and 2,928 (CV=0.31) Cuvier's beaked whale were generated from vessel surveys conducted in U.S. waters of the western North Atlantic during the summer of 2021 (Table 1; Garrison and Aichinger-Dias 2023; Palka 2023). One survey was conducted from 16 June to 23 August in waters north of 36ºN latitude and consisted of 5,871 km of on-effort trackline along the shelf break and offshore to the outer edge of the U.S. EEZ (NEFSC and SEFSC 2022). The second vessel survey covered waters from central Florida (25ºN latitude) to approximately 38ºN latitude between the 200-m isobaths and the outer edge of the U.S. EEZ during 12 June–31 August. A total of 5,659 km of trackline was covered on effort (NEFSC and SEFSC 2022). Both surveys utilized two visual teams and an independent observer approach to estimate detection probability on the trackline (Laake and Borchers 2004). Mark-recapture distance sampling was used to estimate abundance. Estimates from the two surveys were combined and CVs pooled to produce a species abundance estimate for the stock area.

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

Month/Year	Area	N_{best}	CV
Jun–Sep 2016	Central Virginia to lower Bay of Fundy	3,897	0.47
Jun–Aug 2016	Central Florida to Virginia	1,847	0.49
Jun–Aug 2016	Central Florida to lower Bay of Fundy (COMBINED)	5,744	0.36
Jun–Aug 2021	New Jersey to lower Bay of Fundy	1,742	0.39
Jun–Aug 2021	Central Florida to New Jersey	2,928	0.31
Jun–Aug 2021	**Central Florida to lower Bay of Fundy (COMBINED)**	**4,670**	**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 Cuvier's beaked whales is 4,670 (CV=0.24). The minimum population estimate for Cuvier's beaked whales in the western North Atlantic is 3,817.

Current Population Trend

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

CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

Current and maximum net productivity rates are unknown for this stock. Life history parameters that could be used to estimate net productivity include: length at birth is 2 to 3 m, length at sexual maturity is 6.1m for females, and 5.5 m for males, maximum age expressed in dental growth layer groups (GLG's) which are presumed to each correspond to a single year of growth is 30 for for females and 36 for males (Mitchell 1975; Mead 1984; Houston 1990).

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

POTENTIAL BIOLOGICAL REMOVAL

Potential Biological Removal (PBR) is the product of minimum population size, one-half the maximum productivity rate, and a recovery factor (MMPA Sec. 3. 16 U.S.C. 1362; Wade and Angliss 1997). The minimum population size for Cuvier's beaked whales is 3,817. The maximum productivity rate is 0.04, the default value for cetaceans. The recovery factor, which accounts for endangered, depleted, threatened stocks, or stocks of unknown status relative to optimum sustainable population (OSP) is assumed to be 0.5. PBR for Cuvier's beaked whales is 38.

Table 2. Best and minimum abundance estimates for the Western North Atlantic stock of Cuvier's beaked whales with Maximum Productivity Rate (R_{max}), Recovery Factor (F_r) and PBR.

N_{est}	CV	N_{min}	F_r	R_{max}	PBR
4,670	0.24	3,817	0.5	0.04	38

ANNUAL HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

The 2017-2021 minimum annual rate of human-caused mortality of Cuvier's beaked whales averaged 0.2 animals per year. This is from 1 stranding record that reported signs of human interaction (plastic ingestion; Table 3).

Fishery Information

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

In 2017–2021, estimated annual average fishery-related mortality or serious injury of this stock in U.S. fisheries was 0 for all beaked whales. Detailed U.S. fishery information is reported in Appendix III.

Other Mortality

During 2017-2021, 4 Cuvier's beaked whales stranded along the U.S. Atlantic coast without evidence of human interaction (Table 3; NOAA National Marine Mammal Health and Stranding Response Database, accessed 15 October 2022).

Table 3. Cuvier's beaked whale (Ziphius cavirostris) strandings along the U.S. Atlantic coast from 2017-2021.

State	2017	2018	2019	2020	2021	Total
North Carolina	1	0	0	0	0	1
South Carolina	0	0	0	1	0	1
Florida[a]	1	1	0	0	0	2
Total	**2**	**1**	**0**	**1**	**0**	**4**

a. Animal in Florida in 2018 had trash in stomach.

STATUS OF STOCK

The western North Atlantic stock of Cuvier's beaked whale is not a strategic stock because average annual human-related mortality and serious injury does not exceed PBR. The total U.S. fishery mortality and serious injury for this group of species is less than 10% of the calculated PBR and, therefore, can be considered to be insignificant and approaching zero mortality and serious injury rate. The status of Cuvier's beaked whale relative to OSP is unknown. This species is not listed as threatened or endangered under the Endangered Species Act.

OTHER FACTORS THAT MAY BE AFFECTING THE STOCK

Habitat Issues

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

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

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

Fourteen beaked whales (mostly Cuvier's beaked whales but also including Gervais' and Blainville's beaked whales) stranded in the Canary Islands in 2002 (Cox et al. 2006, Fernandez et al. 2005; Martin et al. 2004). Gas bubble-associated lesions and fat embolism were found in necropsied animals from this event, leading researchers to link nitrogen supersaturation with sonar exposure (Fernandez et al. 2005).

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). Chavez-Rosales et al. (2022) documented an overall 178 km northeastward spatial distribution shift of the seasonal core habitat of Northwest Atlantic cetaceans that was related to changing habitat/climatic factors. Results varied by season and species. This study used sightings data collected during seasonal aerial and shipboard line transect abundance surveys during 2010 to 2017. There is uncertainty in how, if at all, the changes in distribution and population size of cetacean species may interact with changes in distribution of prey species and how the ecological shifts will affect human impacts to the species.

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

STOCK DEFINITION AND GEOGRAPHIC RANGE

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

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

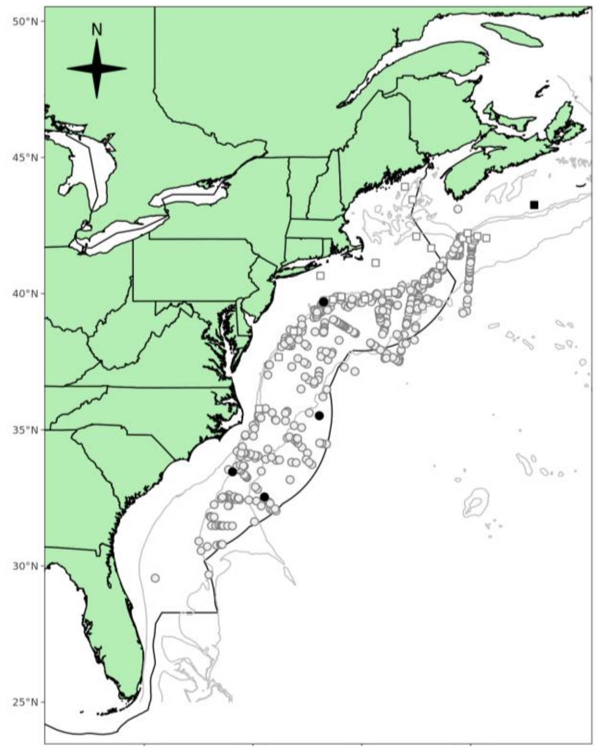

Figure 1. Distribution of beaked whale (includes* Ziphius *and* Mesoplodon *spp.) sightings from NEFSC and SEFSC shipboard and aerial surveys during the summers of 1995, 1998, 1999, 2002, 2004, 2006, and 2007, 2008, 2010, 2011, 2016, and 2021 and Department of Fisheries and Oceans Canada 2007 TNASS and 2016 NAISS surveys. Isobaths are the 200-m, 1000-m and 4000-m depth contours. Circle symbols represent shipboard sightings and squares are aerial sightings. Black symbols are sightings identified as Blainsville's beaked whales.

POPULATION SIZE

The best abundance estimate for Blainville's beaked whales is the 2021 survey estimate: 2,936 (CV=0.26). This estimate, derived from shipboard surveys, covers most of this stock's known range. In the 2021 survey, improvements to field protocols for both visual observers and passive acoustic monitoring of *Mesoplodon* spp. facilitated differentiation of species during encounters. This enabled abundance estimates to be calculated for each species individually rather than grouping together at the genus level.

Recent Surveys and Abundance Estimates

Abundance estimates of 6,760 (CV=0.37) and 3,347 (CV=0.29) *Mesoplodon* spp. beaked whales (not including *Ziphius*) were generated from vessel surveys conducted in U.S. waters of the western North Atlantic during the summer of 2016 (Table 1; Garrison 2020; Palka 2020). One survey was conducted from 27 June to 25 August in waters north of 38ºN latitude and consisted of 5,354 km of on-effort trackline along the shelf break and offshore to the outer limit of the U.S. EEZ (NEFSC and SEFSC 2018). The second vessel survey covered waters from Central Florida to approximately 38ºN latitude between the 100-m isobath and the outer limit of the U.S. EEZ during 30 June–19 August. A total of 4,399 km of trackline was covered on effort (NEFSC and SEFSC 2018). Both surveys utilized two visual teams and an independent observer approach to estimate detection probability on the trackline (Laake and Borchers 2004). Mark-recapture distance sampling was used to estimate abundance. Estimates from the two surveys were combined and CVs pooled to produce an abundance estimate for the stock area, yielding a combined total of 10,107 *Mesoplodon* beaked whales (CV=0.27). These estimates are known to be biased low due to the fact that unidentified Ziphiidae abundance was estimated at 3,755 (CV=0.42) in the NE and at 2,812 (CV=0.43) in the SE. These estimates likely include an unknown number of *Mesoplodon* beaked whales.

A more recent abundance estimate of 2,936 (CV=0.26) Blainville's beaked whales was generated from vessel surveys conducted in U.S. waters of the western North Atlantic during the summer of 2021 (Table 1; Garrison and Dias 2023; Palka 2023). One survey was conducted from 16 June to 23 August in waters north of 36ºN latitude and consisted of 5,871 km of on-effort trackline along the shelf break and offshore to the outer edge of the U.S. EEZ (NEFSC and SEFSC 2022). The second vessel survey covered waters from central Florida (25ºN latitude) to approximately 38ºN latitude between the 200-m isobaths and the outer edge of the U.S. EEZ during 12 June–31 August. A total of 5,659 km of trackline was covered on effort (NEFSC and SEFSC 2022). Both surveys utilized two visual teams and an independent observer approach to estimate detection probability on the trackline (Laake and Borchers 2004). Mark-recapture distance sampling was used to estimate abundance. Estimates from the two surveys were combined and CVs pooled to produce a species abundance estimate for the stock area.

Table 1.Summary of abundance estimates for the Mesoplodon beaked whales (2016 surveys) and Blainville's beaked whales (2021 surveys). Month, year, area covered during each abundance survey, resulting abundance estimate (N_{best}) and coefficient of variation (CV) are represented. The estimate considered best is in bold font.

Month/Year	Area	N_{best}	CV
Jun-Sep 2016	Central Virginia to lower Bay of Fundy (*Mesoplodon* spp.)	6,760	0.37
Jun-Aug 2016	Central Florida to Virginia (*Mesoplodon* spp.)	3,347	0.29
Jun-Aug 2016	Central Florida to lower Bay of Fundy (COMBINED, (*Mesoplodon* spp.)	10,107	0.27
Jun–Aug 2021	New Jersey to lower Bay of Fundy (*M. densirostris* only)	0	0
Jun–Aug 2021	Central Florida to New Jersey (*M. densirostris* only)	2,936	0.26
Jun–Aug 2021	**Central Florida to lower Bay of Fundy (COMBINED, *M. densirostris* only)**	**2,936**	**0.26**

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 Blainville's beaked whales is 2,936 (CV=0.26). The minimum population estimate for Blainville's beaked whales in the western North Atlantic is 2,374.

Current Population Trend

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

CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

Current and maximum net productivity rates are unknown for this stock. *Mesoplodon densirostris* life history parameters that could be used to estimate net productivity include: length at birth of up to 1.9 m, maximum reported adult length of 4.7, and minimum reported age at sexual maturity of 9 dental growth layer groups (GLG's), which may each correspond to a single year of growth (Mead 1984).

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

POTENTIAL BIOLOGICAL REMOVAL

Potential Biological Removal (PBR) is the product of minimum population size, one-half the maximum productivity rate, and a recovery factor (MMPA Sec. 3. 16 U.S.C. 1362; Wade and Angliss 1997). The minimum population size for Blainville's beaked whales is 2,374. The maximum productivity rate is 0.04, the default value for cetaceans. The recovery factor, which accounts for endangered, depleted, or threatened stocks, or stocks of unknown status relative to optimum sustainable population (OSP) is assumed to be 0.5. PBR for Blainville's beaked whales in the western North Atlantic is 24.

Table 2. Best and minimum abundance estimates for Blainville's beaked whales (**Mesoplodon densirostris*****) of the Western North Atlantic with Maximum Productivity Rate (R_{max}), Recovery Factor (F_r) and PBR.***

N_{est}	CV N_{est}	N_{min}	F_r	R_{max}	PBR
2,936	0.26	2,374	0.5	0.04	24

ANNUAL HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

The 2017–2021 total average estimated annual human-caused mortality of Blainville's beaked whales in the U.S. Atlantic EEZ is 0.2 based on one stranded animal likely killed in 2017 by plastic ingestion (Table 3).

Table 3. Total annual estimated average human-caused mortality and serious injury for the North Atlantic stock of Blainville's beaked whales (**Mesoplodon densirostris*****).***

Years	Source	Annual Avg.	CV
2017–2021	U.S. fisheries using observer data	0	NA
2017–2021	Possible non-fishery human-caused stranding mortalities	0.2	NA
TOTAL		NA	NA

Fishery Information

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

In 2017–2021, estimated annual average fishery-related mortality or serious injury of this stock in U.S. fisheries was 0 for all beaked whales.

Other Mortality

From 2017–2021, a total of 4 Blainville's beaked whales stranded along the U.S. Atlantic coast between Florida and Massachusetts (NOAA National Marine Mammal Health and Stranding Response Database, accessed 19 October 2022, Table 4). One animal in 2017 that stranded in Florida was classified as a human interaction due to plastic ingestion.

*Table 4. Blainville's beaked whale (***Mesoplodon densirostris***) strandings along the U.S. Atlantic coast.*

State	2017	2018	2019	2020	2021	Total
Maine	0	0	0	1	0	1
North Carolina	0	0	1	0	0	1
South Carolina	0	1	0	0	0	1
Florida[a]	1	0	0	0	0	1
Total	**1**	**1**	**1**	**1**	**0**	**4**

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

STATUS OF STOCK

Blainville's beaked whales are not listed as threatened or endangered under the Endangered Species Act and the western North Atlantic stock of Blainville's beaked whale is not considered strategic under the Marine Mammal Protection Act, although there are insufficient data to determine the population size or trends. The permanent closure of the pelagic drift gillnet fishery has eliminated the principal known source of incidental fishery mortality, and a single 2017 stranding record was the only human-related mortality and serious injury observed during the recent 5-year (2017–2021) period. Therefore, total U.S. fishery-related mortality and serious injury rate is considered insignificant and approaching zero. The status of Blainville's beaked whales relative to OSP is unknown.

OTHER FACTORS THAT MAY BE AFFECTING THE STOCK

Habitat issues

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

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

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

Fourteen beaked whales (mostly Cuvier's beaked whales but also including Gervais' and Blainville's beaked whales) stranded in the Canary Islands in 2002 (Martin et al. 2004; Fernandez et al. 2005; Cox et al. 2006). Gas bubble-associated lesions and fat embolism were found in necropsied animals from this event, leading researchers to link nitrogen supersaturation with sonar exposure (Fernandez et al. 2005).

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). Chavez-Rosales et al. (2022) documented an overall 178 km northeastward spatial distribution shift of the seasonal core habitat of Northwest Atlantic cetaceans that was related to changing habitat/climatic factors. Blainville's beaked whales were not specifically analyzed in that study, and there is uncertainty in how, if at all, the changes in distribution and population size of cetacean species may interact with changes in distribution of prey species and how the ecological shifts will affect human impacts to the species.

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

STOCK DEFINITION AND GEOGRAPHIC RANGE

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

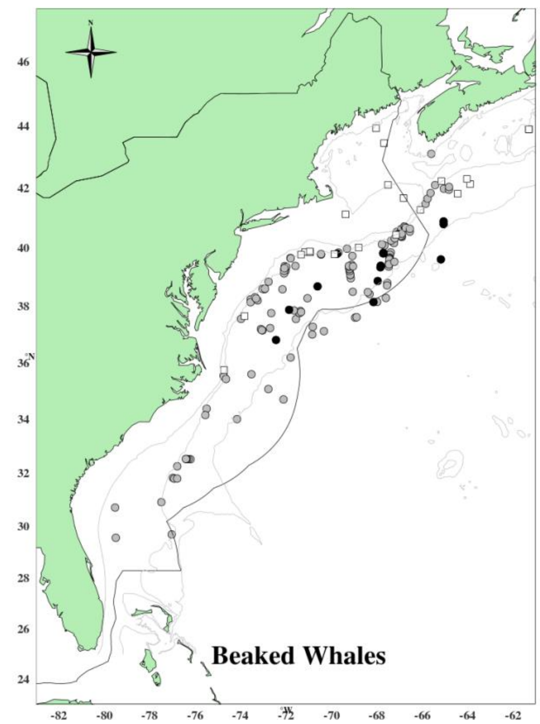

Figure 1. Distribution of beaked whale (includes* Ziphius *and* Mesoplodon *spp.) sightings from NEFSC and SEFSC shipboard and aerial surveys during the summers of 1995, 1998, 1999, 2002, 2004, 2006, and 2007, 2008, 2010, 2011 and 2016 and DFO's 2007 TNASS and 2016 NAISS surveys. Isobaths are the 200-m,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

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

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

POPULATION SIZE

The best abundance estimate for Gervais' beaked whales is the sum of the 2021 survey estimate – 8,595 (CV=0.24). This estimate, derived from shipboard surveys, covers most of this stock's known range. In the 2021 survey, improvements to field protocols for both visual observers and passive acoustic monitoring of *Mesoplodon* spp. facilitated differentiation of species during encounters. This enabled abundance estimates to be calculated for each species individually rather than grouping together at the genus level.

Recent Surveys and Abundance Estimates

Abundance estimates of 6,760 (CV=0.37) and 3,347 (CV=0.29) undifferentiated beaked whales (*Ziphius* and *Mesoplodon* spp.) were generated from vessel surveys conducted in U.S. waters of the western North Atlantic during the summer of 2016 (Table 1; Garrison 2020; Palka 2020). One survey was conducted from 27 June to 25 August in waters north of 38ºN latitude and consisted of 5,354 km of on-effort trackline along the shelf break and offshore to the outer limit of the U.S. EEZ (NEFSC and SEFSC 2018). The second vessel survey covered waters from Central Florida to approximately 38ºN latitude between the 100-m isobath and the outer limit of the U.S. EEZ during 30 June–19 August. A total of 4,399 km of trackline was covered on effort (NEFSC and SEFSC 2018). Both surveys utilized two visual teams and an independent observer approach to estimate detection probability on the trackline (Laake and

Borchers 2004). Mark-recapture distance sampling was used to estimate abundance. Estimates from the two surveys were combined and CVs pooled to produce an abundance estimate for the stock area, yielding a combined total of 10,107 *Mesoplodon* beaked whales (CV=0.27). These estimates are known to be biased low due to the fact that unidentified Ziphiidae abundance was estimated at 3,755 (CV=0.42) in the NE and at 2,812 (CV=0.43) in the SE. These estimates likely include an unknown number of *Mesoplodon* beaked whales.

A more recent abundance estimate of 8,595 (CV=0.24) Gervais' beaked whales was generated from vessel surveys conducted in U.S. waters of the western North Atlantic during the summer of 2021 (Table 1; Garrison and Dias 2023; Palka in prep.). One survey was conducted from 16 June to 23 August in waters north of 36ºN latitude and consisted of 5,871 km of on-effort trackline along the shelf break and offshore to the outer edge of the U.S. EEZ (NEFSC and SEFSC 2022). The second vessel survey covered waters from central Florida (25ºN latitude) to approximately 38ºN latitude between the 200-m isobaths and the outer edge of the U.S. EEZ during 12 June–31 August. A total of 5,659 km of trackline was covered on effort (NEFSC and SEFSC 2022). Both surveys utilized two visual teams and an independent observer approach to estimate detection probability on the trackline (Laake and Borchers 2004). Mark-recapture distance sampling was used to estimate abundance. Estimates from the two surveys were combined and CVs pooled to produce a species abundance estimate for the stock area.

Table 1. Summary of abundance estimates for Mesoplodon beaked whales (2016 surveys) and Gervais' beaked whales (2021 surveys), month, year, and area covered during each abundance survey, and resulting abundance estimate (N_{best}) and coefficient of variation (CV). The estimate considered best is in bold font.

Month/Year	Area	N_{best}	CV
Jun–Sep 2016	Central Virginia to lower Bay of Fundy (*Mesoplodon* spp.)	6,760	0.37
Jun–Aug 2016	Central Florida to Virginia (*Mesoplodon* spp.)	3,347	0.29
Jun–Aug 2016	Central Florida to lower Bay of Fundy (COMBINED, *Mesoplodon* spp.)	10,107	0.27
Jun–Aug 2021	New Jersey to lower Bay of Fundy (*M. europaeus* only)	0	0
Jun–Aug 2021	Central Florida to New Jersey (*M. europaeus* only)	8,595	0.24
Jun–Aug 2021	**Central Florida to lower Bay of Fundy (COMBINED, *M. europaeus* only)**	**8,595**	**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 of Gervais' beaked whales is 8,595 (CV=0.24). The minimum population estimate for Gervais' beaked whales in the western North Atlantic is 7,022.

Current Population Trend

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

CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

Current and maximum net productivity rates are unknown for this stock. *Mesoplodon europaeus* life history parameters that could be used to estimate net productivity include: estimated mean length at birth of 2.1 m, length at sexual maturity of up to 5.2 m for females and up to 4.6 m for males, and maximum age of 27 dental growth layer groups (GLG's), which are presumed to each correspond to a single year of growth (Mead 1984).

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

POTENTIAL BIOLOGICAL REMOVAL

Potential Biological Removal (PBR) is the product of minimum population size, one-half the maximum productivity rate, and a recovery factor (MMPA Sec. 3. 16 U.S.C. 1362; Wade and Angliss 1997). The minimum population size for Gervais' beaked whales is 7,022 (Table 2). The maximum productivity rate is 0.04, the default value for cetaceans. The recovery factor, which accounts for endangered, depleted, threatened stocks, or stocks of unknown status relative to optimum sustainable population (OSP) is assumed to be 0.5. PBR for Gervais' beaked whales in the western North Atlantic is 70.

Table 2. Best and minimum abundance estimates for Gervais' beaked whales (*Mesoplodon europaeus*) of the Western North Atlantic with Maximum Productivity Rate (Rmax), Recovery Factor (Fr) and PBR.

N_{est}	CV N_{est}	N_{min}	F_r	R_{max}	PBR
8,595	0.24	7,022	0.5	0.04	70

ANNUAL HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

The 2017– 2021 total average estimated annual mortality of Gervais' beaked whales in observed fisheries in the U.S. Atlantic EEZ is zero. No information is available on average estimated annual mortality of Gervais' beaked whales from fisheries in Canadian waters.

Fishery Information

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

In 2017–2021, estimated annual average fishery-related mortality or serious injury of this stock in U.S. fisheries was 0 for all beaked whales. Detailed fishery information is reported in Appendix III.

Other Mortality

During 2017–2021, 18 Gervais' beaked whales stranded along the U.S. Atlantic coast (Table 3; NOAA National Marine Mammal Health and Stranding Response Database, accessed 15 October 2022).

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

State	2017	2018	2019	2020	2021	Total
North Carolina	2	8	1	4	0	15
South Carolina	0	0	0	1	0	1
Florida[a]	0	0	2	0	0	2
Total	**2**	**8**	**3**	**5**	**0**	**18**

a. Florida stranding in 2019 deemed human interaction due to plastic ingestion.

STATUS OF STOCK

Gervais' beaked whales are not listed as threatened or endangered under the Endangered Species Act and the western North Atlantic stock of Gervais' beaked whale is not considered strategic under the Marine Mammal Protection Act, although there are insufficient data to determine the population size or trends. The permanent closure of the pelagic drift gillnet fishery has eliminated the principal known source of incidental fishery mortality, and no fishery-related mortality and serious injury has been observed during the recent 5-year (20173–202117) period. Therefore, the total U.S. fishery mortality and serious injury rate can be considered to be insignificant and approaching zero. The status of Gervais' beaked whales relative to OSP is unknown.

OTHER FACTORS THAT MAY BE AFFECTING THE STOCK

Habitat Issues

The chronic impacts of contaminants (polychlorinated biphenyls [PCBs] and chlorinated pesticides [DDT, DDE, dieldrin, etc.]) on marine mammal reproduction and health are of concern (e.g., Pierce et al. 2008; Jepson et al. 2016;

Hall et al. 2018; Murphy et al. 2018), but research on contaminant levels for the western north Atlantic beaked whales is lacking.

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

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

Fourteen beaked whales (mostly Cuvier's beaked whales but also including Gervais' and Blainville's beaked whales) stranded in the Canary Islands in 2002 (Cox et al. 2006, Fernandez et al. 2005; Martin et al. 2004). Gas bubble-associated lesions and fat embolism were found in necropsied animals from this event, leading researchers to link nitrogen supersaturation with sonar exposure (Fernandez et al. 2005).

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). Chavez-Rosales et al. (2022) documented an overall 178 km northeastward spatial distribution shift of the seasonal core habitat of Northwest Atlantic cetaceans that was related to changing habitat/climatic factors. Results varied by season and species. This study used sightings data collected during seasonal aerial and shipboard line transect abundance surveys during 2010 to 2017. There is uncertainty in how, if at all, the changes in distribution and population size of cetacean species may interact with changes in distribution of prey species and how the ecological shifts will affect human impacts to the species.

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

STOCK DEFINITION AND GEOGRAPHIC RANGE

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

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

Sowerby's beaked whales represent a transboundary stock reported to occur from New England waters north to the ice pack (e.g., Davis Strait), and individuals are seen along the Newfoundland coast in summer (Leatherwood et al. 1976; Mead 1989; MacLeod et al. 2006; Jefferson et al. 2008) (Figure 1). Furthermore, a single stranding occurred off the Florida west coast (Mead 1989). This species is considered rare in Canadian waters (Lien et al. 1990) and has been designated as "Special Concern" by the Committee on the Status of Endangered Wildlife in Canada (COSEWIC). Whitehead (2013) reports that in the 23 years of cetacean observations in the Gully Marine Protected Area, on the edge of the Scotian Shelf, Nova Scotia, Canada, they have observed a significant increase in sightings of Sowerby's.

POPULATION SIZE

The best abundance estimate for Sowerby's beaked whales is the sum of the 2021 survey estimates–492 (CV=0.50). This estimate, derived from shipboard surveys, covers most of this stock's known range. In the 2021 survey, improvements to field protocols for both visual observers and passive acoustic monitoring of *Mesoplodon* spp. facilitated differentiation of species during encounters. This enabled abundance estimates to be calculated for each species individually rather than grouping together at the genus level. .

Recent Surveys and Abundance Estimates

Abundance estimates of 6,760 (CV=0.37) and 3,347 (CV=0.29) *Mesoplodon* spp. beaked whales (not including *Ziphius*) were generated from vessel surveys conducted in U.S. waters of the western North Atlantic during the summer of 2016 (Table 1; Garrison 2020; Palka 2020). One survey was conducted from 27 June to 25 August in waters north of 38ºN latitude and consisted of 5,354 km of on-effort trackline along the shelf break and offshore to the outer limit of the U.S. EEZ (NEFSC and SEFSC 2018). The second vessel survey covered waters from Central Florida to approximately 38ºN latitude between the 100-m isobath and the outer limit of the U.S. EEZ during 30 June–19 August. A total of 4,399 km of trackline was covered on effort (NEFSC and SEFSC 2018). Both surveys utilized two visual teams and an independent observer approach to estimate detection probability on the trackline (Laake and Borchers 2004). Mark-recapture distance sampling was used to estimate abundance. Estimates from the two surveys were combined and CVs pooled to produce an abundance estimate for the stock area, yielding a combined total of 10,107 *Mesoplodon* beaked whales (CV=0.27). These estimates are known to be biased low due to the fact that unidentified Ziphiidae abundance was estimated at 3,755 (CV=0.42) in the NE and at 2,812 (CV=0.43) in the SE. These estimates likely include an unknown number of *Mesoplodon* beaked whales.

A more recent abundance estimate of 492 (CV=0.50) Sowerby's beaked whales was generated from vessel surveys conducted in U.S. waters of the western North Atlantic during the summer of 2021 (Table 1; Garrison and Dias 2023; Palka 2023). One survey was conducted from 16 June to 23 August in waters north of 36ºN latitude and consisted of 5,871 km of on-effort trackline along the shelf break and offshore to the outer edge of the U.S. EEZ (NEFSC and SEFSC 2022). The second vessel survey covered waters from central Florida (25ºN latitude) to approximately 38ºN latitude between the 200-m isobaths and the outer edge of the U.S. EEZ during 12 June–31 August. A total of 5,659 km of trackline was covered on effort (NEFSC and SEFSC 2022). Both surveys utilized two visual teams and an independent observer approach to estimate detection probability on the trackline (Laake and Borchers 2004). Mark-recapture distance sampling was used to estimate abundance. Estimates from the two surveys were combined and CVs pooled to produce a species abundance estimate for the stock area.

Table 1. Summary of abundance estimates for the Mesoplodon beaked whales (2016 surveys) and Sowerby's beaked whales (2021 surveys), month, year, and area covered during each abundance survey, and resulting abundance estimate (N_{best}) and coefficient of variation (CV).

Month/Year	Area	N_{best}	CV
Jun–Sep 2016	Central Virginia to lower Bay of Fundy (*Mesoplodon* spp.)	6,760	0.37
Jun–Aug 2016	Central Florida to Central Virginia (*Mesoplodon* spp.)	3,347	0.29
Jun–Sep 2016	Central Florida to lower Bay of Fundy (COMBINED, (*Mesoplodon* spp.)	10,107	0.27
Jun–Aug 2021	New Jersey to lower Bay of Fundy (*M. bidens* only)	492	0.50
Jun–Aug 2021	Central Florida to New Jersey (*M. bidens* only)	0	0
Jun–Aug 2021	**Central Florida to lower Bay of Fundy (COMBINED, *M. bidens* only)**	**492**	**0.50**

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 Sowerby's beaked whales is 492 (CV=0.50). The minimum population estimate for Sowerby's beaked whales in the western North Atlantic is 340.

Current Population Trend

There are insufficient data to determine population trends for this stock. The statistical power to detect a trend in abundance for this stock is poor due to the relatively imprecise abundance estimates and long survey interval. For example, the power to detect a precipitous decline in abundance (i.e., 50% decrease in 15 years) with estimates of low precision (e.g., CV > 0.30) remains below 80% (alpha = 0.30) unless surveys are conducted on an annual basis (Taylor et al. 2007). There is current work to standardize the strata-specific previous abundance estimates to consistently represent the same regions and include appropriate corrections for perception and availability bias. These standardized abundance estimates will be used in state-space trend models that incorporate environmental factors that could potentially influence the process and observational errors for each stratum.

CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

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

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

POTENTIAL BIOLOGICAL REMOVAL

Potential Biological Removal (PBR) is the product of minimum population size, one-half the maximum productivity rate, and a recovery factor (MMPA Sec. 3. 16 U.S.C. 1362; Wade and Angliss 1997). The minimum population size for Sowerby's beaked whales is 492. The maximum productivity rate is 0.04, the default value for cetaceans. The recovery factor, which accounts for endangered, depleted, threatened stocks, or stocks of unknown status relative to optimum sustainable population (OSP) is assumed to be 0.5. PBR for Sowerby's beaked whales in the western North Atlantic is 3.4.

Table 2. Best and minimum abundance estimates for Sowerby's beaked whales (*Mesoplodon bidens*) of the Western North Atlantic with Maximum Productivity Rate (Rmax), Recovery Factor (Fr) and PBR.

N_{est}	CV N_{est}	N_{min}	F_r	R_{max}	PBR
492	0.50	340	0.5	0.04	3.4

ANNUAL HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

The 2017–2021 total average estimated annual mortality of Sowerby's beaked whales in observed fisheries in the U.S. Atlantic EEZ is zero.

Fishery Information

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

In 2017–2021, estimated annual average fishery-related mortality or serious injury of this stock in U.S. fisheries was 0 for all beaked whales. Detailed fishery information is reported in Appendix III.

Earlier Interactions

See Appendix V for more information on historical takes.

Other Mortality

During 2017–2021 1 Sowerby's beaked whale stranded along the U.S. Atlantic coast without evidence of human interaction (Table 3; NOAA National Marine Mammal Health and Stranding Response Database, accessed 15 October 2022).

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

State	2017	2018	2019	2020	2021	Total
Maine	0	0	0	1	0	1
Massachusetts	0	0	0	0	0	0
Total	**0**	**0**	**0**	**0**	**0**	**1**

STATUS OF STOCK

While Sowerby's beaked whales are not listed as threatened or endangered under the Endangered Species Act, they have been listed as a species of Special Concern by both COSEWIC and SARA (the Species at Risk Act) in Canada due to concerns about potential effects on this species from widespread seismic operations as well as occasional military sonar use off of Canada's East Coast (COSEWIC 2006). The western North Atlantic stock of Sowerby's beaked whale is not considered strategic under the Marine Mammal Protection Act. There are insufficient data to determine the population size or trends. The permanent closure of the pelagic drift gillnet fishery has eliminated the principal known source of incidental fishery mortality, and no fishery-related mortality and serious injury has been observed during the recent 5-year (2013–2017) period. Therefore, the total U.S. fishery mortality and serious injury rate can be considered to be insignificant and approaching zero. The status of Sowerby's beaked whales relative to OSP is unknown.

OTHER FACTORS THAT MAY BE AFFECTING THE STOCK

Habitat Issues

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

Anthropogenic sound in the world's oceans has been shown to affect marine mammals, with vessel traffic, seismic surveys, and active naval sonars being the main anthropogenic contributors to low- and mid-frequency noise in oceanic waters (e.g., Nowacek et al. 2015; Gomez et al. 2016; NMFS 2018). Questions have been raised regarding potential effects of human-made sounds on deep-diving cetacean species such as Sowerby's beaked whales (Richardson et al. 1995). 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.

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

Fourteen beaked whales (mostly Cuvier's beaked whales but also including Gervais' and Blainville's beaked whales) stranded in the Canary Islands in 2002 (Cox et al. 2006, Fernandez et al. 2005; Martin et al. 2004). Gas bubble-associated lesions and fat embolism were found in necropsied animals from this event, leading researchers to link nitrogen supersaturation with sonar exposure (Fernandez et al. 2005).

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). Chavez-Rosales et al. (2022) documented an overall 178 km northeastward spatial distribution shift of the seasonal core habitat of Northwest Atlantic cetaceans that was related to changing habitat/climatic factors. Results varied by season and species. This study used sightings data collected during seasonal aerial and shipboard line transect abundance surveys during 2010 to 2017. There is uncertainty in how, if at all, the changes in distribution and population size of cetacean species may interact with changes in distribution of prey species and how the ecological shifts will affect human impacts to the species.

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

STOCK DEFINITION AND GEOGRAPHIC RANGE

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

The distributions of *Mesoplodon* spp. in the Northwest Atlantic are known principally from stranding records (Mead 1989; Nawojchik 1994; Mignucci-Giannoni et al. 1999; MacLeod et al. 2006; Jefferson et al. 2008). Off the U.S. Atlantic coast, beaked whale (*Mesoplodon* spp.) sightings have occurred principally along the shelf-edge and in deeper oceanic waters (Figure 1; CETAP 1982; Waring et al. 1992, 2001; Tove 1995; Hamazaki 2002; Palka 2006; NEFSC and SEFSC 2018, NEFSC and SEFSC 2022). Most sightings occurred in late spring and summer, corresponding to survey effort.

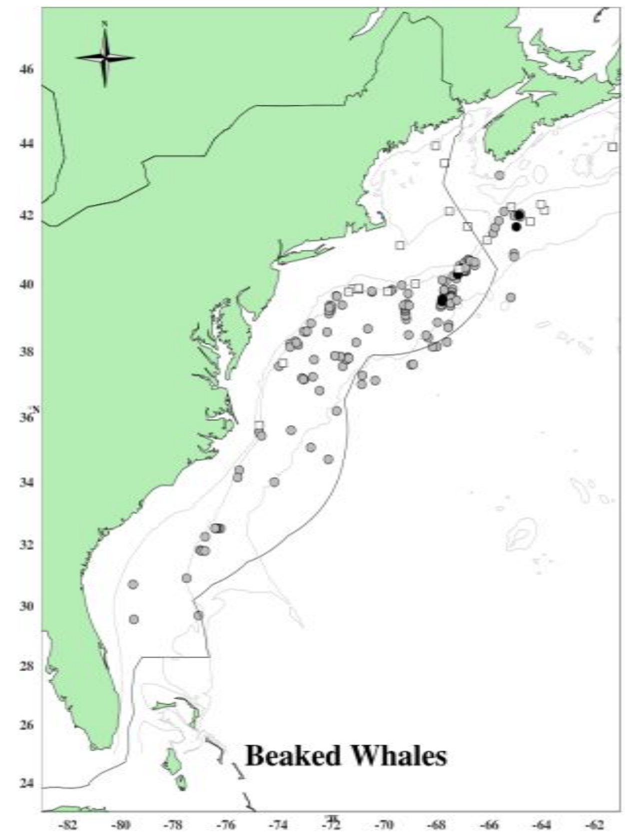

Figure 1: Distribution of beaked whale (includes* Ziphius *and* Mesoplodon *spp.) sightings from NEFSC and SEFSC shipboard and aerial surveys during the summers of 1995, 1998, 1999, 2002, 2004, 2006, 2007, 2008, 2010, 2011, 2016, and 2021 and DFO's 2007 TNASS and 2016 NAISS surveys. Isobaths are the 200-m, 1000-m and 4000-m depth contours. Circle symbols represent shipboard sightings and squares are aerial sightings. Black symbols are the sightings identified as True's beaked whales.

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

POPULATION SIZE

The best abundance estimate for True's beaked whales is the sum of the 2021 survey estimates—4,480 (CV=0.34). This estimate, derived from shipboard surveys, covers most of this stock's known range. In the 2021 survey, improvements to field protocols for both visual observers and passive acoustic monitoring of *Mesoplodon* spp. facilitated differentiation of species during encounters. This enabled abundance estimates to be calculated for each species individually rather than grouping together at the genus level.

Recent Surveys and Abundance Estimates

Abundance estimates of 6,760 (CV=0.37) and 3,347 (CV=0.29) *Mesoplodon* spp. beaked whales (not including *Ziphius*) were generated from vessel surveys conducted in U.S. waters of the western North Atlantic during the summer of 2016 (Table 1; Garrison 2020; Palka 2020). One survey was conducted from 27 June to 25 August in waters north of 38ºN latitude and consisted of 5,354 km of on-effort trackline along the shelf break and offshore to the outer limit of the U.S. EEZ (NEFSC and SEFSC 2018). The second vessel survey covered waters from Central Florida to approximately 38ºN latitude between the 100-m isobath and the outer limit of the U.S. EEZ during 30 June–

19 August. A total of 4,399 km of trackline was covered on effort (NEFSC and SEFSC 2018). Both surveys utilized two visual teams and an independent observer approach to estimate detection probability on the trackline (Laake and Borchers 2004). Mark-recapture distance sampling was used to estimate abundance. Estimates from the two surveys were combined and CVs pooled to produce an abundance estimate for the stock area, yielding a combined total of 10,107 *Mesoplodon* beaked whales (CV=0.27). These estimates are known to be biased low due to the fact that unidentified Ziphiidae abundance was estimated at 3,755 (CV=0.42) in the NE and at 2,812 (CV=0.43) in the SE, and these numbers likely include an unknown number of *Mesoplodon* beaked whales.

A more recent abundance estimate of 4,480 (CV=0.34) True's beaked whales was generated from vessel surveys conducted in U.S. waters of the western North Atlantic during the summer of 2021 (Table 1; Garrison and Dias 2023; Palka 2023). One survey was conducted from 16 June to 23 August in waters north of 36ºN latitude and consisted of 5,871 km of on-effort trackline along the shelf break and offshore to the outer edge of the U.S. EEZ (NEFSC and SEFSC 2022). The second vessel survey covered waters from central Florida (25ºN latitude) to approximately 38ºN latitude between the 200-m isobaths and the outer edge of the U.S. EEZ during 12 June–31 August. A total of 5,659 km of trackline was covered on effort (NEFSC and SEFSC 2022). Both surveys utilized two visual teams and an independent observer approach to estimate detection probability on the trackline (Laake and Borchers 2004). Mark-recapture distance sampling was used to estimate abundance. Estimates from the two surveys were combined and CVs pooled to produce a species abundance estimate for the stock area.

Table 1. Summary of abundance estimates for Mesoplodon spp. (2016 surveys) and True's beaked whales (2021 surveys), month, year, area covered during each abundance survey, and resulting abundance estimate (N_{best}) and coefficient of variation (CV). The estimate considered best is in bold font.

Month/Year	Area	N_{best}	CV
Jun-Sep 2016	Central Virginia to lower Bay of Fundy (*Mesoplodon* spp.)	6,760	0.37
Jun-Aug 2016	Central Florida to Virginia (*Mesoplodon* spp.)	3,347	0.29
Jun-Aug 2016	Central Florida to lower Bay of Fundy (COMBINED, *Mesoplodon* spp.)	10,107	0.27
Jun–Aug 2021	New Jersey to lower Bay of Fundy (*M. mirus*)	4,480	0.34
Jun–Aug 2021	Central Florida to New Jersey (*M. mirus*)	0	0
Jun–Aug 2021	**Central Florida to lower Bay of Fundy (COMBINED, *M. mirus*)**	**4,480**	**0.34**

Minimum Population Estimate

The minimum population estimate is the lower limit of the two-tailed 60% confidence interval of the log-normally distributed best abundance estimate. 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 True's beaked whales is 4,480 (CV= 0.34). The minimum population estimate for True's beaked whales in the western North Atlantic is 3,391.

Current Population Trend

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

CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

Current and maximum net productivity rates are unknown for this stock. Life history parameters that could be used to estimate net productivity include: length at birth is 2 to 3 m, length at sexual maturity 6.1 m for females, and 5.5 m for males, maximum age for females were 30 expressed in dental growth layer groups (GLG's) which may each correspond to a single year of growth is 30 for and for females and 36 for males (Mead 1984).

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

POTENTIAL BIOLOGICAL REMOVAL

Potential Biological Removal (PBR) is the product of minimum population size, one-half the maximum productivity rate, and a recovery factor (MMPA Sec. 3. 16 U.S.C. 1362; Wade and Angliss 1997). The minimum population size for True's beaked whales is 3,391. The maximum productivity rate is 0.04, the default value for cetaceans. The recovery factor, which accounts for endangered, depleted, threatened stocks, or stocks of unknown status relative to optimum sustainable population (OSP) is assumed to be 0.5. PBR for True's beaked whales in the western North Atlantic is 34.

Table 2. Best and minimum abundance estimates for True's beaked whales (**Mesoplodon mirus*****) of the Western North Atlantic with Maximum Productivity Rate (R_{max}), Recovery Factor (F_r) and PBR.***

N_{est}	CV N_{est}	N_{min}	F_r	R_{max}	PBR
4,480	0.34	3,391	0.5	0.04	34

ANNUAL HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

The 2017–2021 total average estimated annual mortality of True's beaked whales in observed fisheries in the U.S. Atlantic EEZ is zero.

Fishery Information

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

In 2017–2021, estimated annual average fishery-related mortality or serious injury of this stock in U.S. fisheries was 0 for all beaked whales. Detailed fishery information is reported in Appendix III.

STATUS OF STOCK

True's beaked whales are not listed as threatened or endangered under the Endangered Species Act, and the western North Atlantic stock is not considered strategic under the Marine Mammal Protection Act. There are insufficient data to determine the population size or trends. The permanent closure of the pelagic drift gillnet fishery has eliminated the principal known source of incidental fishery mortality. Therefore, total U.S. fishery-related mortality and serious injury rate can be considered to be insignificant and approaching zero. The status of True's beaked whales relative to OSP is unknown.

OTHER FACTORS THAT MAY BE AFFECTING THE STOCK

Strandings

During 2017–2021, 1 True's beaked whale stranded along the U.S. Atlantic coast without evidence of human interaction (Table 3; NOAA National Marine Mammal Health and Stranding Response Database, accessed 15 October 2022).

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

Fourteen beaked whales (mostly Cuvier's beaked whales but also including Gervais' and Blainville's beaked whales) stranded in the Canary Islands in 2002 (Martin et al. 2004; Fernandez et al. 2005; Cox et al. 2006). Gas bubble-associated lesions and fat embolism were found in necropsied animals from this event, leading researchers to link nitrogen supersaturation with sonar exposure (Fernandez et al. 2005).

Table 3. True's beaked whale (**Mesoplodon mirus*****) strandings along the U.S. Atlantic coast.***

State	2017	2018	2019	2020	2021	Total
Virginia	1	0	0	0	0	1
Total	**1**	**0**	**0**	**0**	**0**	**1**

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 beaked whales is lacking.

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

Climate-related changes in spatial distribution and abundance, including poleward and depth shifts, have been documented in or predicted for plankton species and commercially important fish stocks (Nye et al. 2009; 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). Chavez-Rosales et al. (2022) documented an overall 178 km northeastward spatial distribution shift of the seasonal core habitat of Northwest Atlantic cetaceans that was related to changing habitat/climatic factors. Results varied by season and species. This study used sightings data collected during seasonal aerial and shipboard line transect abundance surveys during 2010 to 2017. There is uncertainty in how, if at all, the changes in distribution and population size of cetacean species may interact with change sin distribution of prey species and how the ecological shifts will affect human impacts to the species.

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

STOCK DEFINITION AND GEOGRAPHIC RANGE

The melon-headed whale is distributed worldwide in tropical and subtropical waters (Jefferson et al. 1994). However, sightings of this species in the western North Atlantic are extremely rare. Most stranding records are from Florida and South Carolina, with a few from Virginia and one from New Jersey. There have been two sightings during NMFS vessel surveys between 1992 and 2016. Melon-headed whales in the western North Atlantic are managed separately from those in the northern Gulf of Mexico. Although there have been no directed studies of the degree of demographic independence between the two areas, this management structure is consistent with evidence for strong population structuring in other areas (Martien et al. 2014) and is further supported because the two stocks occupy distinct marine ecoregions (Spalding et al. 2007; Moore and Merrick 2011). Due to the paucity of sightings in the western North Atlantic, there are insufficient data to determine whether the western North Atlantic stock comprises multiple demographically independent populations. Additional morphological, acoustic, genetic, and/or behavioral data are needed to further delineate population structure within the western North Atlantic and across the broader geographic area. Because there are confirmed sightings within waters of the Bahamas, this is likely a transboundary stock (e.g., Halpin et al. 2009; Dunn 2013).

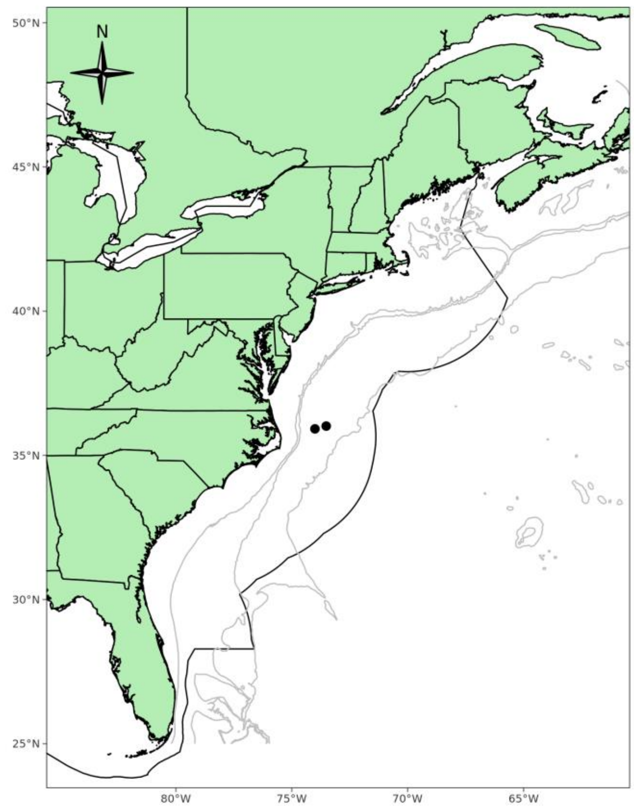

Figure 1. Distribution of melon-headed whale sightings from NEFSC and SEFSC shipboard and aerial surveys during 1995, 1998, 1999, 2002, 2004, 2006, 2007, 2008, 2010, 2011, 2016, and 2021. Isobaths are the 200-m, 1,000-m, and 4,000-m depth contours. The darker line indicates the U.S. EEZ.

POPULATION SIZE

The number of melon-headed whales off the U.S. Atlantic coast is unknown because they were rarely seen in any surveys. A single group of melon-headed whales was sighted off of Cape Hatteras, North Carolina, in waters >2500 m deep during both a summer 1999 (20 whales) and a winter 2002 (80 whales) vessel survey of the western North Atlantic (Figure 1; NMFS 1999; NMFS 2002). Abundances have not been estimated from these single sightings. Therefore the population size of melon-headed whales is unknown. No confirmed sightings of melon-headed whales have been observed in any other NMFS surveys. Several cruises—a winter 2002 cruise (NMFS 2002), a summer 2005 cruise (NMFS 2005), a summer 2016 cruise (NEFSC and SEFSC 2016), and a summer 2021 cruise (NEFSC and SEFSC 2022)—each had one or two sightings of pygmy killer or melon-headed whales (identity was not confirmed), and these groups were recorded off Cape Hatteras or off the North Carolina/South Carolina border. However, there have been at least two additional sightings of melon-headed whales off Cape Hatteras (Halpin et al. 2009; McLellan 2014).

Minimum Population Estimate

Present data are insufficient to calculate a minimum population estimate for this stock (Table 1).

Current Population Trend

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

CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

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

POTENTIAL BIOLOGICAL REMOVAL

Potential Biological Removal level (PBR) is the product of the minimum population size, one-half the maximum productivity rate, and a recovery factor (MMPA Sec. 3.16 U.S.C. 1362; Wade and Angliss 1997). The minimum population size is unknown. The maximum productivity rate is 0.04, the default value for cetaceans. The "recovery" factor, which accounts for endangered, depleted, threatened stocks, or stocks of unknown status relative to optimum sustainable population (OSP), is assumed to be 0.5 because this stock is of unknown status. PBR for the western North Atlantic stock of melon-headed whales is unknown because the minimum population size is unknown (Table 1).

Table 1. Best and minimum abundance estimates for the western North Atlantic melon-headed whale (*Peponocephala electra*) with Maximum Productivity Rate (R_{max}), Recovery Factor (F_r) and PBR.

N_{est}	CV N_{est}	N_{min}	F_r	R_{max}	PBR
Unknown	-	Unknown	0.5	0.04	Unknown

ANNUAL HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

Total annual estimated human-caused mortality and serious injury to this stock during 2017–2021 was presumed to be zero, as there were no reports of mortalities or serious injuries to melon-headed whales in the western North Atlantic. This species is rare and as a result the likelihood of observing a take is very low. Survey effort and observer effort are insufficient to effectively estimate takes for this species.

Fishery Information

There is one commercial fishery that could potentially interact with this stock in the Atlantic Ocean, the Category I Atlantic Ocean, Caribbean, Gulf of Mexico large pelagics longline fishery (Appendix III). Pelagic swordfish, tunas and billfish are the target species of the longline fishery. Percent observer coverage (percentage of sets observed) for this fishery in the Atlantic for each year during 2017–2021 was 11, 10, 10, 9, and 8, respectively. There were no observed mortalities or serious injuries to melon-headed whales by this fishery in the Atlantic Ocean during 2017–2021 (Garrison and Stokes 2020a; 2020b; 2021; 2023a; 2023b).

STATUS OF STOCK

Melon-headed whales are not listed as threatened or endangered under the Endangered Species Act, and the Western North Atlantic stock is not considered strategic under the Marine Mammal Protection Act. No fishery-related mortality or serious injury has been observed during recent years; however, because this stock is rare, it is unknown whether total fishery-related mortality and serious injury can be considered insignificant and approaching the zero mortality and serious injury rate. The status of melon-headed whales in the western U.S. Atlantic EEZ relative to optimum sustainable population is unknown. There are insufficient data to determine the population trends for this species.

OTHER FACTORS THAT MAY BE AFFECTING THE STOCK

Strandings

During 2017–2021, three melon-headed whales were reported stranded along the U.S. East Coast (NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 13 October 2022 (Southeast Region) and 18 September 2022 (Northeast Region)). Two strandings occurred in Florida (one in 2018, one in 2020), and the remaining stranding occurred in South Carolina (in 2020). No evidence of human interaction was detected for one stranding, and for the remaining two strandings, it could not be determined if there was evidence of human interaction.

Stranding data underestimate the extent of human and fishery-related mortality and serious injury because not all of the marine mammals that die or are seriously injured in human interactions wash ashore, or, if they do, they are not all recovered (Peltier et al. 2012; Wells et al. 2015; 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

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.

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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 represent a transboundary stock which occurs 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 concluded that Risso's dolphins were 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, 2022).

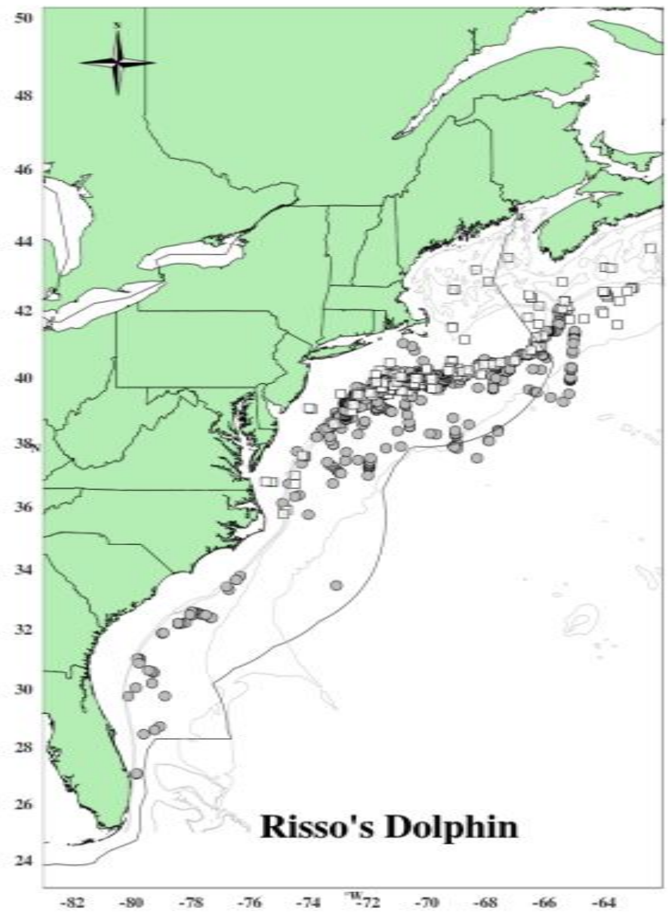

Figure 1. Distribution of Risso's dolphin sightings from NEFSC and SEFSC shipboard and aerial surveys during the summers of 1995, 1998, 1999, 2002, 2004, 2006, 2007, 2008, 2010, 2011, 2016 and 2021 and Department of Fisheries and Oceans Canada 2007 TNASS and 2016 NAISS surveys. Isobaths are the 100-m, 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 2021 NEFSC and SEFSC surveys—44,067 (CV=0.45; 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. While some Canadian waters were covered in the aerial portion of the 2021 survey, the full Canadian portion of the species' range was not as well represented in the 2021 survey compared to the 2016 survey. Nevertheless, the 2021 estimate is considered best for this stock.

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 was 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.

More recent abundance estimates of 39,612 (CV=0.50) and 4,455 (CV=0.45) Risso's dolphins were generated from vessel surveys conducted in U.S. waters of the western North Atlantic during the summer of 2021 (Table 1; Garrison and Dias 2023; Palka 2023). One survey was conducted from 16 June to 23 August in waters north of 36ºN latitude and consisted of 5,871 km of on-effort trackline along the shelf break and offshore to the outer edge of the U.S. EEZ (NEFSC and SEFSC 2022). The second vessel survey covered waters from central Florida (25ºN latitude) to approximately 38ºN latitude between the 200-m isobaths and the outer edge of the U.S. EEZ during 12 June–31 August. A total of 5,659 km of trackline was covered on effort (NEFSC and SEFSC 2022). 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 (Nest) and coefficient of variation (CV). The estimate considered best is in bold font.

Month/Year	Area	N_{est}	
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
Jun–Aug 2021	New Jersey to lower Bay of Fundy	39,612	0.50
Jun–Aug 2021	Central Florida to New Jersey	4,455	0.45
Jun–Aug 2021	**Central Florida to lower Bay of Fundy (COMBINED)**	**44,067**	**0.45**

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 44,067 (CV=0.45), obtained from the 2021 surveys. The minimum population estimate for the western North Atlantic Risso's dolphin is 30,662. This estimate covers U.S. waters and a portion of the range in Canadian waters.

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 30,662. 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 368 (Table 2). As noted above, the surveys upon which this estimate and PBR are based did not cover all of the species' range in Canadian waters, however, the estimate is still considered the best estimate for the entire stock.

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
44,067	0.45	30,662	0.5	0.04	307

ANNUAL HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

Total annual estimated average human-caused mortality or serious injury to this stock during 2017–2021 was 18 Risso's dolphins, derived from estimated mortalities and serious injuries in observed U.S. 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
2017–2021	U.S. fisheries using observer data	18	0.09
2017–2021	Non-fishery human caused stranding mortalities	0	-
TOTAL		18	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 2017–2021 are documented in Garrison and Stokes (2019, 2020a, 2020b, 2021, 2023a, 2023b). 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 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 (Orphanides 2020, 2021; Precoda and Orphanides 2022; Precoda 2023). 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

One Risso's dolphin wasobserved taken in northeast bottom trawl fisheries in 2021(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.

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	2017 2018 2019 2020 2021	Obs. Data, Logbook	0.12 0.10 0.10 0.09 0.08	1 1 0 2 0	0 0 0 0 0	0.2 0.2 0 12.2 0	0 0 0 0 0	0.2 0.2 0 12.2 0	1 0.94 0 0.71 0	2.5 (0.68)
Northeast Sink Gillnet	2017 2018 2019 2020 2021	Obs. Data, Trip Logbook, Allocated Dealer Data	0.12 0.11 0.12 0.02 0.11	0 0 0 0 0	0 0 1 0 3	0 0 0 0 0	0 0 5 2 3	0 0 5 2 3	0 0 0.7 1.01 0	2 (1.81)
Northeast Bottom Trawl	2017 2018 2019 2020 2021	Obs. Data, Weighout	0.12 0.12 0.16 0.08 0.19	0 0 0 0 0	0 0 0 0 1	0 0 0 0 0	0 0 0 0 3.8	0 0 0 0 3.8	0 0 0 0 0.88	0.75 (0.88)
Mid-Atlantic Bottom Trawl	2017 2018 2019 2020 2021	Obs. Data, Dealer Data	0.14 0.12 0.12 0.02 0.04	2 0 0 0 0	5 0 0 2 0	12 0 0 4 0	31 0 0 14 0	43 0 0 18 0	0.51 0 0 0.51 0	12 (0.39)
TOTAL										18 (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 (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 2017–2021 period and include both at-sea monitor and traditional observer data (Josephson and Lyssikatos 2023).

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 2017–2021 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 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.

OTHER FACTORS THAT MAY BE CAUSING A DECLINE OR IMPEDING RECOVERY

Strandings

From 2017 to 2021, 2931 Risso's dolphin strandings were recorded along the U.S. Atlantic coast (NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 197 OctoberNovember 20220). None of the animals had indications of human interaction.

Table 5. Risso's dolphin (**Grampus griseus*****) reported strandings along the U.S. Atlantic coast and Puerto Rico, 2017–2021.***

STATE	2017	2018	2019	2020	2021	TOTALS
Massachusetts	141	02	014	00	00	1714

STATE	2017	2018	2019	2020	2021	TOTALS
Rhode Island	10	0	01	00	01	12
New York[a]	02	0	30	00	30	53
New Jersey	10	0	01	00	01	12
Maryland[b]	00	0	10	00	10	11
Virginia	0	0	0	0	1	1
North Carolina	10	10	1	12	10	35
Florida	10	02	01	00	00	41
TOTAL	**184**	**14**	**54**	**12**	**53**	**3129**

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.

Chavez-Rosales et al. (2022) documented an overall 178 km northeastward spatial distribution shift of the seasonal core habitat of Northwest Atlantic cetaceans that was related to changing habitat/climatic factors. Results varied by season and species. This study used sighting data collected during seasonal aerial and shipboard line transect abundance surveys during 2010 to 2017. During this time frame, the weighted centroid of Risso's dolphin core habitat moved farthest during spring (232 km towards the northeast) and least during summer (89 km). There is uncertainty in how, if at all, the changes in distribution and population size of cetacean species may interact with changes in distribution of prey species and how the ecological shifts will affect human impacts to the species.

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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 Atlantic waters is complex and requires additional information on seasonal spatial distribution. In the North Atlantic 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.

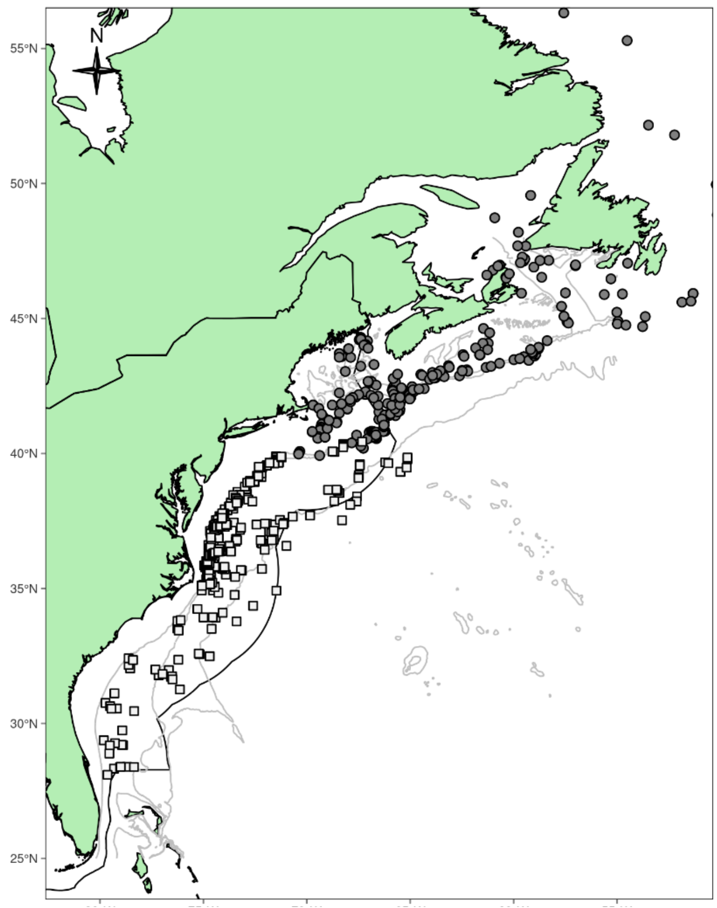

Figure 1. Distribution of long-finned (filled circles) and short-finned (open squares) pilot whale sightings from NEFSC and SEFSC shipboard and aerial surveys during the summers of 1998, 1999, 2002, 2004, 2006, 2007, 2011, 2016, and 2021 and the Department of Fisheries and Oceans Canada 2007 TNASS and 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 Northwest Atlantic population represents a transboundary stock occupying waters in both the U.S. and Canada. In U.S. Atlantic waters, pilot whales (*Globicephala* spp.) are distributed principally along the continental shelf edge off the northeastern U.S. coast in winter and early spring (Figure 1; 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, while 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 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. These survey data have been combined with an analysis of the spatial distribution of the 2 species of pilot whales based on genetic analyses of biopsy samples to derive separate abundance estimates (Garrison and Rosel 2017). Estimates generated from the 2021 surveys are more recent and focus on U.S. waters, although more of the stock range was covered in the 2016 survey.

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.

Recent Surveys and Abundance Estimates for *Globicephala* spp.

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 was flown over the Gulf of St. Lawrence/Bay of Fundy/Scotian Shelf stratum using two Cessna Skymaster 337s and 21,037 km were flown over the Newfoundland/Labrador stratum using a DeHavilland Twin Otter. The Newfoundland estimate was derived from the Twin Otter data using two-team mark-recapture multi-covariate distance sampling methods. 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.

A more recent abundance estimate of 5,734 (CV=0.62) long-finned pilot whales was generated from vessel surveys conducted in U.S. waters of the western North Atlantic during the summer of 2021 (Table 1; Garrison and Dias 2023; Palka 2023). One survey was conducted from 16 June to 23 August in waters north of 36ºN latitude and consisted of 5,871 km of on-effort trackline along the shelf break and offshore to the outer edge of the U.S. EEZ (NEFSC and SEFSC 2022). The second vessel survey covered waters from central Florida (25ºN latitude) to approximately 38ºN latitude between the 200-m isobaths and the outer edge of the U.S. EEZ during 12 June–31 August. A total of 5,659 km of trackline was covered on effort (NEFSC and SEFSC 2022). 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. The 2016 estimate is larger than that from 2021 because the 2016 estimate is derived from a survey area extending from Newfoundland to

Florida, which is about 1,300,000 km2 larger than the 2021 survey area.

Spatial Distribution and Abundance Estimates

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 2021. 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 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). The estimate considered best is in bold font.***

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**
Jun–Aug 2021	New Jersey to lower Bay of Fundy	5,711	0.62
Jun–Aug 2021	Central Florida to New Jersey	0	0
Jun–Aug 2021	Central Florida to lower Bay of Fundy (COMBINED)	5,711	0.62

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).This was based on the combined 2016 surveys,which covered a greater proportion of the stock range than the more recent 2021 survey 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

Total annual estimated average human-caused mortality or serious injury during 2017–2021 was 5.5long-finned pilot whales (CV=0.29 from U.S. fisheries using observer data and an annual average of 0.2 animals from non-fishery stranding records (Table 3). In bottom trawls, mid-water trawls and the gillnet fisheries, mortalities were generally observed north of 40°N latitude and in areas where long-finned pilot whales were expected to occur. Takes in these fisheries were therefore considered to be long-finned pilot whales. Takes in the pelagic longline fishery were partitioned according to a logistic regression model (Garrison and Rosel 2017).

Table 3. Total annual estimated average human-caused mortality and serious injury for the western North Atlantic long-finned pilot whale (*Globicephala melas melas*).

Years	Source	Annual Avg.	CV
2017–2021	U.S. commercial fisheries using observer data	5.5	0.29
2017–2021	Non-fishery human caused stranding mortalities	0.2	-
TOTAL		5.7	0.29

Fishery Information

Detailed fishery information is reported in Appendix III.

Earlier Interactions

See Appendix V for more information on historical takes.

United States

Pelagic Longline

During 2017–2021, 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; Garrison and Stokes 2023a; and 2023b). 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 2017–2021 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

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 4. 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	2017 2018 2019 2020 2021	Obs. Data, Logbook	0.12 0.12 0.16 0.08 0.19	0 0 0 0 0	0 0 1 1 2	0 0 0 0 0	0 0 5.4 1.8 7.5	0 0 5.4 1.8 7.5	na na 0.88 0.88 0.62	2.9 (0.46)
Pelagic Longline Fishery	2017 2018 2019 2020 2021	Obs. Data, Logbook Data	0.12 0.10 0.10 0.09 0.08	1 1 1 1 1	0 0 0 0 0	3.3 0.4 0.4 5.7 2.8	0 0 0 0 0	3.3 0.4 0.4 5.7 2.8	0.98 0.93 1.0 0.44 0.67	2.53(0.36)
TOTAL										5.5 (0.29)

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 and Lyssikatos 2023).

STATUS OF STOCK

The long-finned pilot whale is not listed as threatened or endangered under the Endangered Species Act.he 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 more than 10% of the calculated PBR and, therefore, cannot be considered to be insignificant and approaching zero mortality and serious injury rate. Due to lack of observed fisheries data from Canada, the U.S. fishery-related mortality and serious injury represents a minimum estimate for the stock. The status of this stock relative to OSP 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.

OTHER FACTORS THAT MAY BE CAUSING A DECLINE OR IMPEDING RECOVERY

Strandings

Pilot whales have a propensity to mass strand throughout their range, but the role of human activity in these events is unknown. From 2017 to 2021, 11 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 15 October 2022). Of these, one of the animals had plastic in its stomach, indicating human interaction (Table 5).

Table 5. Pilot whale (**Globicephala melas melas*****) strandings along the Atlantic coast, 2017-2021. 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	2017	2018	2019	2020	2021	Total
Nova Scotia[a]	12	12	12	3	9	29
Newfoundland and Labrador[b]	1	0	1	15	10	28
Maine	1	1	1	0	1	5
Massachusetts[c]	1	1	1	3	1	5
New York	0	0	0	0	1	1
TOTAL	**15**	**6**	**4**	**21**	**22**	**68**

a. Data supplied by Nova Scotia Marine Animal Response Society (pers. comm.).
b. See Ledwell and Huntington 2018, 2019, 2020, 2021a, 2021b.
c. 2021 Massachusetts animal coded as human interaction due to plastic in stomach.

Stranding data probably underestimate the extent of human and fishery-related mortality and serious injury, particularly for offshore species such as pilot whales. 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, because of decomposition and scavenger damage, not all carcasses will show evidence of human interaction, entanglement or other fishery-related interaction . (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 around the Faroe Islands. 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 has been shown to affect marine mammals. Vessel traffic, seismic surveys, and active naval sonars are the main human-caused contributors to low- and mid-frequency noise in oceanic waters (e.g., Nowacek et al. 2015; Gomez et al. 2016; NMFS 2018). Exposure experiments conducted in the northern Norwegian Seas between 2006 and 2009 indicated that long-finned pilot whales conducted fewer deep dives during low-frequency active sonar exposures, while their diving behavior did not change when mid-frequency active sonar exposures were performed (Sivle et al. 2012) The long-term and population consequences of these impacts are less well-documented and likely vary by species and other factors. Impacts from sound on marine mammal prey are also possible (Carroll et al. 2017), but the duration and severity of 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).

Chavez-Rosales et al. (2022) documented an overall 178 km northeastward spatial distribution shift of the seasonal core habitat of Northwest Atlantic cetaceans that was related to changing habitat/climatic factors. This study used sighting data collected during seasonal aerial and shipboard line transect abundance surveys from 2010 to 2017. During this time frame, the weighted centroid of long-finned pilot whale core habitat moved less than 70 km in all seasons. There is uncertainty in how, if at all, the changes in distribution and population size of cetacean species may interact with changes in distribution of prey species and how the ecological shifts will affect human impacts to the species.

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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). Because there are confirmed sightings within waters of the Bahamas, this is likely a transboundary stock (e.g., Halpin et al. 2009; Dunn 2013).

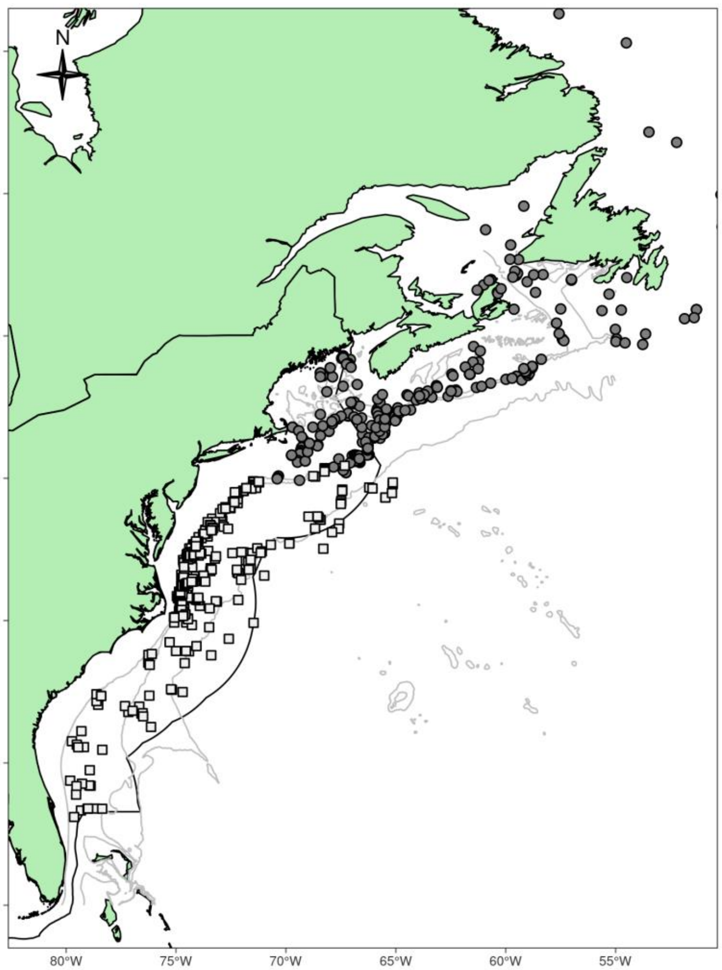

Figure 1. Distribution of long-finned (filled circles) and short-finned (open squares) pilot whale sightings from NEFSC and SEFSC shipboard and aerial surveys during 1995, 1998, 1999, 2002, 2004, 2006, 2007, 2008, 2010, 2011, 2016, and 2021, and DFO's 2007 TNASS and 2016 NAISS surveys. The inferred distribution of the two species is valid for June–August only. Isobaths are the 200-m, 1000-m and 4000-m depth contours.

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 visual tags or satellite-linked telemetry have traveled 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 18,749 (CV=0.33; Table 1; Garrison and Dias 2023; Palka 2023). This estimate is from summer 2021 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* spp.

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 (NEFSC and SEFSC 2016). 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 (NEFSC and SEFSC 2016; 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).

More recent abundance estimates of 3,745 (CV=0.67) and 15,004 (CV=0.38) Globicephala sp. were generated from vessel surveys conducted in U.S. waters of the western North Atlantic during the summer of 2021 (Table 1; Garrison and Dias 2023; Palka 2023). One survey was conducted from 16 June to 23 August in waters north of 36ºN latitude and consisted of 5,871 km of on-effort trackline along the shelf break and offshore to the outer edge of the U.S. EEZ (NEFSC and SEFSC 2022). The second vessel survey covered waters from central Florida (25ºN latitude) to approximately 38ºN latitude between the 200-m isobaths and the outer edge of the U.S. EEZ during 12 June–31 August. A total of 5,659 km of trackline was covered on effort (NEFSC and SEFSC 2022). 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.

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 in 1998, 2001, 2004, 2005, 2006, 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 2021 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 (15,004; CV=0.38) and the estimated number of short-finned pilot whales from the northeast vessel survey (3,722; CV=0.68). The best available abundance estimate is thus 18,726 (CV=0.33).

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. The estimate considered best is in bold font.

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
Jun–Aug 2021	New Jersey to lower Bay of Fundy	3,722	0.67
Jun–Aug 2021	Central Florida to New Jersey	15,004	0.38
Jun–Aug 2021	**Central Florida to lower Bay of Fundy (COMBINED)**	**18,726**	**0.33**

Minimum Population Estimate

The minimum population estimate is the lower limit of the two-tailed 60% confidence interval of the log-normally distributed best abundance estimate. This is equivalent to the 20th percentile of the log-normal distribution as specified by Wade and Angliss (1997). The best estimate of abundance for western North Atlantic short-finned pilot whale is 18,726 animals (CV=0.33). The minimum population estimate is 14,292 (Table 2).

Current Population Trend

There are four available coastwide abundance estimates for short-finned pilot whales from the summers of 2004, 2011, 2016, and 2021. Each of these is derived from vessel surveys with similar survey designs and all four 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, 28,924 (CV=0.24) in 2016, and 18,749 (CV=0.33) in 2021 (Garrison and Palka 2018; Garrison and Dias 2023). A generalized linear model indicated no significant trend (p=0.697) 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 14,292. 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 143 (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}	CV N_{est}	N_{min}	F_r	R_{max}	PBR
18,726	0.33	14,292	0.5	0.04	143

ANNUAL HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

The estimated mean annual fishery-related mortality and serious injury during 2017–2021 due to the large pelagics longline commercial fishery was 218 short-finned pilot whales (CV=0.19; 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 2017 to 2021 was 218 short-finned pilot whales (CV=0.19; Table 3). During 2017–2021, 72 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 2017–2021, one mortality was observed (in 2021) in the Mid-Atlantic Bight fishing area (Garrison and Stokes 2020a; 2020b; 2021; 2023a; 2023b).

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 2020a; 2020b; 2021; 2023a; 2023b).

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 U.S. commercial large pelagics longline 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 mortality and serious injury usingon-board observer data, the annual estimated mortality and serious injury, the combined annual estimates of mortality and serious injury (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]	Observer Coverage[c]	Observed Serious Injury	Observed Mortality	Estimated Serious Injury	Estimated Mortality	Estimated Combined Mortality	Est. CVs	Mean Annual Mortality
Pelagic Longline	2017	65	Obs. Data, Logbook	11	14	0	133	0	133	0.29	218 (0.19)
	2018	57		10	7	0	102	0	102	0.39	
	2019	50		10	10	0	131	0	131	0.37	
	2020	50		9	22	0	371	0	371	0.45	
	2021	49		8	19	1	332	23	355	0.31	

a. Number of vessels in the fishery within the Atlantic 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 in the Atlantic

Hook and Line (Rod and Reel)

During 2017–2021, 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.

STATUS OF STOCK

The short-finned pilot whale is not listed as threatened or endangered under the Endangered Species Act, but the western North Atlantic stock is a strategic stock under the MMPA because the mean annual human-caused mortality and serious injury exceeds PBR. Total U.S. fishery-related mortality and serious injury attributed to short-finned pilot whales exceeds 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 is unknown. There is no evidence for a trend in population size for this stock.

OTHER FACTORS THAT MAY BE AFFECTING THE STOCK

Strandings

During 2017–2021, 65 short-finned pilot whales were reported stranded along the U.S. East Coast between Massachusetts and Florida (Table 4; NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 13 October 2022 (Southeast Region [SER]) and 18 September 2022 (Northeast Region [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 13 strandings, and for the remaining 50 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, 2017–2021. Data are from the NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 13 October 2022 (SER) and 18 September 2022 (NER). EEZ=U.S. Exclusive Economic Zone (offshore U.S. waters).

State	2017	2018	2019	2020	2021	TOTALS
Massachusetts	0	0	3[a]	0	0	3
New York	0	4	0	0	0	4
Maryland	0	0	1	0	0	1
Virginia	0	0	1	2	0	3
North Carolina	1	2	2	0	2	7
South Carolina	0	0	5	0	0	5
Georgia	1	0	40[b]	0	0	41
Florida	0	1	0	0	0	1
TOTALS	**2**	**7**	**52**	**2**	**2**	**65**

a. These 3 animals were a live mass stranding event.

b. There were two mass strandings of short-finned pilot whales in 2019 off Georgia encompassing 39 of the 40 reported strandings. One mass stranding occurred in July, and these 21 animals were part of a mass stranding event of ~50 live whales; the second occurred in September, and these 18 whales were part of a mass stranding event of ~28 live and dead 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). Chavez-Rosales et al. (2022) documented an overall 178 km northeastward spatial distribution shift of the seasonal core habitat of Northwest Atlantic cetaceans that was related to changing habitat/climatic factors. Results varied by season and species. This study used sightings data collected during seasonal aerial and shipboard line transect abundance surveys during 2010 to 2017. During this time frame, the weighted centroid of the short-finned pilot whale core habitat moved towards the northeast in fall and winter (296 and 218 km, respectively) and towards the southwest in spring and summer (120 and 149 km, respectively). There is uncertainty in how, if at all, the changes in distribution and population size of cetacean species may interact with changes in distribution of prey species and how the ecological shifts will affect human impacts to the species.

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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 transboundary and 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, 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 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.

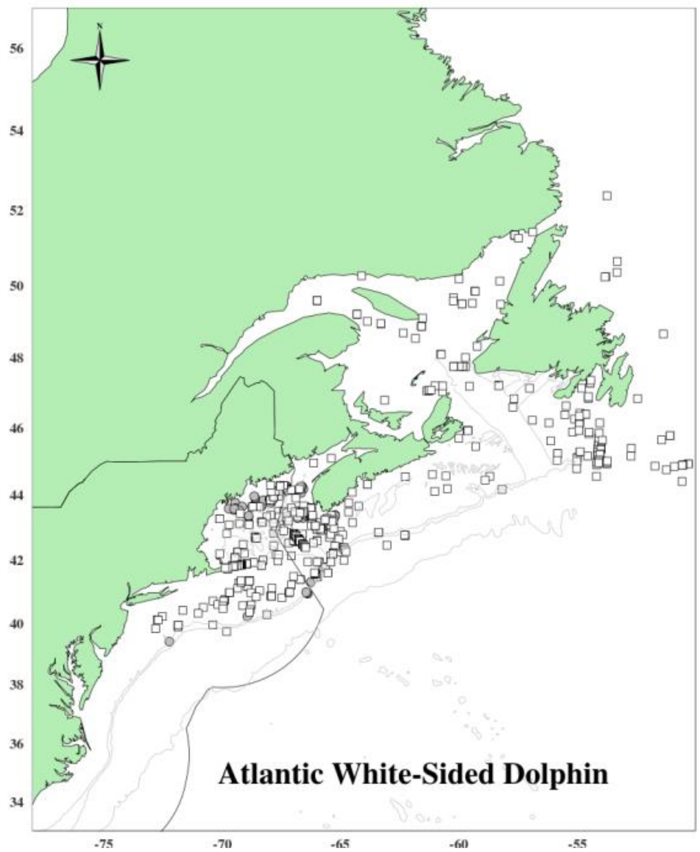

Figure 1. Distribution of white-sided dolphin sightings from NEFSC and SEFSC shipboard and aerial surveys during the summers of 1995, 1998, 1999, 2002, 2004, 2006, 2007, 2008, 2010, 2011, 2016, and 2021 and Department of Fisheries and Oceans Canada 2007 TNASS and 2016 NAISS surveys. Isobaths are the 200-m, 1000-m and 4000-m depth contours.

The white-sided dolphins in US waters are 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).

A genetic analysis of white-sided dolphin samples taken in US waters from Maine to 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 possibility 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.

In summary, the Western North Atlantic stock of white-sided dolphins may contain multiple demographically-independent populations, where the animals in U.S. waters may be part of a Gulf of Maine (sub-)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). This estimate was derived from the June–September 2016 surveys conducted by the U.S. and Canada from Labrador to the U.S. east coast, covering nearly the entire range of the western North Atlantic stock: all of the Gulf of Maine,Gulf of St. Lawrence, and part of the Labrador area 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. Estimates generated from the 2021 surveys are more recent and focus on U.S. waters, although more of the stock range was covered in the 2016 survey.

Earlier Abundance Estimates

Please see Appendix IV for earlier abundance estimates.

Recent Surveys and Abundance Estimates

An abundance estimate of 31,912 (CV=0.61) U.S. 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 visual 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. Estimates generated from the 2021 surveys are more recent and focus on U.S. waters, although more of the stock range was covered in the 2016 survey.

An abundance estimate of 61,321 (CV=1.04) white-sided dolphins from the Canadian side of the Gulf of Maine, and the entire Gulf of St. Lawrence region was generated from an aerial survey conducted by the Department of Fisheries and Oceans, Canada (DFO, Table 1). No white-sided dolphins in the Labrador region were detected on the east side of Labrador. 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.

A more recent abundance estimate of 4,632 (CV=0.55) white-sided dolphins was generated from vessel surveys conducted in U.S. waters of the western North Atlantic during the summer of 2021 (Table 1; Garrison and Dias 2023; Palka 2023). One survey was conducted from 16 June to 23 August in waters north of 36ºN latitude and consisted of 5,871 km of on-effort trackline along the shelf break and offshore to the outer edge of the U.S. EEZ (NEFSC and SEFSC 2022). The second vessel survey covered waters from central Florida (25ºN latitude) to approximately 38ºN latitude between the 200-m isobaths and the outer edge of the U.S. EEZ during 12 June–31 August. A total of 5,659 km of trackline was covered on effort (NEFSC and SEFSC 2022). 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 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). The estimate considered best is in bold font.***

Month/Year	Area	N_{est}	CV
Jun–Sep 2016	Central Virginia to Maine	31,912	0.61
Aug–Sep 2016	Bay of Fundy to Gulf of St. Lawrence	61,321	1.04
Aug–Sep 2016	Newfoundland and Labrador	0	-
Jun–Sep 2016	**Central Virginia to Labrador (COMBINED)**	**93,233**	**0.71**
Jun–Aug 2021	New Jersey to lower Bay of Fundy	4,632	0.55
Jun–Aug 2021	Central Florida to New Jersey	0	-
Jun–Aug 2021	Central Florida to lower Bay of Fundy (COMBINED)	4,632	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 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:1) a calving interval of 2–3 years; 2) a lactation period of 18 months; 3) a gestation period of 10–12 months with births occurring from May to early August, mainly in June and July; 4) observed lengths- at birth of 110 cm; at sexual maturity of 230–240 cm for males, and 201–222 cm for females; 5) age at sexual maturity of 8–9 years for males and 6–8 years for females; 6) mean adult length of 250 cm for males and 224 cm for females (Evans 1987); and 7) a maximum reported age for males of 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 dolphins 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 the U.S portion of this stock during 2017–2021 was 28 (CV = 0.19) white-sided dolphins from fisheries observer data and 0.2from 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 Est. Avg.	CV
2017–2021	U.S. fisheries using observer data	28	0.19
2017–2021	Possible non-fishery human-caused stranding mortalities	0.2	
TOTAL		28.2	0.19

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 especially during years of COVID restrictions. The effect of this is unknown.

There are no major known sources of unquantifiable human-caused mortality or serious injury in U.S. waters. Fishery bycatch 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 Sink Gillnet

In the northeast sink gillnet fishery, white-sided dolphin interactions have historically been rare, but in 2021 two animals were bycaught in this fishery (Orphanides 2020, 2021; Precoda and Orphanides 2022; Precoda 2023). 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

Fishery-related bycatch rates for the Northeast Bottom Trawl fishery 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	2017	Obs. Data, Trip Logbook	0.12	1	1	7.4	7.4	14.8	0.64	28 (0.19)
	2018		0.12	0	0	0	0	0	na	
	2019		0.16	0	14	0	79	79	0.28	
	2020		0.08	0	5	0	31	31	0.26	
	2021		0.19	1	2	5.1	10	15	0.52	
Northeast Gillnet	2017	Observer Data, Weighout	0.12	0	0	0	0	0	0	0.2 (na)
	2018		0.11	0	0	0	0	0	0	
	2019		0.12	0	0	0	0	0	0	
	2020		0.02	0	0	0	0	0	0	
	2021		0.11	0	2	0	2	2	0	
TOTAL										28 (0.19)

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 2017–2021 period and include both at-sea monitor and traditional observer data (Josephson and Lyssikatos 2023).

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 reported by 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.

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 because the estimated average annual human-related mortality does not exceed PBR. Total fishery-related mortality and serious injury for white-sided dolphins is less than 10% of the calculated PBR and, therefore, it is considered to be insignificant and approaching zero mortality and serious injury rate. The status of white-sided dolphins, relative to OSP is unknown. The data are insufficient to establish population trends 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.

OTHER FACTORS THAT MAY BE AFFECTING THE STOCK

Strandings

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 19 October 2022). Sixteen of these animals were released alive. Human Interaction (HI) was indicated in 7 records during this period, while in another 65 cases of human interaction was entered as “Could Not be Determined”. In only one of the positive HI cases was the HI listed as a possible contributor to the mortality (entanglement with beach protection mesh). 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 or stranding, 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 human interaction manual (Barco and Moore 2013) and case criteria for human interaction determinations (Moore et al. 2013) published in 2013 aimed to improve determination consistency among responders.

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 2018, 2019, 2020; Ledwell et al. 2021a, 2021b).

Table 5. Atlantic white-sided dolphin (*Lagenorhynchus acutus*) reported strandings along the U.S. and Canadian Atlantic coast, 2017–2021.

Area	2017	2018	2019	2020	2021	Total
Maine[b]	0	6	5	1	1	13
New Hampshire	0	0	2	0	0	2
Massachusetts[a, b, c, d]	10	41	65	7	4	127
Connecticut	1	0	0	0	0	12
TOTAL US	10	47	72	8	5	142
Nova Scotia[e]	8	0	0	3	4	15
Newfoundland and Labrador[f]	1	0	0	1	2	4
TOTAL US & CANADA	19	47	72	12	11	161

a. 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.
b. 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.
c. In 2020, 3 white-sided dolphins were coded as HI due to public attempts to refloat.
d. In 2021, 1 white-sided dolphin was coded as HI due to public attempts to refloat.
e. Data supplied by Nova Scotia Marine Animal Response Society (pers. comm.).
f. Ledwell and Huntington (2018, 2019, 2020) and Ledwell et al. (2021a, 2021b).

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.

Chavez-Rosales et al. (2022) documented an overall 178 km northeastward spatial distribution shift of the seasonal core habitat of Northwest Atlantic cetaceans that was related to changing habitat/climatic factors. Results varied by season and species. This study used sighting data collected during seasonal aerial and shipboard line transect abundance surveys during 2010 to 2017. During this time frame, the weighted centroid of white-sided dolphin core habitat moved less than 30 km in all seasons. Similarly, using historical stranding records, Thorne et al. (2022) demonstrated a poleward shift in cool water species of odontocetes, including a shift in white-sided dolphin strandings of approximately 5 km per year at the center of the distribution and 26 km per year at the trailing edge of their distribution. There is uncertainty in how, if at all, the changes in distribution and population size of cetacean species may interact with changes in distribution of prey species and how the ecological shifts will affect human impacts to the species.

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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) (Figure 1). 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).

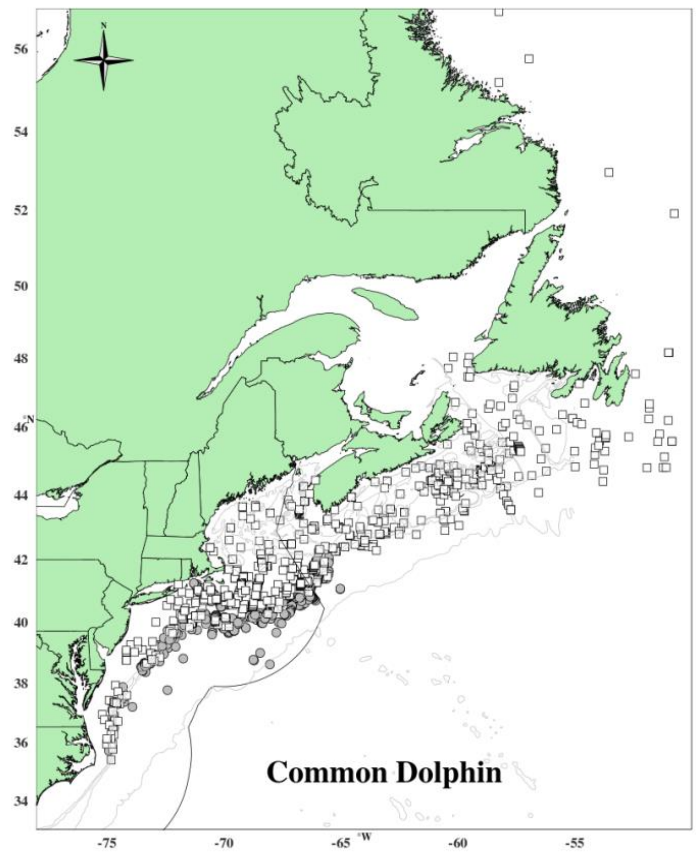

Figure 1. Distribution of common dolphin sightings from NEFSC and SEFSC shipboard (circles) and aerial surveys (squares) during the summers of 1998, 1999, 2002, 2004, 2006, 2007, 2010, 2011, 2016, and 2021 and Department of Fisheries and Oceans Canada 2007 TNASS and 2016 NAISS surveys. Isobaths are the 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 93,100 (CV=0.56) which is the total of NEFSC and SEFSC surveys conducted in 2021(Table 1). This estimate, derived from shipboard surveys, covers most of this stock's known range. Because the survey areas did not overlap, the estimates from the two surveys were added together and the CVs pooled using a delta method to produce a species abundance estimate for the stock area.

Earlier Abundance Estimates

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

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.

More recent abundance estimates of 85,035 (CV=0.61) and 8,065 (CV=0.86) common dolphins were generated from vessel surveys conducted in U.S. waters of the western North Atlantic during the summer of 2021 (Table 1; Garrison and Dias 2023; Palka 2023). One survey was conducted from 16 June to 23 August in waters north of 36ºN latitude and consisted of 5,871 km of on-effort trackline along the shelf break and offshore to the outer edge of the U.S. EEZ (NEFSC and SEFSC 2022). The second vessel survey covered waters from central Florida (25ºN latitude) to approximately 38ºN latitude between the 200-m isobaths and the outer edge of the U.S. EEZ during 12 June–31 August. A total of 5,659 km of trackline was covered on effort (NEFSC and SEFSC 2022). 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 the current best 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 is 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

Month/Year	Area	N_{est}	CV
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
Jun–Aug 2021	New Jersey to lower Bay of Fundy	85,035	0.61
Jun–Aug 2021	Central Florida to New Jersey	8,065	0.86
Jun–Aug 2021	**Central Florida to lower Bay of Fundy (COMBINED)**	**93,100**	**0.56**

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 93,100 animals (CV=0.56), derived from the 2021 shipboard surveys. The minimum population estimate for the western North Atlantic common dolphin is 59,897.

Current Population Trend

There are insufficient data to support a population trend analysis for this species.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 that suggests 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 59,897 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 599.

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
93,100	0.56	59,897	0.5	0.04	599

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 Est. Avg.	CV
2017–2021	U.S. fisheries using observer data	413	0.10
2017–2021	Research mortalities	0.2	
2017–2021	Non-fishery human-caused stranding mortalities	0.6	
TOTAL		413.8	

Uncertainties not accounted for include the potential that the observer coverage was not representative of the fishery during all times and places and was lower in multiple fisheries during the COVID-19 pandemic years (2020-2021) (Table 4). There are no major known sources of unquantifiable human-caused mortality or serious injury for this stock.

Pelagic Longline

Pelagic longline bycatch estimates of common dolphins for 2017–2021 were documented in Garrison and Stokes (2020a, 2020b, 2021, 2023a, 2023b). 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 Sink Gillnet

Annual common dolphin mortalities were estimated using annual ratio-estimator methods (Orphanides 2020, 2021; Precoda and Orphanides 2022, Precoda 2023). 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.

Mid-Atlantic Gillnet

Common dolphins were taken during observed trips in most years. Annual common dolphin mortalities were estimated using annual ratio-estimator methods (Orphanides 2019, 2020, 2021; Precoda and Orphanides 2022, Precoda 2023). 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

The Northeast Fisheries Science Center reported a common dolphin mortality that occurred during the fall research trawl survey in 2021.

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 Estimated Mortality
Northeast Sink Gillnet	2017 2018 2019 2020 2021	Obs. Data, Trip Logbook, Allocated Dealer Data	0.12 0.11 0.13 0.02 0.11	0 0 0 0 0	20 10 1 2 3	0 0 0 0 0	133 93 5.0 50 39	133 93 5.0 50 39	0.28 0.45 0.68 0.25 0.24	64(0.129)
Mid-Atlantic Gillnet	2017 2018 2019 2020 2021	Obs. Data, Weighout	0.09 0.09 0.13 0.03 0.01	1 0 0 0 0	1 1 3 0 0	11 1 0 5 4	11 7.7 20 25 20	22 7.7 20 30 24	0.71 0.91 0.56 0.55 0.33	21(0.33)
Northeast Bottom Trawl[c]	2017 2018 2019 2020 2021	Obs. Data, Logbook	0.12 0.12 0.16 0.08 0.19	0 0 0 0 0	0 4 2 2 8	0 0 0 0 0	0 28 10 50 38	22 16 0 28 10 50 43	0 0.54 0.62 0.25 0.42	64(0.18)
Mid-Atlantic Bottom Trawl[c]	2017 2018 2019 2020 2021	Obs. Data, Dealer Data	0.14 0.12 0.12 0.02 0.04	0 1 2 0 0	66 34 52 54 13	0 5 15 7 0	380 200 380 237 230	380 205 395 333 230	0.23 0.54 0.23 0.14 0.57	309(0.13)
Pelagic Longline	2017 2018 2019 2020 2021	Obs. Data, Logbook Data	0.12 0.10 0.10 0.09 0.08	1 1 0 0 0	0 0 0 0 0	4.92 1.44 0 0 0	0 0 0 0 0	4.92 1.44 0 0 0	1 1 0 0 0	1.27(0.81)
TOTAL										413(0.10)

a. Observer data (Obs. Data), used to measure bycatch rates, are collected within the Northeast Fisheries Observer Program and At-sea Monitoring Program. NEFSC collects landings data (unallocated Dealer Data or Allocated Dealer Data) which are used as a measure of total landings and mandatory Vessel Trip Reports (VTR; Trip Logbook) are used to determine the spatial distribution of landings and fishing effort.
b. Observer coverage is defined as the ratio of observed to total metric tons of fish landed for the gillnet fisheries and the ratio of observed to total trips for bottom trawl and Mid-Atlantic mid-water trawl (including pair trawl) fisheries.
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 and Lyssikatos 2023)

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 2017–2021 average annual human-related mortality does not exceed PBR. The total U.S. fishery-related mortality and serious injury for this stock is over 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.

OTHER FACTORS THAT MAY BE AFFECTING THE STOCK

Strandings

Common dolphin strandings between Maine and Florida are reported in Table 5 (NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 19 October 2022). The total includes mass-stranded common dolphins in Massachusetts during 2017 (over 90 animals in 20 events), 2018 (a total of 28 animals in 9 events), 2019 (28 animals in 9 events), 2020 (79 animals in ~8 events) and 2021 (47 animals in ~11 events). Animals released or last sighted alive include 70 in 2017, 18 in 2018, 29 in 2019, 60 in 2020 and 48 in 2021. Six common dolphin mortalities in 2017 were coded as confirmed human interaction (HI), 1 in Rhode Island and 5 in Massachusetts. Of these, 2 were classified as fishery interactions (1 in Massachusetts and 1 in Rhode Island), 1 was classified as a possible boat collision, and 1 was released alive. Another dolphin was euthanized after multiple restrandings, and another was determined to be a human interaction case due to beachgoer intervention. In 95 stranding cases in 2017, human interaction was listed as CBD (could not be determined). In 2018, 5 cases were coded as definite human interactions, 1 in Virginia and 4 in Massachusetts. Of these, two were public harassment and 3 involved fishing gear, though only one was classified as a fishery interaction. (In the other 2 cases, HI was deemed "other human interaction" instead of fishery interaction possibly because it was unknown if gear was actively fished). Another 55 records in 2018 had CBD listed in the HI column. Eight stranding mortalities in Massachusetts in 2019 were classified as human interactions and 1each 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. Another 69 mortalities in 2019 were listed as CBD in the HI column. In 2020, a total of 6 common dolphin strandings were classified as confirmed HI, and another 88 as CBD HI, with 1 North Carolina and 1 New York stranding classified as fisheries interaction. However, only 1 of those had the fishery interaction deemed a "probable" contribution to cause stranding. In 2021, 11 stranding mortalities were classified as confirmed human interactions, 4 in New York and 7 in Massachusetts and 72 as CBD human interactions. One of those NY HI cases was classified as a fishery interaction. This was the only case where the interaction event was coded as a probable contributor to the stranding. In this 5-year period, only 3 interactions (the boat strike in 2017 and the 2 "other HI" cases in 2018) were likely non-fishery human-caused mortalities.

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 5 common dolphins stranded in 2017, 5 in 2018, 4 in 2019, 4 in 2020 and 15 in 2021 (Tonya Wimmer/Andrew Reid, pers. comm.).

*Table 5. Common dolphin (***Delphinus delphis delphis***) reported strandings along the U.S. Atlantic coast, 2017–2021.*

STATE	2017	2018	2019	2020	2021	TOTALS
Maine	0	0	0	1	1	2
New Hampshire	2	0	0	0	1	3
Massachusetts	166	61	95	136	122	580
Rhode Island	5	4	5	13	6	33
Connecticut	1	0	0	0	0	1
New York	15	11	9	15	31	81
New Jersey	0	2	4	6	5	17
Delaware	0	0	1	0	1	2
Maryland	0	0	2	2	2	6
Virginia	1	3	5	2	2	13
North Carolina	0	3	4	7	0	14
TOTALS	190	84	125	182	171	752

It should be recognized that evidence of human interaction does not always indicate cause of death or stranding, 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 human interaction manual (Barco and Moore 2013) and case criteria for human interaction determinations (Moore et al. 2013) published in 2013 aimed to improve determination consistency among responders.

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 as the main 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 not 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.

Chavez-Rosales et al. (2022) documented an overall 178 km northeastward spatial distribution shift of the seasonal core habitat of Northwest Atlantic cetaceans that was related to changing habitat/climatic factors. Results varied by season and species. This study used sightings data collected during seasonal aerial and shipboard line transect abundance surveys during 2010 to 2017. During this time frame, the weighted centroid of the common dolphin core habitat moved farthest during fall (216 km towards the northeast) and least during summer (111 km). There is uncertainty in how, if at all, the changes in distribution and population size of cetacean species may interact with changes in distribution of prey species and how the ecological shifts will affect human impacts to the species.

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

STOCK DEFINITION AND GEOGRAPHIC RANGE

Atlantic spotted dolphins are distributed in tropical and warm temperate waters of the western North Atlantic (Leatherwood et al. 1976). Their distribution ranges from southern New England, south through the Gulf of Mexico and the Caribbean to at least Venezuela (Leatherwood et al. 1976; Perrin et al. 1994). Atlantic spotted dolphins regularly occur in continental shelf and continental slope waters (Figure 1; Payne et al. 1984; Mullin and Fulling 2003). Sightings have also been made along the north wall of the Gulf Stream and warm-core ring features (Waring et al. 1992). Because there are confirmed sightings within waters of Canada and the Bahamas, this is likely a transboundary stock (e.g., Halpin et al. 2009; Dunn 2013; DFO 2017; Emery 2020; Figure 1).

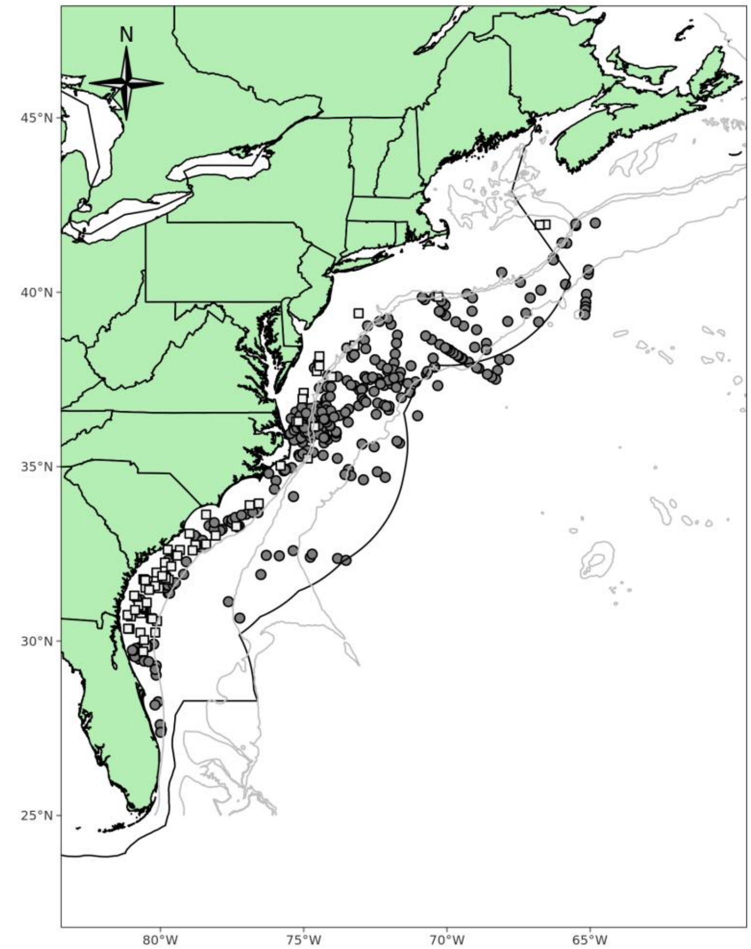

Figure 1. Distribution of Atlantic spotted dolphin sightings from NEFSC and SEFSC shipboard (circles) and aerial (squares) surveys during 1995, 1998, 1999, 2002, 2004, 2006, 2007, 2008, 2010, 2011, 2016 and 2021. Isobaths are the 200-m, 1,000-m, and 4,000-m depth contours. The darker line indicates the U.S. EEZ.

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

Genetic analyses of mtDNA and microsatellite DNA data from samples collected in the Gulf of Mexico and the western North Atlantic revealed significant genetic differentiation between these two areas (Adams and Rosel 2006; Viricel and Rosel 2014; do Amaral et al. 2021), supporting delineation of a demographically independent population for each area. In addition, the genetic data provided evidence for separation of dolphins within the western North Atlantic, suggesting the Western North Atlantic stock of Atlantic spotted dolphins may comprise multiple demographically independent populations (Adams and Rosel 2006; Viricel and Rosel 2014). One population consists of the smaller, pelagic form and occupies waters over the continental slope and deeper. The second population is restricted to continental shelf waters at and south of Cape Hatteras. The two genetically-identified populations correspond with the two morphological forms identified by Perrin et al. (1987), and the level of genetic differentiation between them indicates they are independent evolutionary pathways with dispersal rates of less than 0.3% per generation (Viricel and Rosel 2014). Population-level differentiation appears to exist within the Gulf of Mexico as well, with a break between western and eastern populations occurring in the north central Gulf of Mexico (Viricel and Rosel 2014).

POPULATION SIZE

The best abundance estimate available for Atlantic spotted dolphins in the western North Atlantic is 31,506 (CV=0.28; Table 1; Garrison and Dias 2023; Palka 2023). This estimate is from summer 2021 surveys covering waters from central Florida to the lower Bay of Fundy. Distinction between the two Atlantic spotted dolphin ecotypes has not regularly been made during surveys, and at their November 1999 meeting, the Atlantic SRG recommended that without a genetic determination of stock structure for the two ecotypes, the abundance estimates for the coastal and offshore forms should be combined. The abundance estimate provided here is a species-specific estimate combining both ecotypes of Atlantic spotted dolphins.

Earlier Abundance Estimates

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

Recent Surveys and Abundance Estimates

Abundance estimates of 8,247 (CV=0.24) and 31,674 (CV=0.33) Atlantic spotted dolphins were generated from vessel surveys conducted in U.S. waters of the western North Atlantic during the summer of 2016 (Table 1; Garrison 2020; Palka 2020). One survey was conducted from 27 June to 25 August in waters north of 38ºN latitude and consisted of 5,354 km of on-effort trackline along the shelf break and offshore to the outer edge of the U.S. EEZ (NEFSC and SEFSC ?2018). The second vessel survey covered waters from Central Florida to approximately 38ºN latitude between the 100-m isobaths and the outer edge of the U.S. EEZ during 30 June–19 August. A total of 4,399 km of trackline was covered on effort (NEFSC and SEFSC 2016). 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.

More recent abundance estimates of 8,112 (CV=0.22) and 23,394 (CV=0.37) Atlantic spotted dolphins were generated from vessel surveys conducted in U.S. waters of the western North Atlantic during the summer of 2021 (Table 1; Garrison and Dias 2023; Palka 2023). One survey was conducted from 16 June to 23 August in waters north of 36ºN latitude and consisted of 5,871 km of on-effort trackline along the shelf break and offshore to the outer edge of the U.S. EEZ (NEFSC and SEFSC 2022). The second vessel survey covered waters from central Florida (25ºN latitude) to approximately 38ºN latitude between the 200-m isobaths and the outer edge of the U.S. EEZ during 12 June–31 August. A total of 5,659 km of trackline was covered on effort (NEFSC and SEFSC 2022). Both surveys utilized two visual teams and an independent observer approach to estimate detection probability on the trackline (Laake and Borchers 2004). Mark-recapture distance sampling was used to estimate abundance. Estimates from the two surveys were combined and CVs pooled to produce a species abundance estimate for the stock area.

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

Month/Year	Area	N_{best}	CV
Jun–Aug 2016	New Jersey to Bay of Fundy	8,247	0.24
Jun–Aug 2016	Central Florida to New Jersey	31,674	0.33
Jun–Aug 2016	Central Florida to Bay of Fundy (COMBINED)	39,921	0.27
Jun–Aug 2021	New Jersey to lower Bay of Fundy	8,112	0.22
Jun–Aug 2021	Central Florida to New Jersey	23,394	0.37
Jun–Aug 2021	**Central Florida to lower Bay of Fundy (COMBINED)**	**31,506**	**0.28**

Minimum Population Estimate

The minimum population estimate is the lower limit of the two-tailed 60% confidence interval of the log- normally distributed best abundance estimate. This is equivalent to the 20th percentile of the log-normal distribution as specified by Wade and Angliss (1997). The best abundance estimate is 31,506 (CV=0.28). The minimum population estimate based on the 2021 abundance estimates is 25,042 (Table 2).

Current Population Trend

There are four available coastwide abundance estimates for Atlantic spotted dolphins from the summers of 2004, 2011, 2016, and 2021. Each of these is derived from vessel surveys with similar survey designs and all four used the two-team independent observer approach to estimate abundance. The resulting estimates were 50,978 (CV=0.42) in 2004, 44,715 (CV=0.43) in 2011, 39,921 (CV=0.27) in 2016, and 31,506 (CV=0.28) in 2021 (Garrison 2020; Garrison and Dias 2023). A generalized linear model indicated a statistically significant (p=0.028) linear decrease in these abundance estimates. A key uncertainty in this assessment of trend is that interannual variation in abundance may be caused by either changes in spatial distribution associated with environmental variability or changes in the population size of the stock.

CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

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

POTENTIAL BIOLOGICAL REMOVAL

Potential Biological Removal (PBR) is the product of minimum population size, one-half the maximum productivity rate, and a recovery factor (MMPA Sec. 3. 16 U.S.C. 1362; Wade and Angliss 1997). The minimum population size for the Atlantic spotted dolphin is 25,042. The maximum productivity rate is 0.04, the default value for cetaceans. The recovery factor, which accounts for endangered, depleted, threatened stocks, or stocks of unknown status relative to optimum sustainable population (OSP), is set to 0.5 because this stock is of unknown status. PBR for the combined offshore and coastal forms of Atlantic spotted dolphins is 250 (Table 2).

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

N_{est}	CV N_{est}	N_{min}	F_r	R_{max}	PBR
31,506	0.28	25,042	0.5	0.04	250

ANNUAL HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

Total annual estimated human-caused mortality and serious injury to this stock during 2017–2021 was presumed to be zero, as there were no reports of mortalities or serious injuries to Atlantic spotted dolphins in the western North Atlantic. Recorded takes of Atlantic spotted dolphins in fisheries in the western North Atlantic are rare. However, observer coverage in the fisheries is relatively low. Furthermore, the likelihood is low that a dolphin killed at sea due to a fishery interaction or vessel strike will be recovered (Williams et al. 2011). These factors introduce some uncertainty into estimating the true level of human-caused mortality and serious injury for this stock.

Fishery Information

There are two commercial fisheries that interact, or that could potentially interact, with this stock in the Atlantic Ocean. These are the Category I Atlantic Highly Migratory Species longline and the Atlantic Ocean, Caribbean, Gulf of Mexico large pelagics longline fisheries (Appendix III). Percent observer coverage (percentage of sets observed) for these longline fisheries in the Atlantic for each year during 2017–2021 was 11, 10, 10, 9, and 8, respectively.

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

The Atlantic Ocean, Caribbean, Gulf of Mexico large pelagics longline fishery operates in the U.S. Atlantic (including Caribbean) and Gulf of Mexico EEZ, and pelagic swordfish, tunas and billfish are the target species. There were no observed mortalities or serious injuries to Atlantic spotted dolphins by this fishery in the Atlantic Ocean during 2017–2021 (Garrison and Stokes 2020a; 2020b; 2021; 2023a; 2023b).

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

STATUS OF STOCK

Atlantic spotted dolphins are not listed as threatened or endangered under the Endangered Species Act, and the Western North Atlantic stock is not considered strategic under the Marine Mammal Protection Act. No fishery-related mortality or serious injury has been observed during recent years; therefore, total fishery-related mortality and serious injury can be considered insignificant and approaching the zero mortality and serious injury rate. The status of Atlantic spotted dolphins relative to optimum sustainable population is unknown. Available abundance estimates indicate a decline in population size for this species between 2004 and 2021, but it is uncertain if this is a true decline or simply a change in distribution with animals moving outside of the study area.

OTHER FACTORS THAT MAY BE AFFECTING THE STOCK

Strandings

During 2017–2021, 21 Atlantic spotted dolphins were reported stranded along the U.S. East Coast (Table 3; NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 13 October 2022 (Southeast Region [SER]) and 18 September 2022 (Northeast Region [NER])). Evidence of human interaction was detected for four of the strandings (all animals pushed out to sea by members of the public). No evidence of human interaction was detected for seven strandings, and for the remaining ten 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.

Stranding data underestimate the extent of human and fishery-related mortality and serious injury because not all of the marine mammals that die or are seriously injured in human interactions wash ashore, or, if they do, they are not all recovered (Peltier et al. 2012; Wells et al. 2015; 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), and decomposition can also introduce uncertainty in visual species identification of a carcass, particularly for closely related species like those in the genus Stenella. Finally, the level of technical expertise among stranding network personnel varies widely as does the ability to recognize signs of human interaction.

Table 3. Atlantic spotted dolphin (**Stenella frontalis*****) reported strandings along the U.S. Atlantic coast, 2017–2021. Data are from the NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 13 October 2022 (SER) and 18 September 2022 (NER).***

STATE	2017	2018	2019	2020	2021	TOTALS
New York	0	0	1	0	0	1
New Jersey	0	0	0	2	1	3
North Carolina	1	3	2	0	2	8
South Carolina	0	1	1	3	2	7
Florida	0	0	0	2	0	2
TOTALS	1	4	4	7	5	21

Habitat Issues

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

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

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

Climate-related changes in spatial distribution and abundance, including poleward and depth shifts, have been documented in or predicted for plankton species and commercially important fish stocks (Nye et al. 2009; Pinsky et al. 2013; Poloczanska et al. 2013; Grieve et al. 2017; Morley et al. 2018) and cetacean species (e.g., MacLeod 2009; Sousa et al. 2019). Chavez-Rosales et al. (2022) documented an overall 178 km northeastward spatial distribution shift of the seasonal core habitat of Northwest Atlantic cetaceans that was related to changing habitat/climatic factors. Results varied by season and species. This study used sightings data collected during seasonal aerial and shipboard line transect abundance surveys during 2010 to 2017. During this time frame, the weighted centroid of the Atlantic spotted dolphin moved farthest in winter (165 km towards the northeast) and least in fall (25 km towards the southeast). There is uncertainty in how, if at all, the changes in distribution and population size of cetacean species may interact with changes in distribution of prey species and how the ecological shifts will affect human impacts to the species.

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

STOCK DEFINITION AND GEOGRAPHIC RANGE

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

Sightings during surveys in the Atlantic north of Cape Hatteras have been along the continental slope while in waters south of Cape Hatteras sightings were recorded over the Blake Plateau and in deeper offshore waters of the mid-Atlantic (Figure 1). Because there are confirmed sightings within waters of the Bahamas, this is likely a transboundary stock (e.g., Halpin et al. 2009; Dunn 2013).
Pantropical spotted dolphins in the western North Atlantic are managed separately from those in the northern Gulf of Mexico. Although there have been no directed studies of the degree of demographic independence between the two areas, this management structure is consistent with evidence for population structure in other areas, including more pelagic waters of the eastern tropical Pacific (Leslie and Morin 2016), and is further supported because the two stocks occupy distinct marine ecoregions (Spalding et al. 2007; Moore and Merrick 2011). Due to the paucity of sightings, there are insufficient data to determine whether the western North Atlantic stock comprises multiple demographically independent populations. Additional morphological, acoustic, genetic, and/or behavioral data are needed to further delineate population structure within the western North Atlantic and across the broader geographic area.

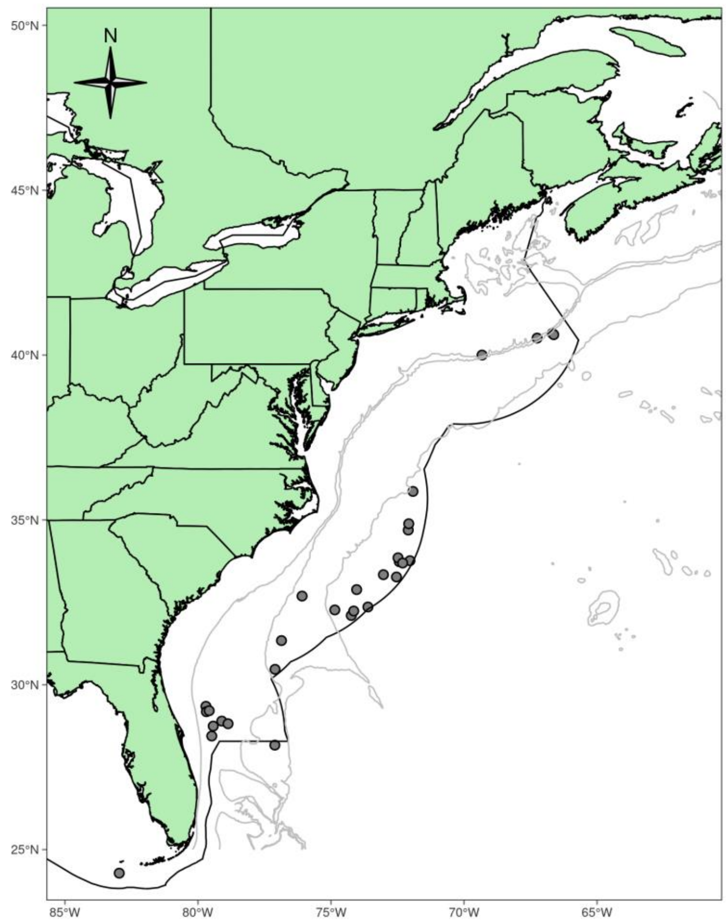

Figure 1. Distribution of pantropical spotted dolphin sightings from NEFSC and SEFSC shipboard (circles) and aerial (squares) surveys during 1995, 1998, 1999, 2002, 2004, 2006, 2007, 2008, 2010, 2011, 2016 and 2021. Isobaths are the 200-m, 1,000-m, and 4,000-m depth contours. The darker line indicates the U.S. EEZ.

POPULATION SIZE

The best abundance estimate available for western North Atlantic pantropical spotted dolphins is 2,757 (CV=0.50; Table 1; Garrison and Dias 2023; Palka 2023). This estimate is from summer 2021 surveys covering waters from central Florida to the lower Bay of Fundy.

Earlier Abundance Estimates

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

Recent Surveys and Abundance Estimates

Abundance estimates of 0 and 6,593 (CV=0.52) pantropical spotted dolphins were generated from two non-overlapping vessel surveys conducted in U.S. waters of the western North Atlantic during the summer of 2016 (Table 1; Garrison 2020; Palka 2020). One survey was conducted from 27 June to 25 August in waters north of 38ºN latitude and included 5,354 km of on-effort trackline along the shelf break and offshore to the U.S. EEZ (NEFSC and SEFSC 2018). The second vessel survey covered waters from Central Florida to approximately 38ºN latitude between the 100-m isobaths and the U.S. EEZ from 30 June–19 August. A total of 4,399 km of trackline was covered on effort (NEFSC and SEFSC 2018). Both surveys utilized two visual teams and an independent observer approach to estimate detection probability on the trackline (Laake and Borchers 2004). Mark-recapture distance sampling was used to estimate abundance (Thomas et al. 2009).

More recent abundance estimates of 0 and 2,757 (CV=0.50) pantropical spotted dolphins were generated from vessel surveys conducted in U.S. waters of the western North Atlantic during the summer of 2021 (Table 1; Garrison and Dias 2023; Palka 2023). One survey was conducted from 16 June to 23 August in waters north of 36ºN latitude and consisted of 5,871 km of on-effort trackline along the shelf break and offshore to the outer edge of the U.S. EEZ (NEFSC and SEFSC 2022). The second vessel survey covered waters from central Florida (25ºN latitude) to approximately 38ºN latitude between the 200-m isobaths and the outer edge of the U.S. EEZ during 12 June–31 August. A total of 5,659 km of trackline was covered on effort (NEFSC and SEFSC 2022). Both surveys utilized two visual teams and an independent observer approach to estimate detection probability on the trackline (Laake and Borchers 2004). Mark-recapture distance sampling was used to estimate abundance. Estimates from the two surveys were combined and CVs pooled to produce a species abundance estimate for the stock area.

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

Month/Year	Area	N_{best}	CV
Jun–Aug 2016	New Jersey to lower Bay of Fundy	0	-
Jun–Aug 2016	Central Florida to New Jersey	6,593	0.52
Jun–Aug 2016	Central Florida to lower Bay of Fundy (COMBINED)	6,593	0.52
Jun–Aug 2021	New Jersey to lower Bay of Fundy	0	-
Jun–Aug 2021	Central Florida to New Jersey	2,757	0.50
Jun–Aug 2021	**Central Florida to lower Bay of Fundy (COMBINED)**	**2,757**	**0.50**

Minimum Population Estimate

The minimum population estimate is the lower limit of the two-tailed 60% confidence interval of the log-normally distributed best abundance estimate. This is equivalent to the 20th percentile of the log-normal distribution as specified by Wade and Angliss (1997). The best estimate of abundance for pantropical spotted dolphins is 2,757 (CV=0.50). The minimum population estimate for pantropical spotted dolphins is 1,856 (Table 2).

Current Population Trend

There are four available coastwide abundance estimates for pantropical spotted dolphins from the summers of 2004, 2011, 2016, and 2021. Each of these is derived from vessel surveys with similar survey designs and all three used the two-team independent observer approach to estimate abundance. The resulting estimates were 4,439 (CV=0.49) in 2004, 3,333 (CV=0.91) in 2011, 6,593 (CV=0.52) in 2016, and 2,757 (CV=0.50) in 2021 (Garrison 2020; Garrison and Dias 2023). A generalized linear model indicated no statistically significant (p=0.659) linear trend in these abundance estimates. The high uncertainty in these estimates limits the ability to detect a population trend. In addition, a key uncertainty in this assessment of trend is that interannual variation in abundance may be caused by either changes in spatial distribution associated with environmental variability or changes in the population size of the stock.

CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

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

POTENTIAL BIOLOGICAL REMOVAL

Potential Biological Removal (PBR) is the product of minimum population size, one-half the maximum productivity rate, and a "recovery" factor (MMPA Sec. 3. 16 U.S.C. 1362; Wade and Angliss 1997). The minimum population size for pantropical spotted dolphins is 1,856. The maximum productivity rate is 0.04, the default value for cetaceans. The "recovery" factor, which accounts for endangered, depleted, threatened stocks, or stocks of unknown status relative to optimum sustainable population (OSP) is assumed to be 0.5 because this stock is of unknown status. PBR for pantropical spotted dolphins is 19 (Table 2).

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

N_{est}	CV N_{est}	N_{min}	F_r	R_{max}	PBR
2,757	0.50	1,856	0.5	0.04	19

ANNUAL HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

Total annual estimated human-caused mortality and serious injury to this stock during 2017–2021 was presumed to be zero, as there were no reports of mortalities or serious injuries to pantropical spotted dolphins in the western North Atlantic. Recorded takes of pantropical spotted dolphins in fisheries in the western North Atlantic are rare. However, observer coverage in the fisheries is relatively low. Furthermore, the likelihood is low that a dolphin killed at sea due to a fishery interaction or vessel-strike will be recovered (Williams et al. 2011). These factors introduce some uncertainty into estimating the true level of human-caused mortality and serious injury for this stock.

Fishery Information

There are two commercial fisheries that interact, or that could potentially interact, with this stock in the Atlantic Ocean. These are the Category I Atlantic Highly Migratory Species longline and the Atlantic Ocean, Caribbean, Gulf of Mexico large pelagics longline fisheries (Appendix III). Percent observer coverage (percentage of sets observed) for these longline fisheries in the Atlantic for each year during 2017–2021 was 11, 10, 10, 9, and 8, respectively.

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

The Atlantic Ocean, Caribbean, Gulf of Mexico large pelagics longline fishery operates in the U.S. Atlantic (including Caribbean) and Gulf of Mexico EEZ, and pelagic swordfish, tunas and billfish are the target species. There were no observed mortalities or serious injuries to pantropical spotted dolphins by this fishery in the Atlantic Ocean during 2017–2021 (Garrison and Stokes 2020a; 2020b; 2021; 2023a; 2023b).

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

STATUS OF STOCK

Pantropical spotted dolphins are not listed as threatened or endangered under the Endangered Species Act, and the Western North Atlantic stock is not considered strategic under the Marine Mammal Protection Act. No fishery-related mortality or serious injury has been observed during recent years; therefore, total fishery-related mortality and serious injury can be considered insignificant and approaching the zero mortality and serious injury rate. The status of pantropical spotted dolphins in the western U.S. Atlantic EEZ relative to optimum sustainable population is unknown. There was no statistically significant trend in population size for this species.

OTHER FACTORS THAT MAY BE AFFECTING THE STOCK

Strandings

During 2017–2021, three pantropical spotted dolphins were reported stranded on the U.S. East Coast, all occurring in Florida during 2018 (n=1) and 2020 (n=2) (NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 13 October 2022 (Southeast Region) and 18 September 2022 (Northeast Region). Evidence of human interaction was detected for two of the strandings (both animals pushed out to sea by members of the public). No evidence of human interaction was detected for the remaining stranding. It should be noted that evidence of human interaction does not necessarily mean the interaction caused the animal's stranding or death.

Stranding data underestimate the extent of human and fishery-related mortality and serious injury because not all of the marine mammals that die or are seriously injured in human interactions wash ashore, or, if they do, they are not all recovered (Peltier et al. 2012; Wells et al. 2015; 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), and decomposition can also introduce uncertainty in visual species identification of a carcass, particularly for closely related species like those in the genus Stenella. Finally, the level of technical expertise among stranding network personnel varies widely as does the ability to recognize signs of human interaction.

Habitat Issues

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

The chronic impacts of contaminants (polychlorinated biphenyls [PCBs] and chlorinated pesticides [DDT, DDE, dieldrin, etc.]) on marine mammal reproduction and health are of concern (e.g., Schwacke et al. 2002; Jepson et al. 2016; Hall et al. 2018), but research on contaminant levels for this stock is lacking.

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

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

STOCK DEFINITION AND GEOGRAPHIC RANGE

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

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

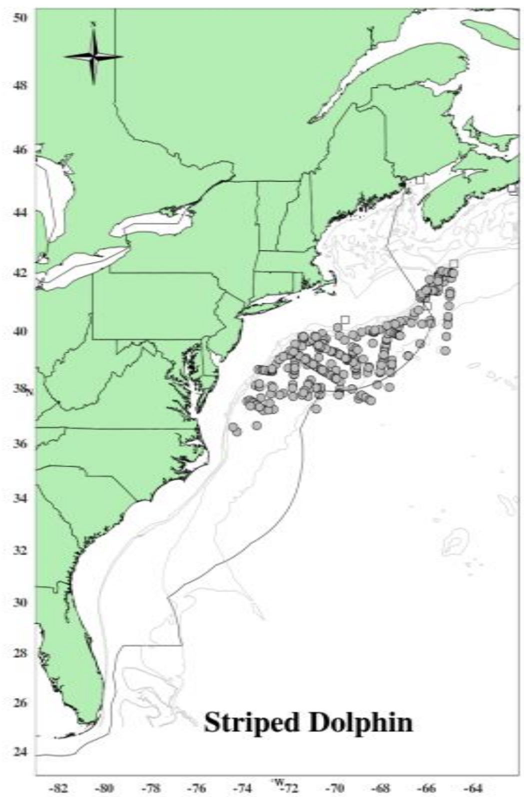

Figure 1: Distribution of striped dolphin sightings from NEFSC and SEFSC shipboard and aerial surveys during the summers of 1998, 1999, 2002, 2004, 2006, 2007, 2010, 2011, 2016, and 2021. Isobaths are the 100-m, 200-m, 1000-m and 4000-m depth contours. Circle symbols represent shipboard sightings and squares are aerial sightings.

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

POPULATION SIZE

Several abundance estimates from selected regions are available for striped dolphins for select time periods. Sightings are almost exclusively in the continental shelf edge and continental slope areas west of Georges Bank (Figure 1). The best abundance estimate for striped dolphins is the sum of the 2021 survey estimates—48,274 (CV=0.29).

Recent Surveys and Abundance Estimates

Abundance estimates of 42,783 (CV=0.25) and 24,163 (CV=0.66) striped dolphins were generated from vessel surveys conducted in U.S. waters of the western North Atlantic during the summer of 2016 (Table 1; Garrison in 2020; Palka 2020). One survey was conducted from 27 June to 25 August in waters north of 38ºN latitude and consisted of 5,354 km of on-effort trackline along the shelf break and offshore to the outer limit of the U.S. EEZ (NEFSC and SEFSC 2018). The second vessel survey covered waters from Central Florida to approximately 38ºN latitude between the 100-m isobaths and the outer limit of the U.S. EEZ during 30 June–19 August. A total of 4,399 km of trackline was covered on effort (NEFSC and SEFSC 2018). Both surveys utilized two visual teams and an independent observer approach to estimate detection probability on the trackline (Laake and Borchers 2004). Mark-recapture distance sampling was used to estimate abundance. Estimates from the two surveys were combined and CVs pooled to produce a species abundance estimate for the stock area.

More recent abundance estimates of 38,522 (CV=0.34) and 9,752 (CV=0.49) striped dolphins were generated from vessel surveys conducted in U.S. waters of the western North Atlantic during the summer of 2021 (Table 1; Garrison and Dias 2023; Palka 2023). One survey was conducted from 16 June to 23 August in waters north of 36ºN latitude and consisted of 5,871 km of on-effort trackline along the shelf break and offshore to the outer edge of the U.S. EEZ (NEFSC and SEFSC 2022). The second vessel survey covered waters from central Florida (25ºN latitude) to approximately 38ºN latitude between the 200-m isobaths and the outer edge of the U.S. EEZ during 12 June–31 August. A total of 5,659 km of trackline was covered on effort (NEFSC and SEFSC 2022). Both surveys utilized two visual teams and an independent observer approach to estimate detection probability on the trackline (Laake and Borchers 2004). Mark-recapture distance sampling was used to estimate abundance. Estimates from the two surveys were combined and CVs pooled to produce a species abundance estimate for the stock area.

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

Month/Year	Area	N_{best}	CV
Jun–Sep 2016	Central Virginia to lower Bay of Fundy	42,783	0.25
Jun–Sep 2016	Florida to Central Virginia	24,163	0.66
Jun–Sep 2016	Florida to lower Bay of Fundy (COMBINED)	67,036	0.29
Jun–Aug 2021	New Jersey to lower Bay of Fundy	38,522	0.34
Jun–Aug 2021	Central Florida to New Jersey	9,752	0.49
Jun–Aug 2021	**Central Florida to lower Bay of Fundy (COMBINED)**	**48,274**	**0.29**

Minimum Population Estimate

The minimum population estimate is the lower limit of the two-tailed 60% confidence interval of the log-normally distributed best abundance estimate. This is equivalent to the 20th percentile of the log-normal distribution as specified by Wade and Angliss (1997). The best estimate of abundance for striped dolphins is 48,274 (CV=0.29), obtained from the 2021 surveys. The minimum population estimate for the western North Atlantic striped dolphin is 38,040.

Current Population Trend

There are insufficient data to establish population trends for this species. The statistical power to detect a trend in abundance for this stock is poor due to the relatively imprecise abundance estimates and long survey interval. For example, the power to detect a precipitous decline in abundance (i.e., 50% decrease in 15 years) with estimates of low precision (e.g., CV > 0.30) remains below 80% (alpha = 0.30) unless surveys are conducted on an annual basis (Taylor et al. 2007). There is current work to standardize the strata-specific previous abundance estimates to consistently represent the same regions and include appropriate corrections for perception and availability bias. These standardized abundance estimates will be used in state-space trend models that incorporate environmental factors that could potentially influence the process and observational errors for each stratum.

CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

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

POTENTIAL BIOLOGICAL REMOVAL

Potential Biological Removal (PBR) is the product of minimum population size, one-half the maximum productivity rate, and a recovery factor (MMPA Sec. 3. 16 U.S.C. 1362; Wade and Angliss 1997). The minimum population size is 38,040. The maximum productivity rate is 0.04, the default value for cetaceans. The recovery factor, which accounts for endangered, depleted, threatened stocks, or stocks of unknown status relative to optimum sustainable population (OSP) is 0.5 because this stock is of unknown status. PBR for the western North Atlantic striped dolphin is 380.

Table 2. Best and minimum abundance estimates for western North Atlantic striped dolphins with Maximum Productivity Rate (R_{max}), Recovery Factor (F_r) and PBR.

N_{est}	CV N_{est}	N_{min}	F_r	R_{max}	PBR
48,274	0.29	38,040	0.5	0.04	380

ANNUAL HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

Total annual estimated average fishery-related mortality to this stock during 2017-2021 was zero striped dolphins.

Fishery Information

Detailed fishery information is reported in Appendix III.

Earlier Interactions

See Appendix V for more information on historical takes.

STATUS OF STOCK

Striped dolphins are not listed as threatened or endangered under the Endangered Species Act, and the Western North Atlantic stock is not considered strategic under the Marine Mammal Protection Act. Average annual human-related mortality and serious injury does not exceed the PBR. The total U.S. fishery-related mortality and serious injury for this stock is less than 10% of the calculated PBR, therefore can be considered to be insignificant and approaching zero mortality and serious injury rate. The status of striped dolphins, relative to OSP is unknown. There are insufficient data to determine the population trends for this species.

OTHER FACTORS THAT MAY BE CAUSING A DECLINE OR IMPEDING RECOVERY

Strandings

A total of 22 striped dolphins were reported stranded along the U.S. Atlantic coast between 2017 and 2021 (Table 3; NOAA National Marine Mammal Health and Stranding Response Database, accessed 15 October 2022).

In eastern Canada, 19 strandings were reported between 2017 and 2021. As noted above, 2017 marked the first time a striped dolphin stranding was reported in Newfoundland and Labrador.

Table 3. Striped dolphin reported strandings along the U.S. Atlantic and Canadian coast 2017-2021.

Area	2017	2018	2019	2020	2021	Total
Maine	0	0	0	1	0	1
Massachusetts	1	0	1	0	0	2
Rhode Island	0	1	0	0	0	1
New York[a]	0	0	0	3	1	4
New Jersey	4	1	1	0	2	8
Delaware	0	0	1	0	0	1
Virginia	0	0	0	1	0	1
North Carolina	0	1	0	1	0	2
Florida	2	0	0	0	0	2
U.S. TOTAL	8	3	4	6	3	22
Nova Scotia/Prince Edward Island[b,c]	9	1	2	1	5	18
Newfoundland	1	0	0	0	0	1

Area	2017	2018	2019	2020	2021	Total
and New Brunswick[d]						
GRAND TOTAL	18	4	6	7	8	41

a. Two of the New York animals were classified as HI due to public harassment/attempts to rescue.
b. Three of the 2017 animals released alive.
c. Data supplied by Nova Scotia Marine Animal Response Society (pers. comm.).
d. Ledwell et al. 2018, Ledwell et al. 2021a, 2021b

Habitat Issues

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

Climate-related changes in spatial distribution and abundance, including poleward and depth shifts, have been documented in or predicted for plankton species and commercially important fish stocks (Nye et al. 2009; Head et al. 2010; Pinsky et al. 2013; Poloczanska et al. 2013; Hare et al. 2016; Grieve et al. 2017; Morley et al. 2018) and cetacean species (e.g., MacLeod 2009; Sousa et al. 2019). There is uncertainty in how, if at all, the distribution and population size of this species will respond to these changes and how the ecological shifts will affect human impacts to the species. A recent study by Chavez-Rosales et al. (2022) documented an overall 178 km northeastward spatial distribution shift of the seasonal core habitat of Northwest Atlantic cetaceans that was related to changing habitat/climatic factors. Results varied by season and species. This study used sightings data collected during seasonal aerial and shipboard line transect abundance surveys during 2010 to 2017. During this time frame, the weighted centroid of striped dolphin core habitat moved farthest during fall (155 km towards the northeast) and least during winter (30 km). There is uncertainty in how, if at all, the changes in distribution and population size of cetacean species may interact with changes in distribution of prey species and how the ecological shifts will affect human impacts to the species.

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

STOCK DEFINITION AND GEOGRAPHIC RANGE

Fraser's dolphin is distributed worldwide in tropical waters (Perrin et al. 1994), and has recently been reported from temperate and subtropical areas of the North Atlantic (Gomes-Pereira et al. 2013). They are generally oceanic in distribution but may be seen closer to shore where deep water can be found near the shore, such as in the Lesser Antilles of the Caribbean Sea (Dolar 2009). Sightings of this species are rare, and in fact there has been only a single sighting on NMFS surveys in the western North Atlantic (Figure 1). Sightings in the more extensively surveyed northern Gulf of Mexico are uncommon but occur on a regular basis in oceanic waters (>200m) and in all seasons (Leatherwood et al. 1993; Hansen et al. 1996; Mullin and Hoggard 2000; Mullin and Fulling, 2004). Fraser's dolphins in the western North Atlantic are managed separately from those in the northern Gulf of Mexico. Although there have been no directed studies of the degree of demographic independence between the two areas, this management structure is consistent with the fact that the western North Atlantic and Gulf of Mexico belong to distinct marine ecoregions (Spalding et al. 2007; Moore and Merrick 2011). Due to the paucity of sightings in the western North Atlantic, there are insufficient data to determine whether the western North Atlantic stock comprises multiple demographically independent populations. Additional morphological, acoustic, genetic, and/or behavioral data are needed to further delineate population structure within the western North Atlantic and across the broader geographic area. Because there are confirmed sightings within waters of the Bahamas, this is likely a transboundary stock (e.g., Halpin et al. 2009; Dunn 2013).

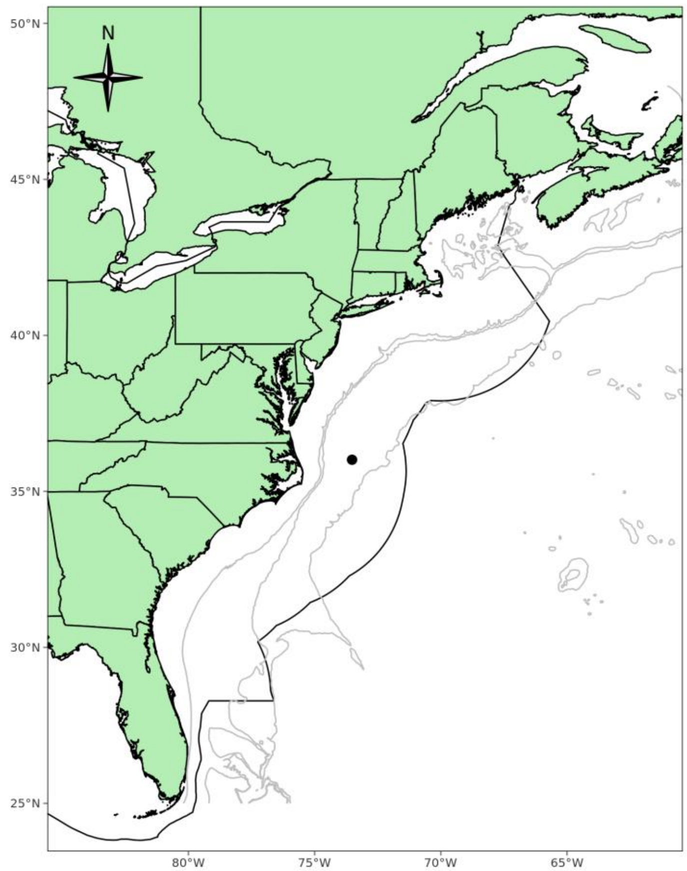

Figure 1. Distribution of a Fraser's dolphin sighting from a SEFSC vessel survey during summer 1999. NEFSC and SEFSC shipboard and aerial surveys were conducted during 1995, 1998, 1999, 2002, 2004, 2006, 2007, 2008, 2010, 2011, 2016, and 2021. Isobaths are the 200-m, 1,000-m, and 4,000-m depth contours. The darker line indicates the U.S. EEZ.

POPULATION SIZE

The numbers of Fraser's dolphins off the U.S. or Canadian Atlantic coast are unknown since it was rarely seen in any surveys. A group of an estimated 250 Fraser's dolphins was sighted in waters 3300 m deep in the western North Atlantic off Cape Hatteras during a 1999 vessel survey (Figure 1; NMFS 1999). Abundances have not been estimated from the 1999 vessel survey in western North Atlantic (NMFS 1999) because the sighting was not made during line- transect sampling effort. Therefore, the population size of Fraser's dolphins is unknown. No Fraser's dolphins have been observed in any other NMFS surveys. However, there has been at least one additional sighting of a Fraser's dolphin off North Carolina (Halpin et al. 2009; McLellan 2014).

Minimum Population Estimate

Present data are insufficient to calculate a minimum population estimate for this stock (Table 1).

Current Population Trend

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

CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

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

POTENTIAL BIOLOGICAL REMOVAL

Potential Biological Removal level (PBR) is the product of the minimum population size, one-half the maximum productivity rate, and a recovery factor (MMPA Sec. 3.16 U.S.C. 1362; Wade and Angliss 1997). The minimum population size is unknown. The maximum productivity rate is 0.04, the default value for cetaceans. The "recovery" factor, which accounts for endangered, depleted, threatened stocks, or stocks of unknown status relative to optimum sustainable population (OSP), is assumed to be 0.5 because this stock is of unknown status. PBR for the western North Atlantic Fraser's dolphin stock is unknown (Table 1).

Table 1. Best and minimum abundance estimates for the western North Atlantic Fraser's dolphin (**Lagenodelphis hosei*****) with Maximum Productivity Rate (R_{max}), Recovery Factor (F_r) and PBR.***

N_{est}	CV N_{est}	N_{min}	F_r	R_{max}	PBR
Unknown	-	Unknown	0.5	0.04	Unknown

ANNUAL HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

Total annual estimated human-caused mortality and serious injury to this stock during 2017–2021 was presumed to be zero, as there were no reports of mortalities or serious injuries to Fraser's dolphins in the western North Atlantic. This species is rare and as a result the likelihood of observing a take is very low. Survey effort and observer effort are insufficient to effectively estimate takes for this species.

Fishery Information

There are two Category I commercial fisheries that interact, or that could potentially interact, with this stock in the Atlantic Ocean. These are theAtlantic Highly Migratory Species longline and the Atlantic Ocean, Caribbean, Gulf of Mexico large pelagics longline fisheries (Appendix III). Percent observer coverage (percentage of sets observed) for these longline fisheries in the Atlantic for each year during 2017–2021 was 11, 10, 10, 9, and 8, respectively.

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

The Atlantic Ocean, Caribbean, Gulf of Mexico large pelagics longline fishery operates in the U.S. Atlantic (including Caribbean) and Gulf of Mexico EEZ, and pelagic swordfish, tunas and billfish are the target species. There were no observed mortalities or serious injuries to Fraser's dolphins by this fishery in the Atlantic Ocean during 2017–2021 (Garrison and Stokes 2020a; 2020b; 2021; 2023a; 2023b).

STATUS OF STOCK

Fraser's dolphins are not listed as threatened or endangered under the Endangered Species Act, and the Western North Atlantic stock is not considered strategic under the Marine Mammal Protection Act. No fishery-related mortality or serious injury has been observed during recent years; however, because this stock is rare, it is unknown whether total fishery-related mortality and serious injury can be considered insignificant and approaching the zero mortality and serious injury rate. The status of Fraser's dolphins in the western U.S. Atlantic EEZ relative to optimum sustainable population is unknown. There are insufficient data to determine the population trends for this species.

OTHER FACTORS THAT MAY BE AFFECTING THE STOCK

Strandings

During 2017–2021, one Fraser's dolphin was reported stranded on the U.S. East Coast (NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 13 October 2022 (Southeast Region) and 18 September 2022 (Northeast Region)). This animal stranded in Florida in 2021, and there was evidence of human interaction (small linear scarring near the mouth/lip region). It should be noted that evidence of human interaction does not necessarily mean the interaction caused the animal's stranding or death.

Stranding data probably underestimate the extent of human and fishery-related mortality and serious injury because not all of the marine mammals that die or are seriously injured in human interactions wash ashore, or, if they do, they are not all recovered (Peltier et al. 2012; Wells et al. 2015; 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

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

The chronic impacts of contaminants (polychlorinated biphenyls [PCBs] and chlorinated pesticides [DDT, DDE, dieldrin, etc.]) on marine mammal reproduction and health are of concern (e.g., Schwacke et al. 2002; Jepson et al. 2016; Hall et al. 2018), but research on contaminant levels for this stock is lacking.

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

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ROUGH-TOOTHED DOLPHIN (*Steno bredanensis*): Western North Atlantic Stock

STOCK DEFINITION AND GEOGRAPHIC RANGE

Rough-toothed dolphins (*Steno bredanensis*) are distributed worldwide in the Atlantic, Pacific and Indian Oceans, generally in warm temperate, subtropical, or tropical waters. They are commonly reported in a wide range of water depths, from shallow, nearshore waters to oceanic waters (West et al. 2011). Most shipboard sightings from the U.S. East Coast have occurred in oceanic waters at depths greater than 1,000 m (Figure 1). Sightings of rough-toothed dolphins along the East Coast of the U.S. are much less common than in the Gulf of Mexico (CETAP 1982; NMFS 1999; Mullin and Fulling 2003). Because there are confirmed sightings within waters of the Bahamas, this is likely a transboundary stock (e.g., Halpin et al. 2009; Dunn 2013).

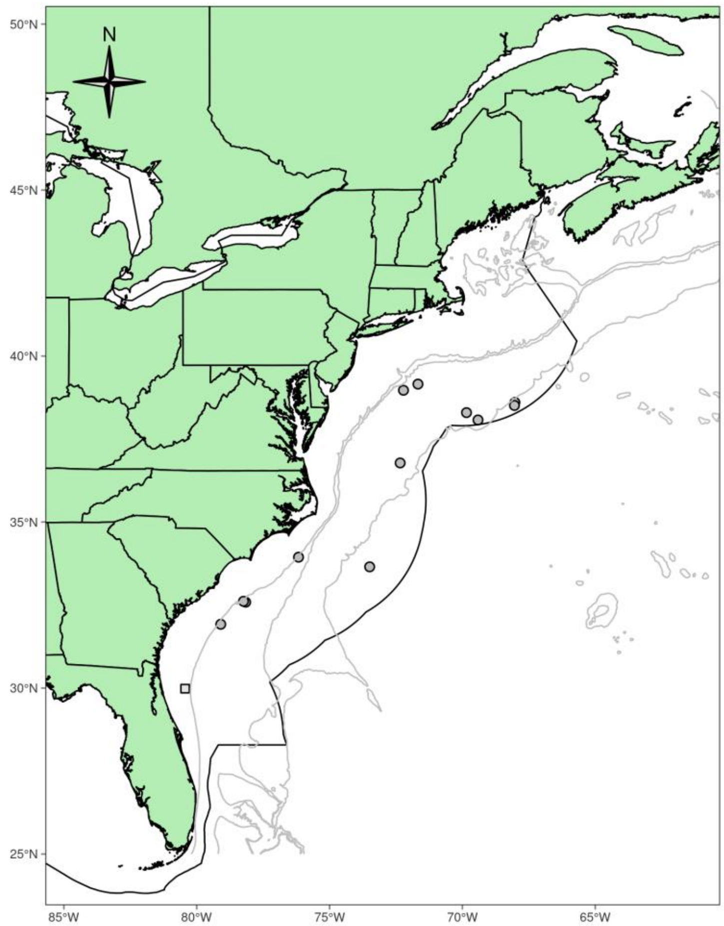

Figure 1. Distribution of rough-toothed dolphin sightings from NEFSC and SEFSC shipboard (circles) and aerial (squares) surveys during 1995, 1998, 1999, 2002, 2004, 2006, 2007, 2008, 2010, 2011, 2016 and 2021. Isobaths are the 200-m, 1,000-m, and 4,000-m depth contours. The darker line indicates the U.S. EEZ.

In the western North Atlantic, tracking of five rough-toothed dolphins that were rehabilitated and released following a mass stranding on the east coast of Florida in 2005, demonstrated a variety of ranging patterns (Wells et al. 2008). All tagged rough-toothed dolphins moved through a large range of water depths averaging greater than 100 m, though each of the five tagged dolphins transited through very shallow waters at some point. These five rough-toothed dolphins moved through waters ranging from 17° to 31°C, with temperatures averaging 21° to 30°C. Recorded dives were rarely deeper than 50 m, with the tagged dolphins staying fairly close to the surface. It is not known how representative of normal species patterns any of these movements are.

Analyses of worldwide genetic differentiation in *Steno* indicate animals in the western Atlantic Ocean are strongly differentiated from those in the Pacific and Indian Oceans (da Silva et al. 2015; Albertson et al. 2022). Albertson et al. (2022) illustrated that this species exhibits population structure within the North Atlantic and da Silva et al. (2015) provided evidence for multiple populations in the western South Atlantic. However, to date there has been no examination of stock structure for this species within the western North Atlantic or the Gulf of Mexico. For management purposes, rough-toothed dolphins observed off the eastern U.S. coast are considered a separate stock from those in the northern Gulf of Mexico. There are insufficient data to determine whether multiple demographically-independent populations exist with the western North Atlantic Stock. Additional morphological, acoustic, genetic and/or behavioral data are needed to further delineate population structure in this region.

POPULATION SIZE

The number of rough-toothed dolphins off the U.S. Atlantic coast is unknown since it has been rarely sighted during surveys. Neither of the two most recent shipboard surveys during summer 2016 and summer 2021, covering waters from central Florida to the lower Bay of Fundy, observed this species (NEFSC and SEFSC 2018; NEFSC and SEFSC 2022). The most recent sightings occurred during 2011 (NMFS 2011). See Appendix IV for a summary of earlier abundance estimates and survey descriptions.

Minimum Population Estimate

Present data are insufficient to calculate a minimum population estimate for this stock (Table 1).

Current Population Trend

A trend analysis cannot be conducted for this stock due to the small number of sightings in any single 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 is unknown. The maximum productivity rate is 0.04, the default value for cetaceans. The “recovery” factor, which accounts for endangered, depleted, threatened stocks, or stocks of unknown status relative to optimum sustainable population (OSP), is assumed to be 0.5 because this stock is of unknown status. PBR for the western North Atlantic stock of rough-toothed dolphins is undetermined (Table 1).

Table 1. Best and minimum abundance estimates for the western North Atlantic rough-toothed dolphin (**Steno bredanensis*****) with Maximum Productivity Rate (R_{max}), Recovery Factor (F_r) and PBR.***

N_{est}	CV N_{est}	N_{min}	F_r	R_{max}	PBR
Unknown	-	Unknown	0.5	0.04	Undetermined

ANNUAL HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

Total annual estimated human-caused mortality and serious injury to this stock during 2017–2021 was presumed to be zero, as there were no reports of mortalities or serious injuries to rough-toothed dolphins in the western North Atlantic. This species is rare and as a result the likelihood of observing a take is very low. Survey effort and observer effort are insufficient to effectively estimate takes for this species.

Fishery Information

There are currently no U.S. fisheries in the western North Atlantic with evidence of interactions that have resulted in incidental mortality or serious injury of rough-toothed dolphins. There has been documented serious injury of rough-toothed dolphins by the Category I large pelagics longline fishery in the northern Gulf of Mexico (Garrison and Stokes 2016). In addition, there has been documented mortality and serious injury of rough-toothed dolphins in the Hawaii shallow-set longline fishery and the American Samoa pelagic longline fishery in the U.S. Pacific (Carretta et al. 2017; Carretta et al. 2018). Rough-toothed dolphins have been taken incidentally in the tuna purse seine nets in the eastern tropical Pacific, and in gillnets off Sri Lanka, Brazil and the offshore North Pacific (Jefferson 2002). A small number of this species are taken in directed fisheries in the Caribbean countries of St. Vincent and the Lesser Antilles, as well as in countries in the Pacific and off Ghana in the eastern north Atlantic Ocean (Northridge 1984; Argones 2001; Jefferson 2002; Reeves et al. 2003).

STATUS OF STOCK

Rough-toothed dolphins are not listed as threatened or endangered under the Endangered Species Act, and the Western North Atlantic stock is not considered strategic under the Marine Mammal Protection Act. No fishery-related mortality or serious injury has been observed during recent years; however, because this stock is rare, it is unknown whether total fishery-related mortality and serious injury can be considered insignificant and approaching the zero mortality and serious injury rate. The status of rough-toothed dolphins in the U.S. EEZ relative to optimum sustainable population is unknown. Given the limited number of sightings of rough-toothed dolphins over the years, the abundance estimate for this stock is unknown and there are insufficient data to determine population trends for this stock. Although there are currently no known habitat issues or other factors causing a decline or impeding recovery, potential sources of human-caused mortality for this stock are poorly understood.

OTHER FACTORS THAT MAY BE AFFECTING THE STOCK

Strandings

Although there have been several mass strandings of rough-toothed dolphins along the U.S. east coast in the past, during 2017–2021 no rough-toothed dolphin strandings were reported (NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 13 October 2022 (Southeast Region) and 18 September 2022 (Northeast Region)).

Habitat Issues

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

The chronic impacts of contaminants (polychlorinated biphenyls [PCBs] and chlorinated pesticides [DDT, DDE, dieldrin, etc.]) on marine mammal reproduction and health are of concern (e.g., Schwacke et al. 2002; Jepson et al. 2016; Hall et al. 2018), but research on contaminant levels for this stock is lacking.

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

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

STOCK DEFINITION AND GEOGRAPHIC RANGE

The Clymene dolphin is endemic to tropical and subtropical waters of the Atlantic Ocean (Jefferson and Curry 2003). Clymene dolphins have been commonly sighted in the Gulf of Mexico since 1990 (Mullin et al. 1994; Fertl et al. 2003). Sightings of this species in the western North Atlantic along the U.S. East Coast are rare (Figure 1); there have generally been only one or two sightings in any given survey year. However, the sightings in addition to stranding records (Fertl et al. 2003) indicate that this species does occur in the western North Atlantic. Nara et al. (2017) analyzed mitochondrial DNA sequence data from samples collected in the western North Atlantic, Gulf of Mexico, and western South Atlantic and found significant genetic differentiation among all three regions, supporting delimitation of separate western North Atlantic and Gulf of Mexico stocks. Given the paucity of sightings, there are insufficient data to determine whether the western North Atlantic stock comprises multiple demographically independent populations. However, those sightings do encompass multiple marine ecoregions (Spalding et al. 2007), and include Cape Hatteras, a known biogeographic break for other marine species, so it is possible that multiple demographically independent populations of *S. clymene* exist within this stock. Additional morphological, acoustic, genetic and/or behavioral data are needed to further delineate population structure in this region.

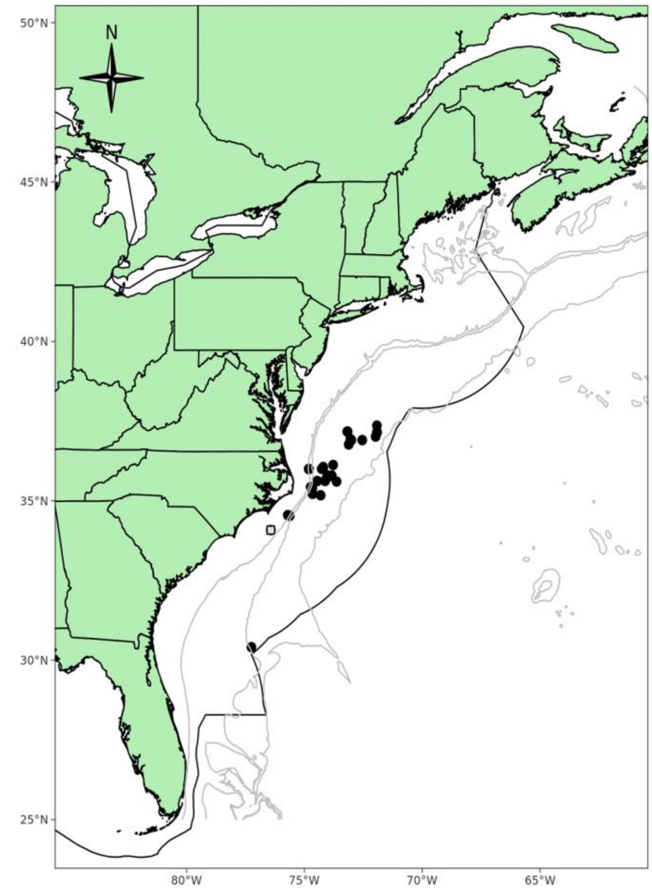

Figure 1. Distribution of Clymene dolphin sightings from NEFSC and SEFSC shipboard (circles) and aerial (squares) surveys during 1995, 1998, 1999, 2002, 2004, 2006, 2007, 2008, 2010, 2011, 2016, and 2021. Isobaths are the 200-m, 1,000-m and 4,000-m depth contours. The darker line indicates the U.S. EEZ.

POPULATION SIZE

The best abundance estimate available for Clymene dolphins in the western North Atlantic is 21,778 (CV=0.72; Garrison and Dias 2023; Palka 2023). This estimate is from summer 2021 surveys covering waters from central Florida to the lower Bay of Fundy. Clymene dolphins were not sighted during surveys of the U.S. Atlantic coast conducted in the summers of 2004 and 2011.

Abundance estimates of 0 and 4,237 (CV=1.03) Clymene dolphins were generated from vessel surveys conducted in U.S. waters of the western North Atlantic during the summer of 2016 (Table 1; Garrison 2020; Palka 2020). One survey was conducted from 27 June to 25 August in waters north of 38ºN latitude and consisted of 5,354 km of on-effort trackline along the shelf break and offshore to the U.S. EEZ (NEFSC and SEFSC 2018). No Clymene dolphins were observed during this survey. Clymene dolphins were observed in the second vessel survey, which covered waters from Central Florida to approximately 38ºN latitude between the 100-m isobaths and the U.S. EEZ during 30 June–19 August. A total of 4,399 km of trackline was covered on effort (NEFSC and SEFSC 2018; Garrison 2020; Palka 2020). Both surveys utilized two visual teams and an independent observer approach to estimate detection probability on the trackline (Laake and Borchers 2004). Mark-recapture distance sampling was used to estimate abundance.

More recent abundance estimates of 2,268 (CV=0.50) and 19,510 (CV=0.80) Clymene dolphins were generated from vessel surveys conducted in U.S. waters of the western North Atlantic during the summer of 2021 (Table 1; Garrison and Dias 2023; Palka 2023). One survey was conducted from 16 June to 23 August in waters north of 36ºN latitude and consisted of 5,871 km of on-effort trackline along the shelf break and offshore to the outer edge of the U.S. EEZ (NEFSC and SEFSC 2022). The second vessel survey covered waters from central Florida (25ºN latitude) to approximately 38ºN latitude between the 200-m isobaths and the outer edge of the U.S. EEZ during 12 June–31 August. A total of 5,659 km of trackline was covered on effort (NEFSC and SEFSC 2022). 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 Clymene dolphin (**Stenella clymene*****) by month, year, and area covered during each abundance survey, and resulting abundance estimate (N_{best}) and coefficient of variation (CV). The estimate considered best is in bold font.***

Month/Year	Area	N_{best}	CV
Jun–Aug 2016	New Jersey to lower Bay of Fundy	0	-
Jun–Aug 2016	Central Florida to New Jersey	4,237	1.03
Jun–Aug 2016	Central Florida to lower Bay of Fundy (COMBINED)	4,237	1.03
Jun–Aug 2021	New Jersey to lower Bay of Fundy	2,268	0.50
Jun–Aug 2021	Central Florida to New Jersey	19,510	0.80
Jun–Aug 2021	**Central Florida to lower Bay of Fundy (COMBINED)**	**21,778**	**0.72**

Minimum Population Estimate

The minimum population estimate is the lower limit of the two-tailed 60% confidence interval of the log- normally distributed best abundance estimate. This is equivalent to the 20th percentile of the log-normal distribution as specified by Wade and Angliss (1997). The best abundance estimate is 21,778 (CV=0.72). The minimum population estimate based on the 2021 abundance estimates is 12,622 (Table 2).

Current Population Trend

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

CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

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

POTENTIAL BIOLOGICAL REMOVAL

Potential Biological Removal (PBR) is the product of minimum population size, one-half the maximum productivity rate, and a recovery factor (MMPA Sec. 3. 16 U.S.C. 1362; Wade and Angliss 1997). The minimum population size for the Clymene dolphin is 12,622. The maximum productivity rate is 0.04, the default value for cetaceans. The recovery factor, which accounts for endangered, depleted, threatened stocks, or stocks of unknown status relative to optimum sustainable population (OSP) is set to 0.5 because this stock is of unknown status. PBR for the western North Atlantic stock of Clymene dolphins is 126 (Table 2).

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

N_{est}	CV N_{est}	N_{min}	F_r	R_{max}	PBR
21,778	0.72	12,622	0.5	0.04	126

ANNUAL HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

Total annual estimated human-caused mortality and serious injury to this stock during 2017–2021 was presumed to be zero, as there were no reports of mortalities or serious injuries to Clymene dolphins in the western North Atlantic. Recorded takes of Clymene dolphins in fisheries in the western North Atlantic are rare. However, observer coverage in the fisheries is relatively low. Furthermore, the likelihood is low that a dolphin killed at sea due to a fishery interaction or vessel-strike will be recovered (Williams et al. 2011). These factors introduce some uncertainty into estimating the true level of human-caused mortality and serious injury for this stock.

Fishery Information

There are two commercial fisheries that interact, or that could potentially interact, with this stock in the Atlantic Ocean. These are the Category I Atlantic Highly Migratory Species longline and the Atlantic Ocean, Caribbean, Gulf of Mexico large pelagics longline fisheries (Appendix III). Percent observer coverage (percentage of sets observed) for these longline fisheries in the Atlantic for each year during 2017–2021 was 11, 10, 10, 9, and 8, respectively.

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

The Atlantic Ocean, Caribbean, Gulf of Mexico large pelagics longline fishery operates in the U.S. Atlantic (including Caribbean) and Gulf of Mexico EEZ, and pelagic swordfish, tunas and billfish are the target species. There were no observed mortalities or serious injuries to Clymene dolphins by this fishery in the Atlantic Ocean during 2017–2021 (Garrison and Stokes 2020a; 2020b; 2021; 2023a; 2023b).

STATUS OF STOCK

Clymene dolphins are not listed as threatened or endangered under the Endangered Species Act, and the Western North Atlantic stock is not considered strategic under the Marine Mammal Protection Act. No fishery-related mortality or serious injury has been observed; therefore, total fishery-related mortality and serious injury can be considered insignificant and approaching the zero mortality and serious injury rate. The status of Clymene dolphins in the U.S. EEZ relative to optimum sustainable population is unknown. There are insufficient data to determine population trends for this stock.

OTHER FACTORS THAT MAY BE AFFECTING THE STOCK

Strandings

During 2017–2021, one Clymene dolphin was reported stranded along the U.S. East Coast (NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 13 October 2022 (Southeast Region) and 18 September 2022 (Northeast Region)). This animal stranded in South Carolina in 2018, and no evidence of human interaction was detected.

Stranding data underestimate the extent of human and fishery-related mortality and serious injury because not all of the marine mammals that die or are seriously injured in human interactions wash ashore, or, if they do, they are not all recovered (Peltier et al. 2012; Wells et al. 2015; 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), and decomposition can also introduce uncertainty in visual species identification of a carcass, particularly for closely related species like those in the genus Stenella. Finally, the level of technical expertise among stranding network personnel varies widely as does the ability to recognize signs of human interaction.

Habitat Issues

The chronic impacts of contaminants (polychlorinated biphenyls [PCBs] and chlorinated pesticides [DDT, DDE, dieldrin, etc.]) on marine mammal reproduction and health are of concern (e.g., Schwacke et al. 2002; Jepson et al.

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

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

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

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

STOCK DEFINITION AND GEOGRAPHIC RANGE

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

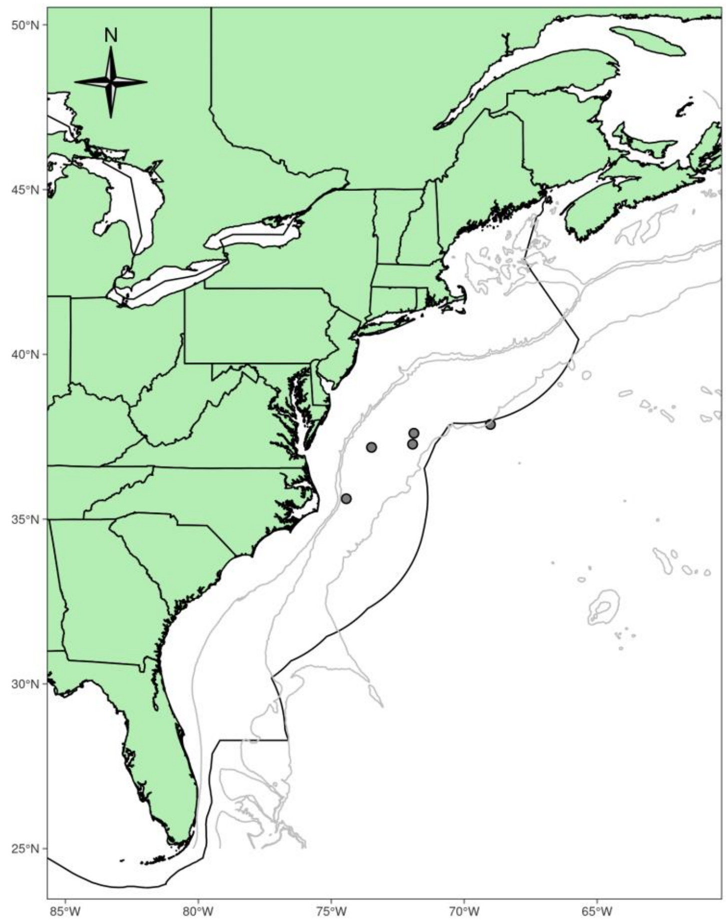

Figure 1. Distribution of spinner dolphin sightings from NEFSC and SEFSC shipboard (circles) and aerial (squares) surveys during 1995, 1998, 1999, 2002, 2004, 2006, 2007, 2008, 2010, 2011, 2016 and 2021. Isobaths are the 200-m, 1,000-m, and 4,000-m depth contours. The darker line indicates the U.S. EEZ.

Spinner dolphins in the western North Atlantic are managed separately from those in the northern Gulf of Mexico. Although there have been no directed studies of the degree of demographic independence between the two areas, this management structure is consistent with evidence for population structure in other areas, including more pelagic waters of the eastern tropical Pacific (Leslie and Morin 2016), and is further supported because the two stocks occupy distinct marine ecoregions (Spalding et al. 2007; Moore and Merrick 2011). Due to the paucity of sightings, there are insufficient data to determine whether the western North Atlantic stock comprises multiple demographically independent populations. Additional morphological, acoustic, genetic, and/or behavioral data are needed to further delineate population structure within the western North Atlantic and across the broader geographic area.

POPULATION SIZE

The best abundance estimate available for spinner dolphins in the western North Atlantic is 3,181 (CV=0.65;

Garrison and Dias 2023; Palka 2023). This estimate is from summer 2021 surveys covering waters from central Florida to the lower Bay of Fundy.

Recent Surveys and Abundance Estimates

Abundance estimates of 160 (CV=0; based on a single sighting) and 3,942 (CV=1.03) spinner dolphins were generated from vessel surveys conducted in U.S. waters of the western North Atlantic during the summer of 2016 (Garrison 2020; Palka 2020). One survey was conducted from 27 June to 25 August in waters north of 38ºN latitude and consisted of 5,354 km of on-effort trackline along the shelf break and offshore to the U.S. EEZ (NEFSC and SEFSC 2018). The second vessel survey covered waters from Central Florida to approximately 38ºN latitude between the 100-m isobaths and the U.S. EEZ during 30 June–19 August. A total of 4,399 km of trackline was covered on effort (NEFSC and SEFSC 2018). Both surveys utilized two visual teams and an independent observer approach to estimate detection probability on the trackline (Laake and Borchers 2004). Mark-recapture distance sampling was used to estimate abundance. Estimates from the two surveys were combined and CVs pooled to produce a species abundance estimate for the stock area.

More recent abundance estimates of 3,181 (CV=0.65) and 0 spinner dolphins were generated from vessel surveys conducted in U.S. waters of the western North Atlantic during the summer of 2021 (Table 1; Garrison and Dias 2023; Palka 2023). One survey was conducted from 16 June to 23 August in waters north of 36ºN latitude and consisted of 5,871 km of on-effort trackline along the shelf break and offshore to the outer edge of the U.S. EEZ (NEFSC and SEFSC 2022). The second vessel survey covered waters from central Florida (25ºN latitude) to approximately 38ºN latitude between the 200-m isobaths and the outer edge of the U.S. EEZ during 12 June–31 August. A total of 5,659 km of trackline was covered on effort (NEFSC and SEFSC 2022). Both surveys utilized two visual teams and an independent observer approach to estimate detection probability on the trackline (Laake and Borchers 2004). Mark-recapture distance sampling was used to estimate abundance. Estimates from the two surveys were combined and CVs pooled to produce a species abundance estimate for the stock area.

Table 1. Summary of abundance estimates for the western North Atlantic spinner dolphin (*Stenella longirostris*) by month, year, and area covered during each abundance survey, and resulting abundance estimate (N_{best}) and coefficient of variation (CV). The estimate considered best is in bold font.

Month/Year	Area	N_{best}	CV
Jun–Aug 2016	New Jersey to Bay of Fundy	160	0
Jun–Aug 2016	Central Florida to New Jersey	3,942	1.03
Jun–Aug 2016	Central Florida to Bay of Fundy (COMBINED)	4,102	0.99
Jun–Aug 2021	New Jersey to lower Bay of Fundy	3,181	0.65
Jun–Aug 2021	Central Florida to New Jersey	0	-
Jun–Aug 2021	**Central Florida to lower Bay of Fundy (COMBINED)**	**3,181**	**0.65**

Minimum Population Estimate

The minimum population estimate is the lower limit of the two-tailed 60% confidence interval of the log-normally distributed best abundance estimate. This is equivalent to the 20th percentile of the log-normal distribution as specified by Wade and Angliss (1997). The best estimate of abundance for spinner dolphins is 3,181 (CV=0.65). The minimum population estimate for spinner dolphins is 1,930 (Table 2).

Current Population Trend

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

CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

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

POTENTIAL BIOLOGICAL REMOVAL

Potential Biological Removal (PBR) is the product of minimum population size, one-half the maximum productivity rate, and a “recovery” factor (MMPA Sec. 3. 16 U.S.C. 1362; Wade and Angliss 1997). The minimum population size is 1,930. The maximum productivity rate is 0.04, the default value for cetaceans. The “recovery” factor, which accounts for endangered, depleted, threatened stocks, or stocks of unknown status, relative to optimum sustainable population (OSP), is assumed to be 0.5 because this stock is of unknown status. PBR for the western North Atlantic spinner dolphin is 19 (Table 2).

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

N_{est}	CV N_{est}	N_{min}	F_r	R_{max}	PBR
3,181	0.65	1,930	0.5	0.04	19

ANNUAL HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

Total annual estimated human-caused mortality and serious injury to this stock during 2017–2021 was presumed to be zero, as there were no reports of mortalities or serious injuries to spinner dolphins in the western North Atlantic. This species is rare and as a result the likelihood of observing a take is very low. Survey effort and observer effort are insufficient to effectively estimate takes for this species.

Fishery Information

There are two commercial fisheries that interact, or that could potentially interact, with this stock in the Atlantic Ocean. These are the Category I Atlantic Highly Migratory Species longline and the Atlantic Ocean, Caribbean, Gulf of Mexico large pelagics longline fisheries (Appendix III). Percent observer coverage (percentage of sets observed) for these longline fisheries in the Atlantic for each year during 2017–2021 was 11, 10, 10, 9, and 8, respectively.

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

The Atlantic Ocean, Caribbean, Gulf of Mexico large pelagics longline fishery operates in the U.S. Atlantic (including Caribbean) and Gulf of Mexico EEZ, and pelagic swordfish, tunas and billfish are the target species. There were no observed mortalities or serious injuries to spinner dolphins by this fishery in the Atlantic Ocean during 2017–2021 (Garrison and Stokes 2020a; 2020b; 2021; 2023a; 2023b).

STATUS OF STOCK

Spinner dolphins are not listed as threatened or endangered under the Endangered Species Act, and the western North Atlantic stock is not considered strategic under the Marine Mammal Protection Act. No fishery-related mortality or serious injury has been observed in recent years; therefore, total fishery-related mortality and serious injury can be considered insignificant and approaching the zero mortality and serious injury rate. The status of this stock relative to optimum sustainable population in the U.S. Atlantic EEZ is unknown. There are insufficient data to determine the population trends for this species.

OTHER FACTORS THAT MAY BE AFFECTING THE STOCK

Strandings

During 2017–2021, one spinner dolphin was reported stranded on the U.S. East Coast, in Florida (in 2017) (NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 13 October 2022 (Southeast Region) and 18 September 2022 (Northeast Region)). No evidence of human interaction was detected.

Stranding data probably underestimate the extent of human and fishery-related mortality and serious injury because not all of the marine mammals that die or are seriously injured in human interactions wash ashore, or, if they do, they are not all recovered (Peltier et al. 2012; Wells et al. 2015; 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), and decomposition can also introduce uncertainty in visual species identification of a carcass, particularly for closely related species like those in the genus Stenella. Finally, the level of technical expertise among stranding network personnel varies widely as does the ability to recognize signs of human interaction.

Habitat Issues

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

The chronic impacts of contaminants (polychlorinated biphenyls [PCBs] and chlorinated pesticides [DDT, DDE, dieldrin, etc.]) on marine mammal reproduction and health are of concern (e.g., Schwacke et al. 2002; Jepson et al. 2016; Hall et al. 2018), but research on contaminant levels for this stock is lacking.

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

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

STOCK DEFINITION AND GEOGRAPHIC RANGE

There are two morphologically and genetically distinct forms of common bottlenose dolphin (Duffield et al. 1983; Mead and Potter 1995; Rosel et al. 2009) described as the coastal and offshore forms in the western North Atlantic (Hersh and Duffield 1990; Mead and Potter 1995; Curry and Smith 1997; Rosel et al. 2009). The two morphotypes are genetically distinct based upon both mitochondrial and nuclear markers (Hoelzel et al. 1998; Rosel et al. 2009). The genetic and morphological differences recently led to the coastal form being described as a new species, *Tursiops erebennus* (Costa et al. 2022).

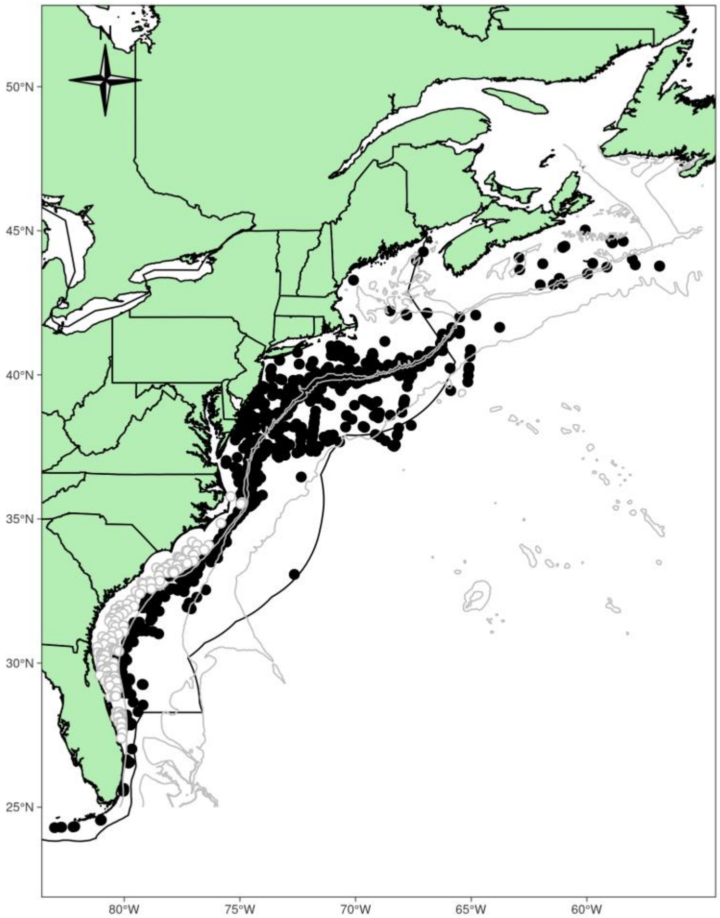

Figure 1. Distribution of offshore common bottlenose dolphin sightings from NEFSC and SEFSC shipboard (circles) and aerial (squares) surveys during 1995, 1998, 1999, 2002, 2004, 2006, 2007, 2008, 2010, 2011, 2016 and 2021. Isobaths are the 200-m, 1,000-m, and 4,000-m depth contours. The darker line indicates the U.S. EEZ. Filled circles represent sightings of the offshore stock. Open circles represent sightings of either the offshore stock or a coastal stock.

The offshore form is distributed primarily along the outer continental shelf and continental slope in the Northwest Atlantic Ocean from Georges Bank to the Florida Keys (Figure 1; CETAP 1982; Kenney 1990), where dolphins with characteristics of the offshore type have stranded. However, common bottlenose dolphins have occasionally been sighted in Canadian waters, on the Scotian Shelf (e.g., Baird et al. 1993; Gowans and Whitehead 1995), and these animals are thought to be of the offshore form. Because there are confirmed sightings within waters of Canada and the Bahamas, this is likely a transboundary stock (e.g., Halpin et al. 2009; Lawson and Gosselin 2009; Dunn 2013; DFO 2017; Emery 2020; Figure 1).

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

Within 7.5 km of shore, all animals were of the coastal morphotype. More recently, offshore morphotype animals have been sampled as close as 7.3 km from shore in water depths of 13 m (Garrison et al. 2003). Systematic biopsy collection surveys were conducted coast-wide during the summer and winter between 2001 and 2005 to evaluate the degree of spatial overlap between the two morphotypes. Over the continental shelf south of Cape Hatteras, North Carolina, the two morphotypes overlap spatially, and the probability of a sampled group being from the offshore morphotype increased with increasing depth based upon a logistic regression analysis (Garrison et al. 2003). Hersh and Duffield (1990) examined common bottlenose dolphins that stranded along the southeast coast of Florida and found four that had hemoglobin profiles matching that of the offshore morphotype. These strandings suggest the offshore form occurs as far south as southern Florida. The range of the offshore common bottlenose dolphin includes waters beyond the continental slope (Kenney 1990), and also waters beyond the U.S. EEZ, and therefore the offshore stock is a transboundary stock (Figure 1). Offshore common bottlenose dolphins may move between the Gulf of Mexico and the Atlantic (Wells et al. 1999).

The western North Atlantic Offshore Stock of common bottlenose dolphins is managed separately from the Gulf of Mexico Oceanic Stock of common bottlenose dolphins. One line of evidence to support this separation comes from Baron et al. (2008), who found that Gulf of Mexico common bottlenose dolphin whistles (collected from oceanic waters) were significantly different from those in the western North Atlantic Ocean (collected from continental shelf and oceanic waters) in duration, number of inflection points and number of steps. In addition, the western North Atlantic and Gulf of Mexico belong to distinct marine ecoregions (Spalding et al. 2007). Restricted genetic exchange has been documented among offshore populations in the Gulf of Mexico (Vollmer and Rosel 2016) but analyses to determine whether multiple demographically independent populations exist within the western North Atlantic have not been performed to date.

POPULATION SIZE

The best available estimate for the offshore stock of common bottlenose dolphins in the western North Atlantic is 64,587 (CV=0.24; Table 1; Garrison and Dias 2023; Palka 2023). This estimate is from summer 2021 surveys covering waters from central Florida to the lower Bay of Fundy.

Earlier Abundance Estimates

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

Recent Surveys and Abundance Estimates

Abundance estimates of 17,958 (CV=0.33; combined northeast vessel and aerial surveys) and 44,893 (CV=0.29; southeast vessel survey) offshore common bottlenose dolphins were generated from surveys conducted in U.S. waters of the western North Atlantic during the summer of 2016 (Table 1; Garrison 2020; Palka 2020). One vessel survey was conducted from 27 June to 25 August in waters north of 38ºN latitude and included 5,354 km of on-effort trackline along the shelf break and offshore to the U.S. EEZ (NEFSC and SEFSC 2018). A concomitant aerial portion was conducted from 14 August to 28 September and included 11,782 km of trackline that were over waters north of New Jersey from the coastline to the 100-m depth contour, throughout the U.S. waters (NEFSC and SEFSC 2018). Estimates from these two surveys were combined to provide an abundance estimate for the area north of 38ºN. The second vessel survey covered waters from central Florida to approximately 38ºN latitude between the 100-m isobaths and the U.S. EEZ during 30 June–19 August. A total of 4,399 km of trackline was covered on effort (NEFSC and SEFSC 2018). All surveys utilized two visual teams and an independent observer approach to estimate detection probability on the trackline (Laake and Borchers 2004). Mark-recapture distance sampling was used to estimate abundance. Estimates from the two surveys were combined and CVs pooled to produce a species abundance estimate for the stock area.

More recent abundance estimates of 37,721 (CV=0.34) and 26,866 (CV=0.34) offshore common bottlenose dolphins were generated from vessel surveys conducted in U.S. waters of the western North Atlantic during the summer of 2021 (Table 1; Garrison and Dias 2023; Palka 2023). One survey was conducted from 16 June to 23 August in waters north of 36ºN latitude and consisted of 5,871 km of on-effort trackline along the shelf break and offshore to the outer edge of the U.S. EEZ (NEFSC and SEFSC 2022). The second vessel survey covered waters from central Florida (25ºN latitude) to approximately 38ºN latitude between the 200-m isobaths and the outer edge of the U.S. EEZ during 12 June–31 August. A total of 5,659 km of trackline was covered on effort (NEFSC and SEFSC 2022). Both surveys utilized two visual teams and an independent observer approach to estimate detection probability on the trackline (Laake and Borchers 2004). Mark-recapture distance sampling was used to estimate abundance. Estimates from the two surveys were combined and CVs pooled to produce an abundance estimate for the stock area.

Table 1. Summary of recent abundance estimates for western North Atlantic offshore stock of common bottlenose dolphins* (Tursiops truncatus truncatus) *by month, year, and area covered during each abundance survey, and resulting abundance estimate (N_{best}) and coefficient of variation (CV). The estimate considered best is in bold font.

Month/Year	Area	N_{best}	CV
Jun–Aug 2016	New Jersey to lower Bay of Fundy	17,958	0.33
Jun–Aug 2016	Central Florida to New Jersey	44,893	0.29
Jun–Aug 2016	Central Florida to lower Bay of Fundy (COMBINED)	62,851	0.23
Jun–Aug 2021	New Jersey to lower Bay of Fundy	37,721	0.34
Jun–Aug 2021	Central Florida to New Jersey	26,866	0.34
Jun–Aug 2021	**Central Florida to lower Bay of Fundy (COMBINED)**	**64,587**	**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 abundance estimate is 64,587 (CV=0.24). The minimum population estimate for western North Atlantic offshore common bottlenose dolphin stock is 52,801 (Table 2).

Current Population Trend

There are four available coastwide abundance estimates for offshore common bottlenose dolphins from the summers of 2004, 2011, 2016, and 2021. Each of these is derived from surveys with similar survey designs and all four used the two-team independent observer approach to estimate abundance. The resulting estimates were 54,739 (CV=0.24) in 2004, 77,532 (CV=0.40) in 2011, 62,851 (CV=0.23) in 2016, and 64,587 (CV=0.24) in 2021 (Garrison 2020; Garrison and Dias 2023; Palka 2020; Palka 2023). A generalized linear model did not indicate a statistically significant (p=0.546) trend in these estimates. The high level of uncertainty in these estimates limits the ability to detect a statistically significant trend. A key uncertainty in this assessment of trend is that interannual variation in abundance may be caused by either changes in spatial distribution associated with environmental variability or changes in the population size of the stock.

CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

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

POTENTIAL BIOLOGICAL REMOVAL

Potential biological removal (PBR) is the product of minimum population size, one-half the maximum productivity rate, and a "recovery" factor (MMPA Sec. 3. 16 U.S.C. 1362; Wade and Angliss 1997). The minimum population size for offshore common bottlenose dolphins is 52,801. The maximum productivity rate is 0.04, the default value for cetaceans. The "recovery" factor is 0.48 because the CV of the average mortality estimate is greater than 0.3 (Wade and Angliss 1997). PBR for the western North Atlantic offshore common bottlenose dolphin is therefore 507 (Table 2).

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

N_{est}	CV N_{est}	N_{min}	F_r	R_{max}	PBR
64,587	0.24	52,801	0.48	0.04	507

ANNUAL HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

The estimated mean annual fishery-related mortality and serious injury of offshore common bottlenose dolphins during 2017–2021 was 28 (CV=0.43; Table 3) incidental to the large pelagics longline, northeast sink gillnet, northeast bottom trawl, and mid-Atlantic bottom trawl commercial fisheries. Additional mean annual mortality and serious injury for offshore common bottlenose dolphins 2017–2021 due to other human-caused sources was presumed to be zero. The minimum total mean annual human-caused mortality and serious injury for offshore common bottlenose dolphins during 2017–2021 was therefore 28. This is considered a minimum because 1) the estimate of fishery-related mortality and serious injury does not include the mid-Atlantic gillnet fishery, and 2) the likelihood is low that a dolphin killed at sea due to a fishery interaction or vessel-strike will be recovered (Williams et al. 2011).

Fisheries Information

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

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

Longline

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

The large pelagics longline fishery operates in the U.S. Atlantic (including Caribbean) and Gulf of Mexico EEZ, and pelagic swordfish, tunas and billfish are the target species. During 2017–2021, there was one observed mortality and three observed serious injuries of common bottlenose dolphins of the offshore stock by this fishery (Garrison and Stokes 2020a; 2020b; 2021; 2023a; 2023b). See Table 3 for bycatch estimates and observed mortality and serious injury for the current 5-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 western North Atlantic offshore common bottlenose dolphins* (Tursiops truncatus truncatus) *by commercial fishery including the years sampled (Years), the type of data used (Data Type), the annual observer coverage (Observer Coverage), the observed mortalities and serious injuries using on-board observer data, the estimated annual mortality and serious injury, the combined annual estimates of mortality and serious injury (Estimated Combined Mortality), the estimated CV of the combined estimates (Estimated CVs) and the mean of the combined estimates (CV in parentheses).

Fishery	Years	Data Type[a]	Observer Coverage[b]	Observed Serious Injury	Observed Mortality	Estimated Serious Injury	Estimated Mortality	Estimated Combined Mortality	Est. CVs	Mean Annual Mortality
Large Pelagics Longline	2017	Obs. Data Logbook	.11	0	0	0	0	0	NA	8.7 (0.50)
	2018		.10	2	0	17.3	0	17.3	0.73	
	2019		.10	0	0	0	0	0	NA	
	2020		.09	1	0	10.2	0	10.2	0.73	
	2021		.08	0	1	0	15.8	15.8	1.00	
Northeast Sink Gillnet[c]	2017	Obs. Data Logbook	.12	0	1	0	8	8	.92	2.3 (3.21)
	2018		.11	0	0	0	0	0	0	
	2019		.12	0	0	0	0	0	0	
	2020		.02	0	0	0	1.9	1.9	0.99	
	2021		.11	0	0	0	1.4	1.4	0.99	
Northeast Bottom Trawl[d]	2017	Obs. Data Logbook	.12	0	0	0	0	0	NA	2.2 (0.56)
	2018		.12	0	0	0	0	0	NA	
	2019		.16	0	1	0	5.6	5.6	0.92	
	2020		.08	0	0	0	1.9	1.9	0.92	
	2021		.19	0	1	0	3.7	3.7	0.86	

Fishery	Years	Data Type[a]	Observer Coverage[b]	Observed Serious Injury	Observed Mortality	Estimated Serious Injury	Estimated Mortality	Estimated Combined Mortality	Est. CVs	Mean Annual Mortality
Mid-Atlantic Bottom Trawl[d]	2017 2018 2019 2020 2021	Obs. Data Logbook	.14 .12 .12 .02 .04	0 0 0 0 0	3 1 0 1 2	0 0 0 0 0	22.1 6.3 0 9.5 37.9	22.1 6.3 0 9.5 37.9	0.66 0.91 NA 0.55 1.03	15.2 (0.56)
TOTAL	2017–2021	-	-	-	-	-	-	-	-	28 (0.43)

[a] Observer data (Obs. Data) are used to measure bycatch rates, and the data are collected within the Northeast Fisheries Observer Program. Mandatory logbook data were used to measure total effort for the longline fishery. These data are collected at the Southeast Fisheries Science Center (SEFSC).
[b] Proportion of sets observed (for Pelagic Longline, in the Atlantic portion of the fishery).
c Observer data in the Northeast sink gillnet fishery in 2020 and 2021 was not used in the bycatch estimation process, because observer coverage was impacted by the COVID-19 pandemic and was not believed to be representative of the fishery in 2020 and 2021. The numbers of observed mortalities and serious injuries are included in the usual columns for the sake of documentation only. Bycatch estimates for 2020 and 2021 were developed using the observed bycatch rate in 2017-2019 and fishing effort in 2020 and 2021 respectively. The CV for the annual mortality over 2017-2021 was calculated using the estimated CVs from 2017-2019 only.
[d] Due to the impact of the COVID-19 pandemic on Northeast and Mid-Atlantic bottom trawl observer coverage, a 3-year average (2017–2019) was used to estimate mortality and serious injury for calendar year 2020. The observed numbers are included in the usual columns for the sake of documentation only. Fishery related bycatch rates for 2017–2019 and 2021 were estimated using an annual stratified ratio-estimator following the methodology described in Chavez-Rosales et al. (2018).

Northeast Sink Gillnet

During 2017–2021, one mortality was observed (in 2017) in the northeast sink gillnet fishery (Orphanides 2020, 2021; Precoda and Orphanides 2022; Precoda 2023). There were no observed injuries of common bottlenose dolphins in the Northeast region during 2017–2021 to assess using new serious injury criteria. 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.

Through the Marine Mammal Authorization Program (MMAP) during 2017–2021, there were four self-reported incidental takes (mortalities) of common bottlenose dolphins off New York (during 2017).

Northeast Bottom Trawl

During 2017–2021, two mortalities were observed in the northeast bottom trawl fishery (Lyssikatos et al. 2020, 2021; Lyssikatos and Chavez-Rosales 2022). There were no observed injuries of common bottlenose dolphins in the northeast region during 2017–2021 to assess using new serious injury criteria. 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.

Through the Marine Mammal Authorization Program (MMAP) during 2017–2021, there was one self-reported incidental take (mortality) of a common bottlenose dolphin off Massachusetts while trawling for *Illex* squid.

Mid-Atlantic Bottom Trawl

During 2017–2021, seven mortalities were observed in the mid-Atlantic bottom trawl fishery (Lyssikatos et al. 2020, 2021; Lyssikatos and Chavez-Rosales 2022). There were no observed injuries of common bottlenose dolphins in the mid-Atlantic region during 2017–2021 to assess using new serious injury criteria. 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.

Mid-Atlantic Gillnet

Through the Marine Mammal Authorization Program (MMAP) during 2017–2021, there was one self-reported incidental take (mortality) of a common bottlenose dolphin off Virginia (during 2019) by a fisherman targeting monkfish.

STATUS OF STOCK

The common bottlenose dolphin in the western North Atlantic is not listed as threatened or endangered under the Endangered Species Act, and the offshore stock is not considered strategic under the MMPA. Total U.S. fishery-related mortality and serious injury for this stock is less than 10% of the calculated PBR and, therefore, can be considered to be insignificant and approaching the zero mortality and serious injury rate. The status of this stock relative to optimum sustainable population in the U.S. Atlantic EEZ is unknown. There was no statistically significant trend in population size for this species; however, the high level of uncertainty in the estimates limits the ability to detect a statistically significant trend.

OTHER FACTORS THAT MAY BE AFFECTING THE STOCK

Strandings

A total of 1,764 common bottlenose dolphins were found stranded along the U.S. East Coast from 2017 through 2021 (NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 13 October 2022 (Southeast Region) and 18 September 2022 (Northeast Region)). Of these, 264 showed evidence of human interactions (e.g., gear entanglement, mutilation, vessel strike). However, none were identified as belonging to the offshore stock. 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. For example, only 19 of 185 *Tursiops* strandings in North Carolina that were genetically tested were assigned to the offshore form (Byrd et al. 2014). Nevertheless, it is possible that some of the stranded common bottlenose dolphins belonged to the offshore stock and that they were among those strandings with evidence of human interactions.

Habitat Issues

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

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

The chronic impacts of contaminants (polychlorinated biphenyls [PCBs] and chlorinated pesticides [DDT, DDE, dieldrin, etc.]) on marine mammal reproduction and health are of concern (e.g., Schwacke et al. 2002; Jepson et al. 2016; Hall et al. 2018), but research on contaminant levels for the offshore stock of bottlenose dolphins is lacking.

Climate-related changes in spatial distribution and abundance, including poleward and depth shifts, have been documented in or predicted for plankton species and commercially important fish stocks (Nye et al. 2009; Pinsky et al. 2013; Poloczanska et al. 2013; Grieve et al. 2017; Morley et al. 2018) and cetacean species (e.g., MacLeod 2009; Sousa et al. 2019). Chavez-Rosales et al. (2022) documented an overall 178 km northeastward spatial distribution shift of the seasonal core habitat of Northwest Atlantic cetaceans that was related to changing habitat/climatic factors. Results varied by season and species. This study used sightings data collected during seasonal aerial and shipboard line transect abundance surveys during 2010 to 2017. During this time frame, the weighted centroid of the offshore common bottlenose dolphin core habitat moved farthest during fall (753 km towards the northeast) and least during winter (211 km). There is uncertainty in how, if at all, the changes in distribution and population size of cetacean species may interact with changes in distribution of prey species and how the ecological shifts will affect human impacts to the species.

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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 (Figure 1). 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 (>1,800 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).

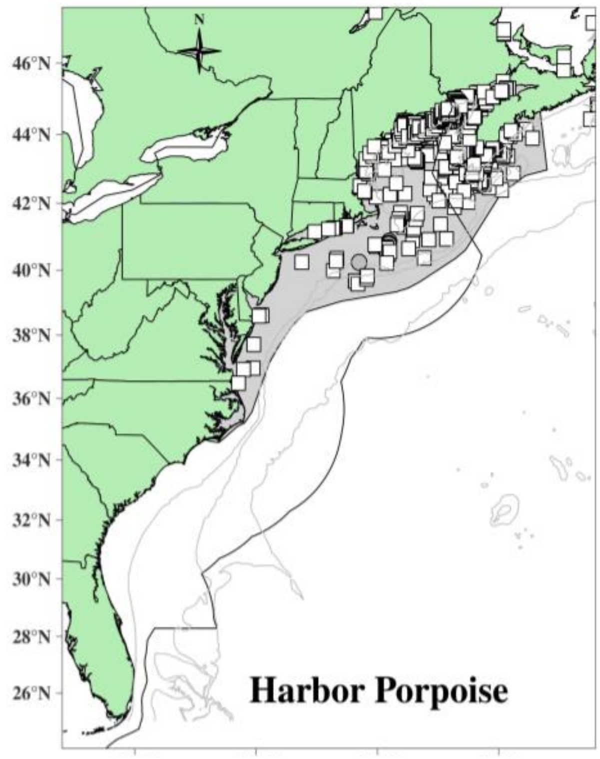

Figure 1. Distribution of harbor porpoises from NEFSC and SEFSC shipboard and aerial surveys during the summers of 1995, 1998, 1999, 2002, 2004, 2006, 2007, 2008, 2010, 2011, 2016, and 2021 and portions of DFO's 2007 TNASS and 2016 NAISS surveys. Circle symbols represent shipboard sightings and squares are aerial sightings. Shaded area represents approximate stock range.

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 was generated from the 2021 NEFSC and SEFSC that covered U.S. and Canadian waters, from Florida to Nova Scotia, Canada surveys: 85,765 (CV=0.53; Table 1; Garrison and Dias 2023; Palka 2023). 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.

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.

A more recent abundance estimate of 85,765 (CV=0.53) harbor porpoises was generated from an aerial survey conducted in U.S. and Canadian waters of the western North Atlantic during the summer of 2021 (Table 1; Garrison and Dias 2023; Palka 2023). The aerial survey was conducted during summer in waters north of 38ºN in the Gulf of Maine to the lower Bay of Fundy and consisted of 5,217 km of on-effort primary tracklines. In addition, two vessel surveys were conducted concurrently covering waters from the Gulf of Maine to Florida with 5,659 km of on-effort track lines. No harbor porpoises were detected during the vessel surveys. All three 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 that was then corrected for availability bias (animals missed due to dive patterns). These surveys missed a small portion of the Gulf of Maine/Bay of Fundy habitat that is on the western part of the Scotian Shelf (about 10% of the known habitat).

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
Jun–Aug 2021	New Jersey to lower Bay of Fundy	85,765	0.53
Jun–Aug 2021	Central Florida to New Jersey	0	-
Jun–Aug 2021	**Central Florida to lower Bay of Fundy (COMBINED)**	**85,765**	**0.53**

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 the Gulf of Maine/Bay of Fundy harbor porpoises is 85,765 (CV=0.53). The minimum population estimate for the Gulf of Maine/Bay of Fundy harbor porpoise is 56,420 (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 56,420. 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 in U.S. and Canadian waters to Nova Scotia is 649(Table 2).

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

N_{est}	CV	N_{min}	F_r	R_{max}	PBR
85,765	0.53	56,420	0.5	0.046	649

ANNUAL HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

The total annual estimated average human-caused mortality and serious injury is 145 harbor porpoises per year (CV=0.18) from U.S. fisheries using observer data and an annual average of 0.2 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*) in U.S. waters.

Years	Source	Annual Avg.	CV
2017–2021	U.S. commercial fisheries using observer data	145	0.18
2017–2021	Non-fishery human caused stranding mortalities	0.2	-
2017–2021	Research takes	0.2	
TOTAL		145.4	-

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 (0.012–0.130) for some times and areas, especially during the COVID-19 pandemic (2020–2022) . 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 2020, 2021; Precoda and Orphanides 2022, Precoda 2023). 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 mortalities 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.

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 2020, 2021; Precoda and Orphanides 2022, Precoda 2023). 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

One harbor porpoise was incidentally killed during research conducted during the NEFSC 2021 Bottom Trawl survey.

Table 4. From observer program data, summary of the incidental mortality of Gulf of Maine/Bay of Fundy harbor porpoise (**Phocoena phocoena phocoena*****) by U.S. 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	2017 2018 2019 2020 2021	Obs. Data, Trip Logbook, Allocated Dealer Data	0.12 0.11 0.12 0.02 0.11	1 0 0 0 0	18 9 33 10 25	7 0 0 2 2	129 92 195 119 109	136 92 195 121 111	0.28 0.52 0.22 0.22 0.19	131(0.19)
Mid-Atlantic Gillnet	2017 2018 2019 2020 2021	Obs. Data, Weighout	0.09 0.09 0.13 0.03 0.01	0 0 0 0 0	1 0 2 2 0	0 0 0 0 0	9.1 0 13 16 10	9.1 0 13 16 10	0.95 0 0.51 0.63 0.65	10 (0.56)
Northeast Bottom Trawl	2017 2018 2019 2020 2021	Obs. Data, Weighout	0.12 0.12 0.16 0.08 0.19	0 0 0 0 0	0 0 2 0 1	0 0 0 0 0	0 0 11 3.6 5.0	0 0 11 3.6 5.0	0 0 0.63 0.63 0.92	3.9 (0.44)
TOTAL										145 (0.18)

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 2017–2021 period and include both at-sea monitor and traditional observer data (Josephson and Lyssikatos 2023).

Canada

Within the habitat of the Gulf of Maine/Bay of Fundy population, nNo current bycatch estimates exist, but harbor porpoise interactions hadve 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 current bycatch is very small, if any (H. Stone, Department of Fisheries and Oceans Canada, pers. comm.).

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 is unknown. Population trends for this species have not been investigated.

OTHER FACTORS THAT MAY BE AFFECTING THE STOCK

Strandings

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 19 October2022). Of the 305 U.S.stranding mortalities reported during this time period, 18 were coded as having signs of human interaction. Of these, 2 were deemed fishery interactions (assumed to be subsumed in the extrapolated fishery bycatch estimates) and 1 was attributed to a vessel strike. Most of the remaining Human Interaction (HI) cases were harassment, unlikely to have contributed to the stranding or post-mortem interactions. However, in 1 case, the non-fishery human interaction was likely to have been a contributing factor 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, 2017–2021.

Area	2017	2018	2019	2020	2021	Total
Maine[a, f]	2	5	8	8	9	40
New Hampshire	0	1	2	0	3	13
Massachusetts[a, b,e, f]	18	8	29	13	14	137
Rhode Island[a, f]	2	2	0	0	0	2
Connecticut	0	0	0	0	0	1
New York[a]	3	1	12	6	0	33
New Jersey[a]	2	5	14	5	1	31
Delaware	0	0	6	0	1	10
Maryland	0	0	2	1	0	9
Virginia	3	2	5	0	0	12
North Carolina	14	1	1	0	0	17
TOTAL U.S.	44	25	79	33	28	305
Nova Scotia/Prince Edward Island[c]	13	16	22	32	37	141
Newfoundland and New Brunswick[d]	2	0	0	0	0	1
GRAND TOTAL	59	41	101	65	65	447

a. 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.
b. Two HI cases in 2018; both in Massachusetts. One was coded as a fishery interaction.
c. Data supplied by Nova Scotia Marine Animal Response Society (pers. comm.)..
d. See Ledwell and Huntington (2018, 2019, 2020, 2021a, 2021b).
e. Three Massachusetts stranding mortalities in 2019 were classified as non-fishery human interaction.
f. Four HI cases in 2020, all of them due to activities by the public post-stranding. In 3 of these cases, the animal was released alive.

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 2018, 2019, 2020, 2021a, 2021b; 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). Chavez-Rosales et al. (2022) documented an overall 178 km northeastward spatial distribution shift of the seasonal core habitat of Northwest Atlantic cetaceans that was related to changing habitat/climatic factors. Results varied by season and species. This study used sightings data collected during seasonal aerial and shipboard line transect abundance surveys during 2010 to 2017. During this time frame, the weighted centroid of harbor porpoise core habitat moved farthest during winter (397 km towards the northeast) and less than 20 km in the other seasons. There is uncertainty in how, if at all, the changes in distribution and population size of cetacean species may interact with changes in distribution of prey species and how the ecological shifts will affect human impacts to the species.

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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 defines the western North Atlantic stock which represents a transboundary stock ranging 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). In the Canadian portion of its range, the Northwest population contains two breeding aggregations: Scotian Shelf (Sable Island and coastal Nova Scotia) and Gulf of St. Lawrence (DFO 2022). Outside of the breeding season, animals from these two breeding aggregations mix with a third breeding aggregation of animals breeding in U.S. waters (Lavigueur and Hammill 1993; Harvey et al. 2008; Breed et al. 2006, 2009), and all three breeding aggregations are considered a single population based on genetic similarity (Boskovic et al. 1996; Wood et al. 2011). The 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).

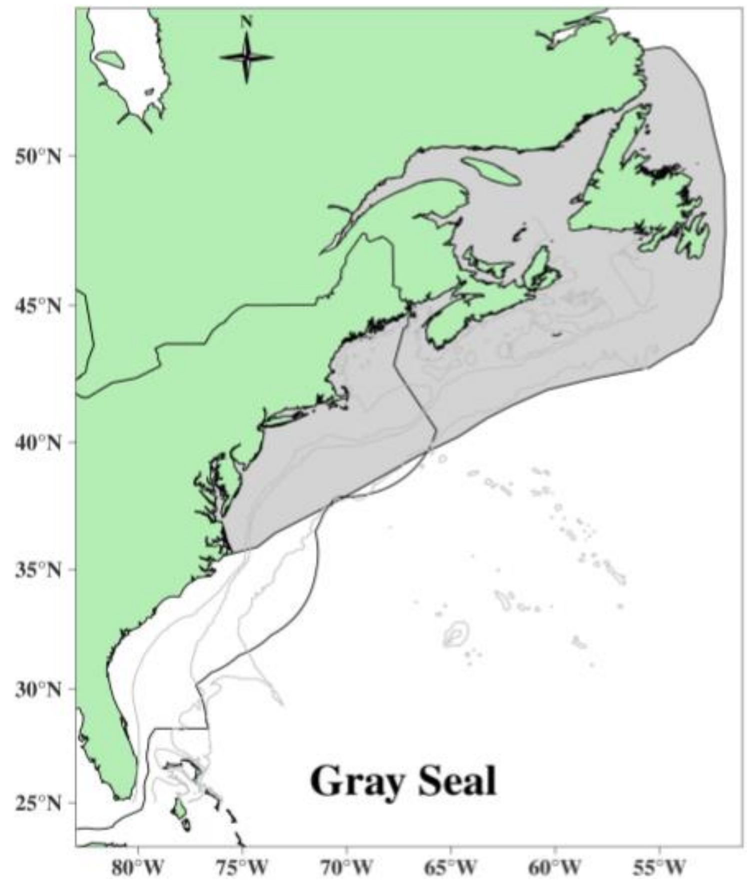

Figure 1. Approximate range of the Western North Atlantic stock of gray seals (**Halichoerus grypus atlantica*****).***

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 metapopulation. 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 (Nowak et al. 2020; Murray et al. 2021). 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 may influence seasonal changes in abundance throughout the range of this transboundary species 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 2021 at breeding colonies in Canada was estimated to be 98,200 pups (95% CI = 86,800 - 109,700; DFO 2022). Production at Gulf of St. Lawrence, and Scotian Shelf colonies accounted for 17%, and 83%, 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 2021 was estimated to be 366,400 (95% CI=317,800 to 409,400; DFO 2022). Uncertainties in the population estimate derive from uncertainties in life history parameters such as mortality rates and sex ratios (DFO 2022).

In U.S. waters, the number of pupping sites has increased from 1 in 1988 to 109 in 2021 and are located in Maine and Massachusetts (Wood et al. 20222020). 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 6,663 pups were born in 2021 at U.S. breeding colonies (Wood et al. 2022), approximately 6% of the total pup production over the entire range of the population (DFO 2022). 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 Nomans Islands) ranged from 11.5% (95%CI: 3.7–19.2%) to 44.1% (95% CI: 28.1–60.2%; Wood et al. 2022). These high rates of increase provide further support that seals are recruiting to some U.S. colonies at various times 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 (4.19:1, based on the ratio of total population to pups in the Canadian portion of the stock in 2016) (Wood et al. 2022). 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). A simple multiplier is used to estimate population size because vital rates (age-specific reproductive rates, survival) necessary for fitting age-structured population models to pup counts are not available for the portion of the population in U.S. waters. The multiplier used assumes the vital rates in Canadian waters are the same as in the U.S.. Using this approach, the population estimate during the pupping season in U.S. waters is 27,911 . There is no coefficient of variation (CV) around the expansion factor, and likewise, the population estimate resulting from the application of the correction factor to the number of pups born. . 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
2016[b]	Gulf of St Lawrence + Nova Scotia Eastern Shore + Sable Island	424,300	263,600–578,300
2016	U.S.	27,300[c]	NA
2021[d]	Gulf of St Lawrence + Scotian Shelf	366,400	317,800 - 409,400
2021[e]	U.S.	27,911[d]	NA

a. These are model-based estimates derived from pup surveys.
b. DFO 2017
c. This is derived from total population size to pup ratios in Canada, applied to U.S. pup counts.
d. DFO 2022
e. Wood et al. 2022

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 20^{th} percentile of the log-normal distribution as specified by Wade and Angliss (1997). When the variance around the expansion factor is unknown, a default CV of 0.20 is recommended for calculating a minimum population estimate rather than assuming zero variance in the correction factor (Wade and Angliss 1997), and was used to calculate Nmin for the U.S. Based on an estimated U.S. population in 2021 of 27,911 , the minimum population estimate in U.S. waters is 23,624 and 359,332 in Canada, for a total Nmin of 376,621 (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, the U.S.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 estimated mean rate of increase in the minimum number of pups born was 20.9% on Muskeget Island from 1988–2021, 19.9% on Monomoy Island from 2009–2021, 44.1% on Nomans Island from 2011–2021, and 11.5% on Seal Island from 2000–2021 (Wood et al. 2022). These increases only reflect increases in pupping and as such are not an accurate or precise measure of total population growth. The latter is also influenced by juvenile and adult survival, as well by immigration from Canadian waters.

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). The 2021 survey marked the first time in 60 years that the estimate of pup production had decreased on Sable Island, though total pup production in the Gulf and Scotian Shelf was not significantly different than in 2016 (den Heyer et al. 2022). 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. Based on the most recent assessment of animals in Canada, the population increased at a rate of 1.5% per year between 2016 and 2021 (DFO 2022) .

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). An adjusted PBR is reported for managing gray seals in U.S. waters (NMFS 2023) because information on fisheries mortality in Canada is unknown (DFO 2022), and some portion of the Canadian animals likely never enters U.S. waters (O'Boyle and Sinclair 2012). The adjusted PBR is based on the portion of the minimum total stock size estimated to be in U.S. waters (Table 2). The minimum population size for the portion of the stock residing in U.S. waters is 23,624. The maximum productivity rate is 0.128. The recovery factor (Fr) 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 1,512 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 a stock-wide and U.S. apportioned PBR.***

Area	N_{est}	CV	N_{min}	F_r	R_{max}	PBR
U.S.	27,911	0.20 (default)	23,624			1,512
Canada	366,400	0.06	349,332			22,592
Total	394,311	0.05	376,621	1	0.128	24,104

ANNUAL HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

For the period 2017–2021, the average annual estimated human-caused mortality and serious injury to gray seals in the U.S. was 1,388 and for Canada was 3,182, not including the unknown mortality from commercial fisheries, resulting in a total minimum estimate of 4,570 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 Est. Avg.
2017–2021	U.S. commercial fisheries using observer data	1,348
2017–2021	U.S. commercial fisheries using stranding data (serious injuries)	24 (minimum count)
2017–2021	U.S. non-fishery human-caused stranding mortalities and serious injuries	14
2017–2021	U.S. research mortalities	2.0
	U.S. Total	1,388
2017–2021	Canadian commercial harvest	1018
2017–2021	DFO Canada scientific collections	84
2017–2021	Canadian removals of nuisance animals	2080
2017–2021	Canadian commercial fisheries bycatch	Unknown
U.S. and Canadian TOTAL		4,570

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 2023); therefore, rates do not reflect the number of live animals that may have broken free of the gear, but remain entangled. Counts of live animals living with entanglements can be informed by strandings data or research studies. For example, at least 24 live seals were observed entangled in monofilament net on Cape Cod in a study where 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) (Table 6). 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

Annual mortalities were estimated using annual stratified ratio-estimator methods *(*Orphanides 2020, 2021; Precoda and Orphanides 2022; Precoda 2023). 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

Annual 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 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 Gillnet

Annual mortalities were estimated using annual stratified ratio-estimator methods (Orphanides 2020, 2021; Precoda and Orphanides 2022; Precoda 2023). 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

No mortalities have been observed in this fishery, during the current 5-year period, however, 1 gray seal was captured and released alive in 2018 (Josephson 2023).

Northeast Mid-Water and Pair Trawl

Only 1 gray seal was observed in these fisheries from 2017–2021 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 mortality and serious injury of gray seals (**Halichoerus grypus atlantica*****) by U.S. commercial fishery including the years sampled, the type of data used (Data Type), the annual observer coverage (Observer Coverage), mortalityrecorded 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 Est. Mortality
Northeast Sink Gillnet	2017 2018 2019 2020 2021	Obs. Data, Weighout, Trip Logbook	0.12 0.11 0.13 0.02 0.11	0 0 0 0 0	158 103 251 14 48	0 0 0 0 0	930 1113 2019 1357 1027	930 1113 2014 1357 1027	0.16 0.32 0.17 0.14 0.14	1,289 (0.13)
Mid-Atlantic Gillnet	2017 2018 2019 2020 2021	Obs. Data, Trip Logbook, Allocated Dealer Data	0.09 0.09 0.013 0.03 0.01	0 0 0 0 0	0 0 3 0 0	0 0 0 0 0	0 0 18 9 7	0 0 18 9 7	0 0 0.40 0.72 0.69	7(1.07)
Northeast Bottom Trawl[c,d]	2017 2018 2019 2020 2021	Obs. Data, Trip Logbook	0.12 0.12 0.16 0.08 0.19	0 0 0 0 0	2 5 6 7 2	0 0 0 0 0	16 32 30 26 7.5	16 32 30 26 7.5	0.24 0.42 0.37 0.26 0.60	22 (0.18)

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 Est. Mortality
Mid-Atlantic Bottom Trawl	2017	Obs. Data, Trip Logbook	0.14	0	5	0	26	26	0.40	28 (0.27)
	2018		0.12	0	7	0	56	56	0.58	
	2019		0.12	0	3	0	22	22	0.53	
	2020		0.02	0	1	0	35	35	0.35	
	2021		0.04	0	0	0	0	0	0	
Northeast Mid-water Trawl – Incl.Pair Trawl	2017	Obs. Data, Trip Logbook	0.16	0	0	0	0	0	0	0.2 (na)[d]
	2018		0.14	0	1	0	na	na	na	
	2019		0.28	0	0	0	0	0	0	
	2020		0.13	0	0	0	0	0	0	
	2021		0.36	0	1	0	0	0	0	
TOTAL										1348 (0.12)

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 2017–2020 period (Josephson et al. 2023). Due to the impact of the COVID-19 pandemic on observer coverage, a 3-year average (2017–2019) was used to estimate mortality and serious injury for the calendar year 2020. The observed numbers are included in the usual columns for the sake of documentation only.
d. No estimate made. Raw counts provided. Fishery related bycatch rates for 2017–2021 were estimated using an annual stratified ratio-estimator following the methodology described in Chavez-Rosales et al. (2018).

Research Takes

From 2017–2021 there were 2 gray seal mortalities which occurred incidental to research activities under MMPA/ESA permits: 1 in 2017 and 1 in 2020. No gray seal mortalities or serious injuries were reported during this period through the Protected Species Incidental Take database, which covers incidentally captured protected species in NMFS fisheries research surveys including those funded and directed by NMFS and includes partner surveys.

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.

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 2017–2021 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 U.S. fishery-related mortality and serious injury for this stock is not less than 10% of the calculated PBR for U.S. waters 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.

OTHER FACTORS THAT MAY BE AFFECTING THE STOCK

Strandings

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 mortalities caused by human interactions include boat strikes, power plant entrainment, oil spill/exposure, harassment, and shooting. Table 5 presents summaries of gray seal strandings as reported to the NOAA National Marine Mammal Health and Stranding Response Database (accessed 15 October2022). Most stranding mortalities were in Massachusetts, which is the center of gray seal abundance in U.S. waters. Stranding data are effort-dependent and opportunistic, and represent only a fraction of both natural and anthropogenic mortality. 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. Stranding mortalities that are attributed to fishery interactions overlap with the modeled analysis of fishery bycatch based on NMFS observer coverage and so, while included in Table 5 below and summarized in Table 6, are not added to the total annual estimated human-caused mortality presented in Table 3.

In addition to stranding mortalities, there are live stranded animals with serious injuries from human interaction. Table 6 presents a summary of these live animals as reported to the NOAA National Marine Mammal Health and Stranding Response Database (accessed 09 September 2023) A serious injury is defined as an injury that has a >50% chance of resulting in a mortality (NMFS 2023), and includes animals with characteristics such as gear constrictions or with the potential to constrict, ingested gear or hooks, and visible fractures. Data on animals living with serious injuries are effort dependent and opportunistic, and may or may not be attributable to a source. Counts of serious injuries currently represent a minimum and may also include repeat sightings of the same individual; these counts could be improved, and potentially expanded from estimated entanglement rates, with systematic surveys, standardized reporting, and a system to uniquely identify individual seals.

An Unusual Mortality Event (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, over 3,000 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 (2017-2021) with subtotals of animals recorded as pups in parentheses.***

State	2017	2018	2019	2020	2021	Total
Maine	14 (1)	25 (0)	15 (0)	24 (2)	12 (3)	90(6)
New Hampshire	3 (0)	9 (3)	5 (0)	5 (0)	3 (1)	25 (4)
Massachusetts	135 (21)	261 (29)	260 (80)	199 (25)	164 (16)	1019 (171)
Rhode Island	16 (5)	20 (3)	28 (8)	19 (0)	3 (2)	86 (18)
Connecticut	3 (0)	1 (0)	0	1 (0)	0	5 (0)
New York	16 (0)	25 (1)	43 (4)	30 (4)	3 (1)	117(10)
New Jersey	4 (3)	14 (10)	9 (8)	5 (4)	3 (3)	30(24)
Delaware	1 (0)	4 (2)	2 (1)	1 (0)	2 (1)	10 (4)
Maryland	0	1 (1)	0	0	0	1 (1)
Virginia	0	1 (1)	0	0	1 (0)	2(1)
North Carolina	0	5 (2)	0	1 (0)	0	6(2)

State	2017	2018	2019	2020	2021	Total
Total	233 (30)	366 (52)	362 (101)	285 (35)	191 (27)	1396(245)
Unspecified seals (all states)	86	92	80	45	31	334

Table 6. Documented gray seal (**Halichoerus grypus atlantica*****) human-interaction related stranding mortalities and serious injuries along the U.S. Atlantic coast (2017–2021) by type of interaction.***

	Type Cause	2017	2018	2019	2020	2021	Total
Mortalities	Fishery Interaction[a]	10	10	8	4	3	35
	Boat Strike	4	2	1	2	0	9
	Shot	0	0	0	1	0	1
	Human Interaction - Other	3	9	13	2	1	28
Serious Injuries	Fishery interaction	41[b]	35	24	9	9	118
	Disentangled and released[c]	7	7	25	22	13	74
	Human interaction - other	2	11	5	8	5	31
TOTAL		67	74	76	48	31	296

[a]Fishery interaction mortalities are not added to the total annual estimated human-caused mortality presented in Table 3 because they are subsumed in the total estimated mortality calculated from observer data.
[b]Includes 24 observed interactions from Iruzun Martins et al. 2019.
[c]Injuries on animals that have been disentangled and released are considered to not be serious. These animals are not included in Table 3.

Canada

Between 2017–2021, the average annual human-caused mortality and serious injury to gray seals in Canadian waters from commercial harvest is 1,018, 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: 1,421 in 2017, 64 in 2018, 1,236 in 2019, 2,219 in 2020, and 240 in 2021(DFO 2022). In addition, between 2017 and 2021, an average of 2,080 nuisance animals per year were killed. This included 3,368 in 2017, 3,462 in 2018, 3,571 in 2019, (DFO 2017), 0 in 2020, and 0in 2021, based on the total number of licenses that were issued. Lastly, DFO took 90 animals in 2017, 61 animals in 2018, 66 animals in 2019, 127 animals in 2020, and 75 animals in 2021 for scientific collections, for an annual average of 84 animals (DFO 2022).

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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 32 bay, sound and estuary stocks of common bottlenose dolphins in the Gulf of Mexico. Until this effort is completed and 32 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

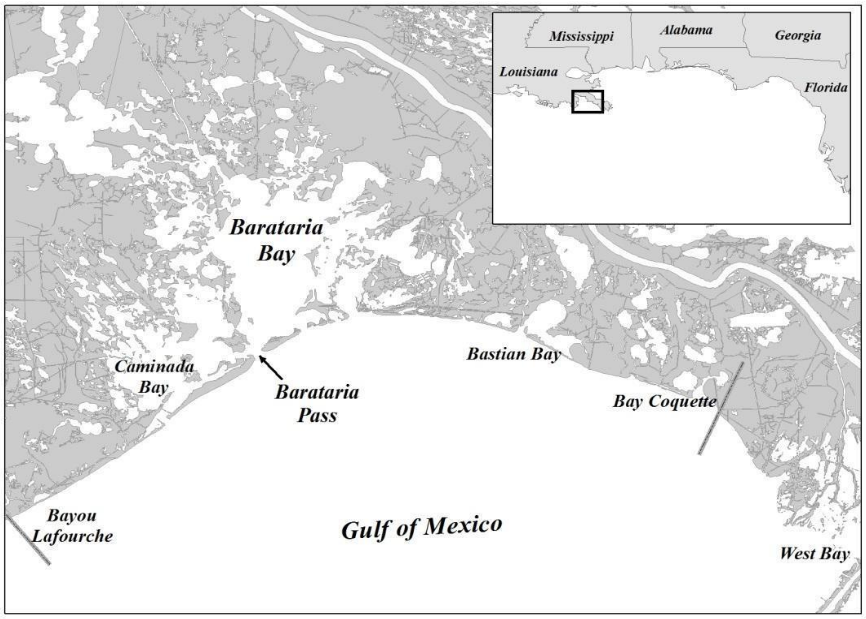

Figure 1. Geographic extent of the Barataria Bay Estuarine System Stock, located on the coast of Louisiana. The borders are denoted by solid lines.

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).

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 analyses also suggested that there may be further partitioning within Barataria Bay (Rosel et al. 2017; Speakman et al. 2022) 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 2017–2021, 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}	CV N_{est}	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 2017–2021 is unknown. Across Louisiana BSE stocks (from Sabine Lake east to Barataria Bay), the most recent (2015–2019) 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 2017–2021 for strandings and at-sea observations identified as fishery-related (hook and line gear) was 0.2. Additional mean annual mortality and serious injury during 2017–2021 due to other human-caused sources (DWH oil spill and unidentified fishing gear) was 34.8. The minimum total mean annual human-caused mortality and serious injury for this stock during 2017–2021 was therefore 35 (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

The most recent mortality estimates for the commercial shrimp trawl fishery are for the years 2015–2019. 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).

Menhaden Purse Seine

During 2017–2021 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 2017–2021 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 2017–2021, three interactions with hook and line gear were documented within the stranding database (NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 13 October 2022; 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 (serious injury averted; 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. In 2021, hook and line gear entanglement was documented for an additional mortality, but available evidence suggested the hook and line gear interaction did not contribute to cause of death. None of the three interactions documented within the stranding data were included in the annual human-caused mortality and serious injury total for this stock (Table 2).

In addition to the interactions documented within the stranding data, during 2017–2021, there was one at-sea observation (in 2019) in Barataria Bay of a dolphin entangled in a “rat’s nest” of monofilament line that was trailing behind the dolphin. This animal was considered seriously injured (Maze-Foley and Garrison 2022) and 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 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

Based on data collected during 2010–2015, a population model was developed to estimate long-term injury to stocks affected by the DWH oil spill (see Population Health Issues Related to the DWH Oil Spill 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). Based on additional data collected during 2016–2019, a more recent model predicted the BBES Stock experienced a 45% (95% CI: 14–74) maximum reduction in population size, and for the years 2017–2021, the model projected 172 mortalities (Table 2; Schwacke et al. 2022). This newer 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. Formal expert elicitation was used to address uncertainty in two parameters for which there were no empirical data, the proportion of the population that would eventually recover to baseline survival and the density-dependent fecundity function. Postspill survival was the greatest contributor to uncertainty in model outputs (Schwacke et al. 2022).

Two common bottlenose dolphins stranded (during 2020 and 2021) with evidence of fishery interactions, and the interactions were believed to contribute to the strandings and deaths of the animals. However, the mortalities could not be attributed to a specific type of gear or fishery (NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 13 October 2022). The two mortalities were included within the annual human-caused mortality and serious injury total for this stock.

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 (2 instead of 1) 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	2017–2021	Pilot Observer Program (2011); MMAP fisherman self-reported takes	NA	0
Atlantic Blue Crab Trap/Pot	2017–2021	Stranding Data	NA	0
Hook and Line	2017–2021	Stranding Data and At-Sea Observations	NA	1*
Mean Annual Mortality due to commercial fisheries (2017–2021)			**0.2**	
Mortality due to DWH (5-year Projection)			172	
Unidentified fishing gear (5-year count)			2	
Mean Annual Mortality due to DWH and unidentified fishing gear (2017–2021)			**34.8**	
Minimum Total Mean Annual Human-Caused Mortality and Serious Injury (2017–2021)			**35**	

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 2017–2021 was 35. 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 Other Mortality 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; 2021) and others indicate continued 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. A recently refined model by Schwacke et al. (2022), based on a decade of data collected following the DWH oil spill, validated the original DWH damage assessment (DWH MMIQT 2015; Schwacke et al. 2017), and projected that the BBES stock experienced a 45% (95% CI: 14–74) maximum reduction in population size, and that it will take 35 years for the stock to recover to 95% of baseline numbers (Schwacke et al. 2022). It is therefore likely that this stock is below its optimum sustainable population (NMFS 2016). In addition, results of modeling work by Thomas et al. (2022) predict there will be greater declines in population size resulting from the mid-Barataria sediment diversion than those caused by the DWH oil spill, which would result in a catastrophic decline and functional extinction of the BBES Stock of common bottlenose dolphins. 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.

OTHER FACTORS THAT MAY BE AFFECTING THE STOCK

Strandings and Unusual Mortality Events

During 2017–2021, 144 common bottlenose dolphins were reported stranded within the BBES area (Table 3; NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 13 October 2022). There was evidence of human interaction (HI) for 19 of the strandings. No evidence of human interaction was detected for 9 strandings, and for the remaining 116 strandings, it could not be determined if there was evidence of human interaction. Human interactions were from numerous sources, including three entanglements with hook and line gear, two mortalities with evidence of gunshot wounds, one animal with evidence of a vessel strike, and two interactions with unidentified gear (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 144 strandings ascribed to the BBES Stock, 29 were ascribed solely to this stock. It is possible, 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 2017 to 2021, 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	2017	2018	2019	2020	2021	Total
Barataria Bay Estuarine System Stock	Total Stranded	35	16	34[b]	24	35	144
	Human Interaction						
	---Yes	2[a]	2	3[c]	5[d]	7[e]	19
	---No	4	1	1	1	2	9
	---CBD	29	13	30	18	26	116

a. Fisheries interactions, both of which were entanglement interactions with hook and line gear (1 mortality and 1 animal released alive without serious injury following mitigation (1 serious injury averted)).
b. 33 strandings were part of the UME event in the northern Gulf of Mexico.
c. Includes 1 animal with evidence of gunshot wounds (mortality).
d. Includes 1 animal with evidence of a vessel strike, 1 animal with evidence of gunshot wounds, and 1 entanglement interaction with unidentified fishing gear (all mortalities).
e. Includes 6 fisheries interactions, 2 of which were believed to have contributed to the cause of death.

Population Health 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 and eight years after the DWH oil spill (Smith et al. 2017; Smith et al. 2022). Immune systems were weakened due to the DWH oil exposure, most noticeably in 2011 compared to subsequent years (De Guise et al. 2017), and immune systems impairments similar to those from 2011 were still present in 2017 and 2018 (De Guise et al. 2021).Health assessment data collected during 2016–2018 by Schwacke et al. (2022) indicated that disease conditions have persisted and worsened in Barataria Bay dolphins presumably exposed to oil from DWH, and that the population declined by 45% compared to pre-spill abundance. The authors suggested the population is at a minimum, vulnerable point in its recovery trajectory.

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).

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). Recently, the final environmental impact statement for a proposed mid-Barataria sediment diversion (MBSD) project has been completed (USACE 2022). This project will divert substantial amounts of freshwater into the Barataria Basin in an effort to reduce wetland loss. Schwacke et al. (2022) cautioned that the MBSD project is likely to be detrimental to population survival for the common bottlenose dolphin stock in Barataria Bay. In addition, results of modeling work by Thomas et al. (2022) predict there will be greater declines in population size resulting from the MBSD than those caused by the DWH oil spill, which would result in a catastrophic decline and functional extinction of the BBES Stock of common bottlenose dolphins.

Additional Human Interactions

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 2017–2021, for two mortalities (2019, 2020), gunshot pellets were found during the necropsies. However, in both instances, the gunshot was not believed to be the cause of death. Both animals were included in the stranding database (NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 13 October 2022) and in the totals presented in Table 3, but were 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 and 2020 shootings 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; Powell et al. 2018). 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; Powell et al. 2018). To date, there are no records within the literature of provisioning for this stock area.

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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).

Note: Stocks not updated for this 2023 report, are included with the most recent year reported.

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	2017-2021	.12, .11, .12, .02, .11	7, 0, 0, 2, 2	129 (.28), 92 (.52), 195(.23), 119 (.22), 109 (.19)	131 (.19)	649
Common Dolphin	2017-2021	.12, .11, .12, .02, .11	0, 0, 0, 0, 0	133 (.28), 93 (.45), 5 (.68), 50 (.25), 39 (.24)	64 (.12)	599
Risso’s Dolphin	2017-2021	.12, .11, .12, .02, .11	0, 0, 0, 0, 0	0, 0, 5 (.7), 2 (1.01), 3 (0)	2.0 (1.8)	307
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	2017-2021	.12, .11, .12, .02, .11	0, 0, 0, 0, 0	930 (.16), 1113 (.32), 2019 (.17), 1357 (.14), 1027 (.14)	1289 (.13)	1,512
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
Atlantic White-sided Dolphin	2017=2021	.12, .11, .12, .02, .11	0, 0, 0, 0, 0	0, 0, 0. 0, 2 (NA)	.2 (NA)	544
Gillnet Fisheries: US Mid-Atlantic Gillnet						
Harbor Porpoise	2017-2021	.09, .09, .13, .03, .01	0, 0, 0, 0, 0	9 (.95), 0, 13 (.51), 16 (.63), 10 (.65)	10(.56)	649
Common Dolphin	2017-2021	.09, .09, .13, .03, .01	11, 1, 0, 5, 4	11 (.71), 8 (.91), 20 (.56), 25 (.55), 20 (.33)	21 (.33)	599
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	2017-2021	.09, .09, .13, .03, .01	0, 0, 0, 0, 0	0, 0, 18 (.40), 9 (.72), 7(.69)	7.0(1.07)	1,512
Longline Fisheries: Pelagic Longline (Excluding NED-E)						
Risso's Dolphin	2017-2021	.12, .10, .10, .09, .08	0.2 (1), 0.2 (.94), 0, 12 (.71), 0	0, 0, 0, 0, 0	2.5 (.68)	307
Short-finned Pilot Whale	2017-2021	.12, .10, .10, .09, .08	133 (.29), 102 (.39), 131 (.37), 371 (.45), 332 (.31)	0, 0, 0, 0, 23	218 (.19)	143

Category, Fishery, Species	Years Observed	Observer Coverage	Est. SI by Year (CV)	Est. Mortality by Year (CV)	Mean Annual Mortality (CV)	PBR
Long-finned Pilot Whale	2017-2021	.12, .10, .10, .09, .08	3.3 (.98), 0.4 (.93), 0.4 (1), 5.7 (.44), 2.8 (.67)	0, 0, 0, 0, 0	2.5 (.36)	306
Common Dolphin	2017-2021	.12, .10. .10, .09, .08	4.9 (1), 1.4 (1), 0, 0, 0	0, 0, 0, 0, 0	1.3 (.81)	599
CATEGORY II						
Trawl Fisheries: Northeast Bottom Trawl						
Harp Seal	2015-2019	.19, .12, .12, .12, 16	0, 0, 0, 0, 0	0, 0, 0, 0, 5.4 (.89)	1.1 (.89)	unk
Harbor Seal	2015-2019	.19, .12, .12, .12, 16	0, 0, 0, 0, 0	0, 0, 8.3 (.96), 0, 5.4 (.88)	2.7 (.68)	2,006
Gray Seal	2017-2021	.12, .12, 16, .08, .19	0, 0, 0, 0, 0	16 (.24), 32 (.42), 30 (.37), 26 (.26), 7.5 (.6)	22 (.18)	1,512
Risso's Dolphin	2017-2021	.12, .12, 16	0, 0, 0, 0, 0	0, 0, 0, 0 3.8).88)	.75 (.88)	307
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	2017-2021	.12, .12, 16, .08, .19	0, 0, 0, 0, 0	0, 0, 5.4 (.88), 1.8(.88), 7.5 (.62)	2.9 (.46)	306
Common Dolphin	2017-2021	.12, .12, 16, .08, .19	0, 0, 0, 0, 0	0, 28(.54), 10 (.62), 50 (.25), 38 (.42)	64 (.18)	599
Atlantic White-sided Dolphin	2017-2021	.12, .12, 16, .08, .19	7.4, 0, 0 5.1	15(.64), 0, 79 (.28)	27 (.21)	544
Harbor Porpoise	2017-2021	.12, .12, 16, .08, .19	0, 0, 0, 0, 0	0, 0, 11 (.63), 3.6 (.63), 5.0 (.92)	3.9 (.44)	649
Mid-Atlantic Bottom Trawl						
Common Dolphin	2017-2021	.14, .12, .12, .02, .04	0, 5, 15, 7, 0	380 (.23), 205 (.54), 395 (.23), 237 (.14), 230 (.57)	309 (.13)	599
Risso's Dolphin	2017-2021	.14, .12, .12, .02, .04	12, 0, 0, 4, 0	31 (.51), 0, 0, 14 (.51), 0	12 (.39)	307
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
Gray Seal	2017-2021	.14, .12, .12, .02, .04	0, 0, 0, 0, 0	26 (.40), 56 (.58), 22 (.53), 35 (.35), 0	28(.27)	1,512
Northeast Mid-water Trawl (Including Pair Trawl)						
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	2017-2021	..16, .14, .28, .13, .36	0, 0, 0, 0, 0	0, .2 (na), 0, 0, 0	0.2 (na)	1,512

Appendix II: Summary of the confirmed observed 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, 2017–2021, with number of events attributed to entanglements or vessel collisions by year.

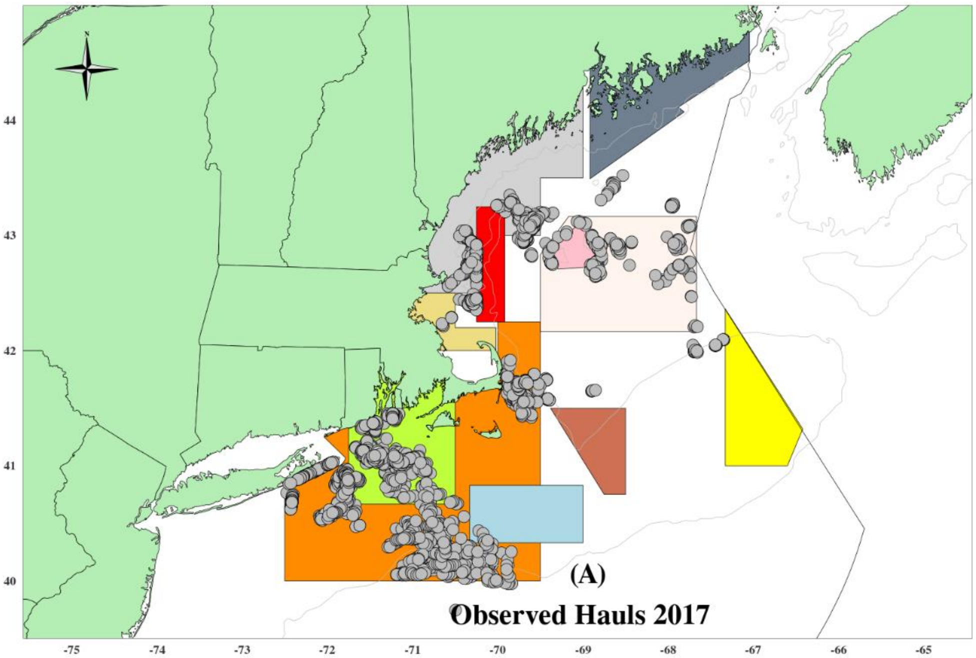
Stock	Mean Annual M/SI rate (PBR[1] for reference)	Entanglements Annual Rate (U.S. waters, Canadian waters, unknown first sighted in U.S., unknown first sighted in Canada)	Entanglements Confirmed Mortalities (2017, 2018, 2019, 2020, 2021)	Entanglements Injury Value Against PBR (2017, 2018, 2019, 2020, 2021)	Vessel Collisions Annual Rate (U.S. waters, Canadian waters, unknown first sighted in U.S., unknown first sighted in Canada)	Vessel Collisions Confirmed Mortalities (2017, 2018, 2019, 2020, 2021)	Vessel Collisions Injury Value Against PBR (2017, 2018, 2019, 2020, 2021)	Mortality due to other human-caused sources including plastic ingestion, oil spill, etc. (2017–2021)
Western North Atlantic Right Whale ***(Eubalaena glacialis)***	7.1 (0.7)	4.60 (0.00/ 1.75/ 2.70/ 0.15)	(4, 3, 1, 1, 0)	(2, 4.25, 1, 2.75, 4)	2.50 (0.90/ 1.60/ 0.00/ 0.00)	(5, 0, 4, 1, 1)	(0, 0, 0, 1, 0.52)	0
Gulf of Maine Humpback Whale ***(Megaptera novaeangliae)***[2]	15.3 (22)	9.80 (2.15/ 0.45/ 6.85/ 0.35)	(2, 3, 1, 4, 2)	(6, 9.25, 6, 7.5, 8.25)	5.5 (4.70/ 0.00/ 0.80/ 0.00)	(8, 7, 5, 3, 0)	(1, 2, 0.52, 0, 1)	0
Western North Atlantic Fin Whale ***(Balaenoptera physalus)***	2.09 (11)	1.45 (0.00/ 0.80/ 0.15/ 0.5 0)	(1, 1, 2, 0, 0)	(0, 0, 0, 1.5, 1.75)	0.64 (0.64/ 0.00/ 0.00/ 0.00)	(1, 1, 0, 0, 1)	(0, 0, 0, 0, 0.2)	0
Nova Scotian Sei Whale ***(B. borealis)***	0.6 (6.2)	0.40 (0, 0, 0.40, 0)	(0, 1, 0, 0, 0)	(1, 0, 0, 0, 0)	0	0	0	1
Canadian East Coast Minke Whale ***(B. acutorostrata)***	9.4 (170)	8.60 (3.55/ 2.40/ 1.95/ 0.70)	(12, 11, 3, 5, 2)	(1.5, 2.25, 3.75, 1, 1.5)	0.80 (0.60/ 0.20/ 0.00/ 0.00)	(2, 1, 0, 0, 1)	0	0
Northern Gulf of Mexico Rice's whale (B. Ricei)	0.4 (0.1)	0	0	0	0	0	0	(.33, .25, 1.19, .14, , .09)

[1] Potential Biological Removal (PBR)
[2] Humpback and Rice's whale SARs were not updated in 2023. Values reported here are published in Henry et al. (2023) and Appendix VI (Appendix VI for Rice's whale only).

Henry, A.G., M. Garron, D. Morin, A. Smith, A. Reid, W. Ledwell and T.V.N. Cole. 2023. Serious injury and mortality determinations for baleen whale stocks along the Gulf of Mexico, United States East Coast and Atlantic Canadian Provinces, 2017–2021. NOAA Tech Memo. NMFS-NE-x.

Appendix III: Fishery Descriptions - List of Figures

Figure 1. 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

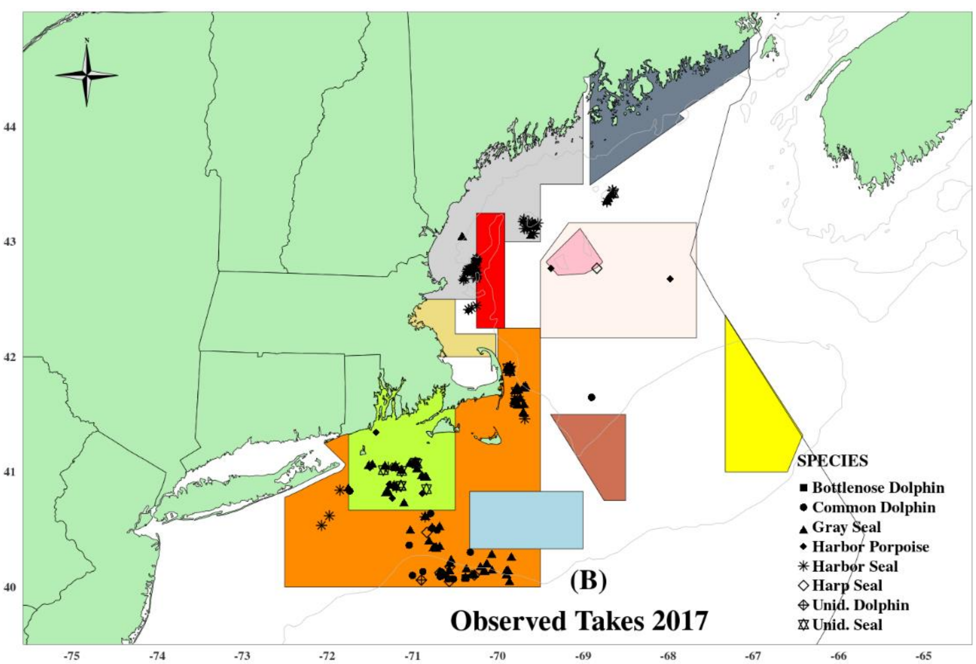
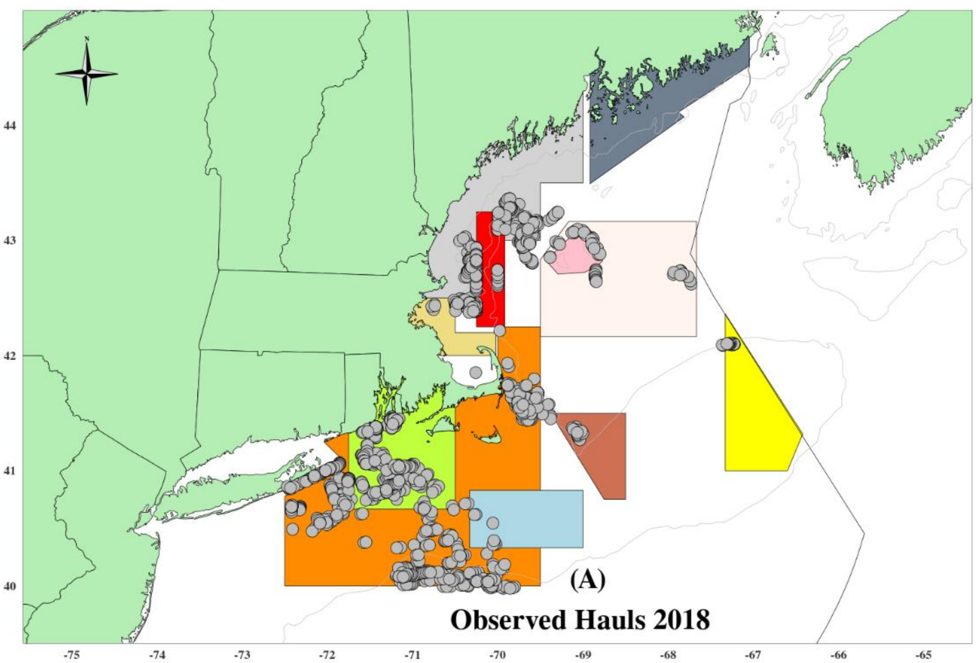


Figure 2. 2018 Northeast sink gillnet observed hauls (A) and observed takes (B).



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

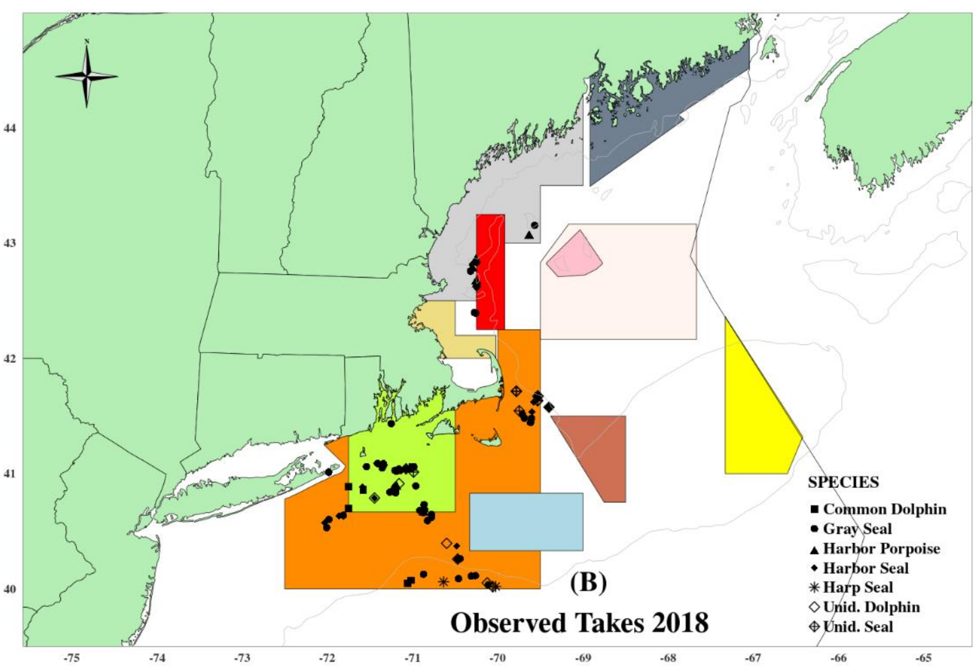
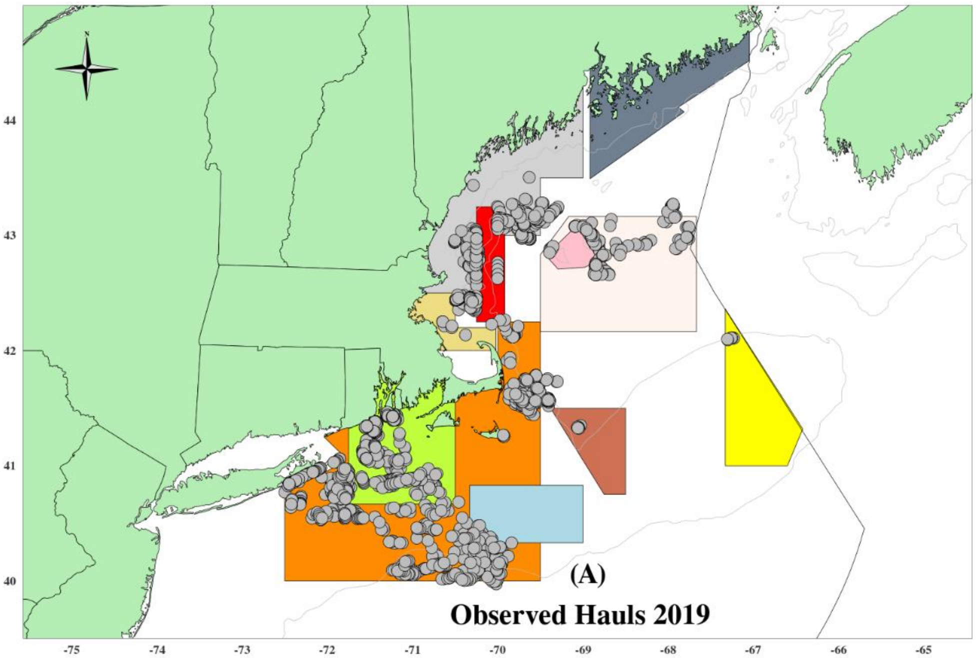


Figure 3. 2019 Northeast sink gillnet observed hauls (A) and observed takes (B).



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

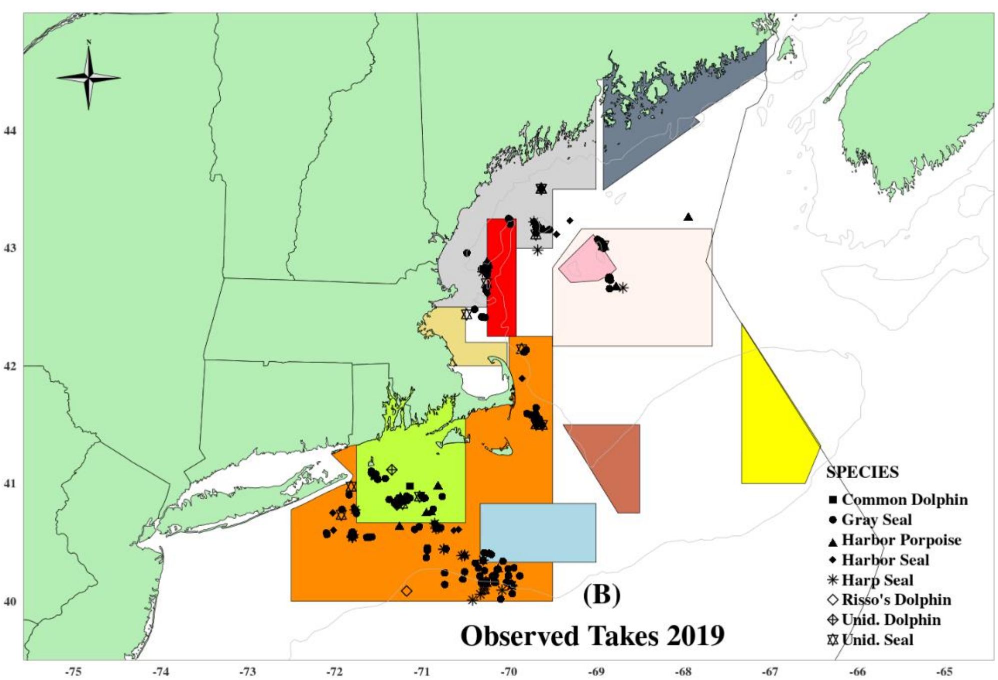
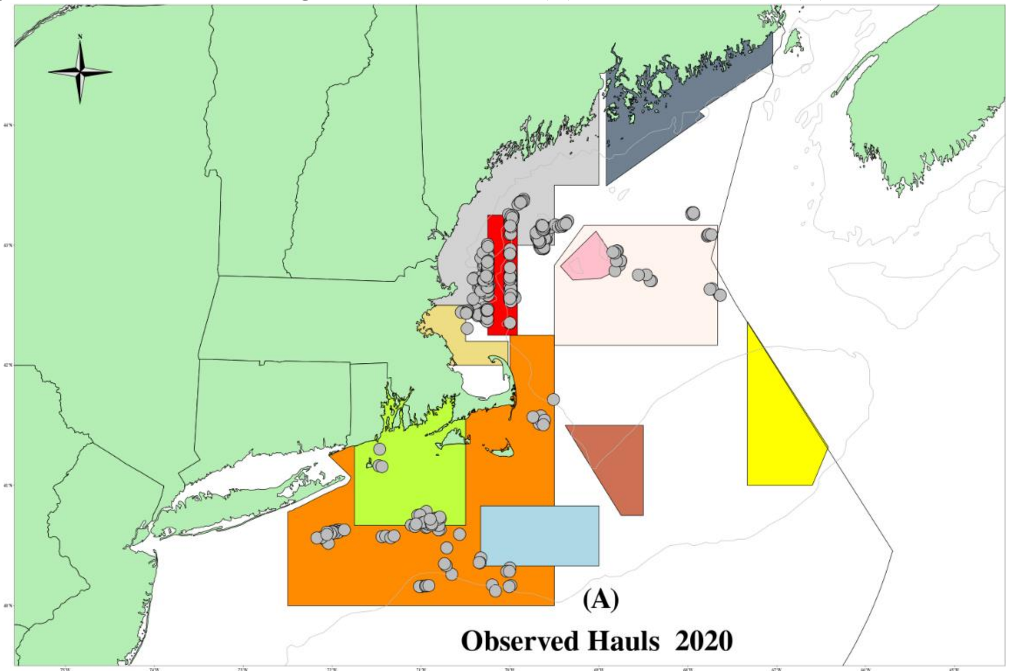


Figure 4. 2020 Northeast sink gillnet observed hauls (A) and observed takes (B).



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

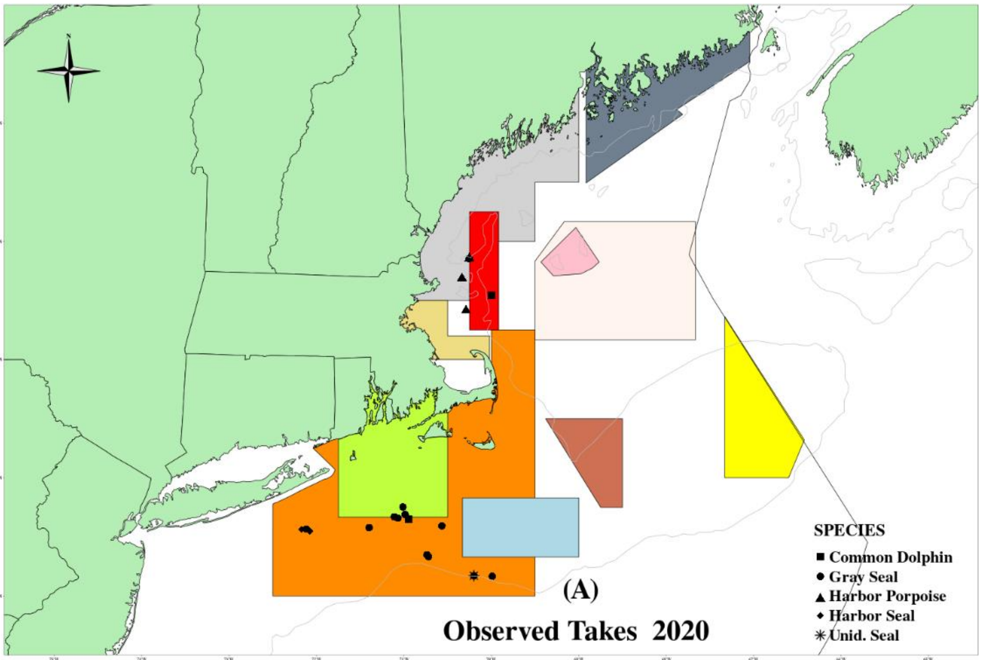
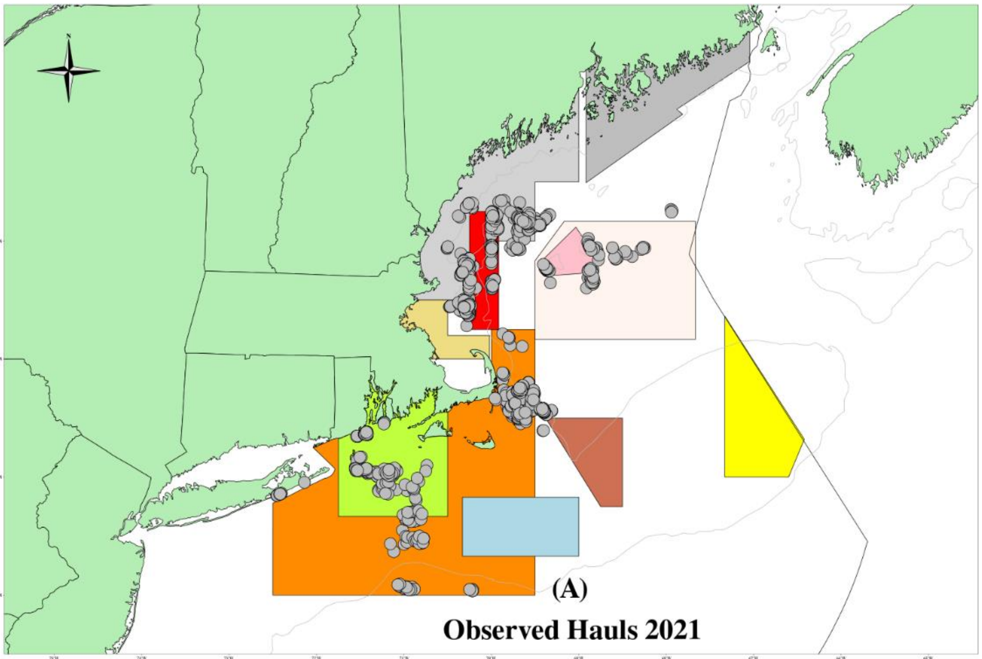


Figure 5. 2021 Northeast sink gillnet observed hauls (A) and observed takes (B).



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

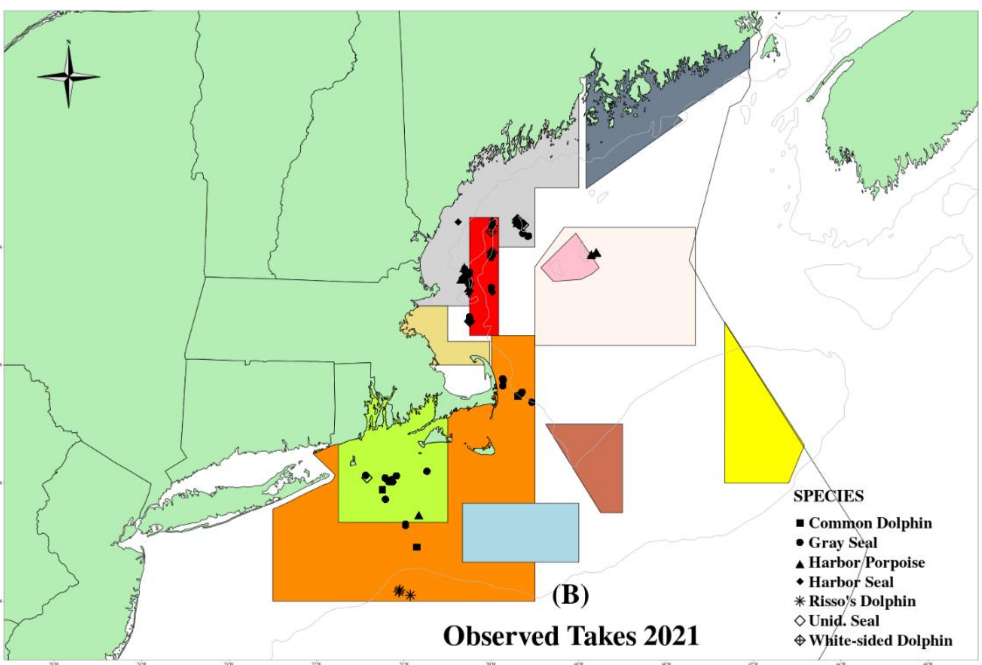
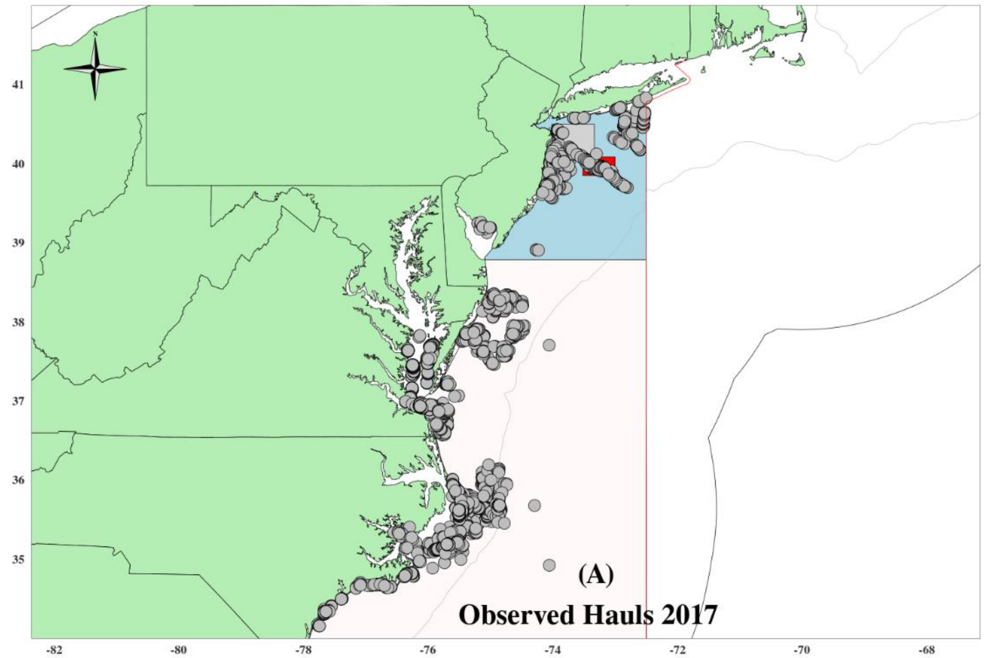


Figure 6. 2017 Mid-Atlantic gillnet observed hauls (A) and observed takes (B).



Harbor porpoise Take Reduction Plan management areas:
Southern mid-Atlantic waters | New Jersey Mudhole | Mudhole South | waters off New Jersey

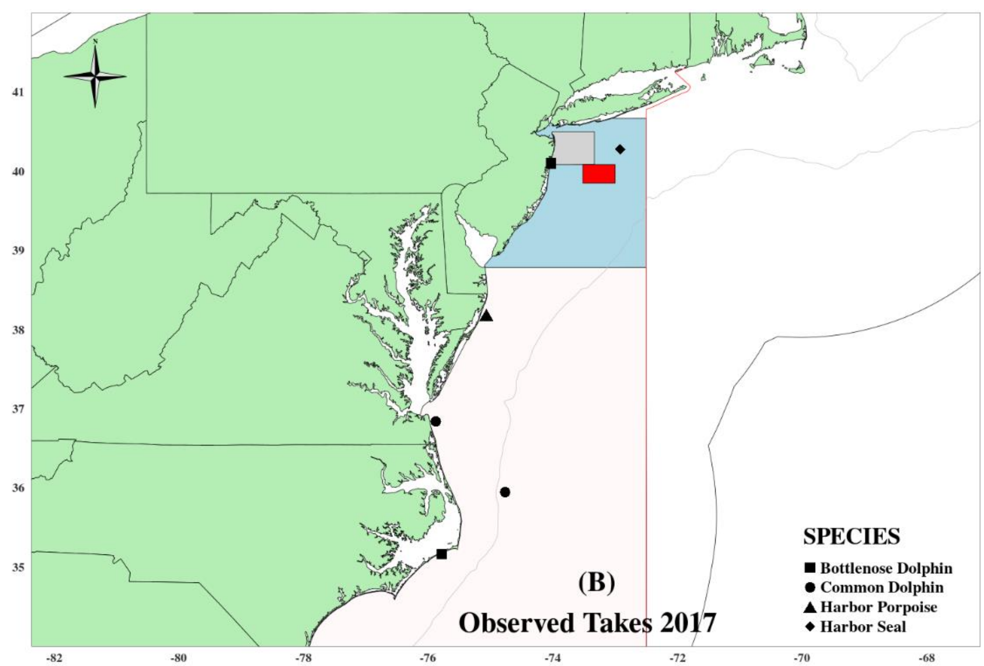
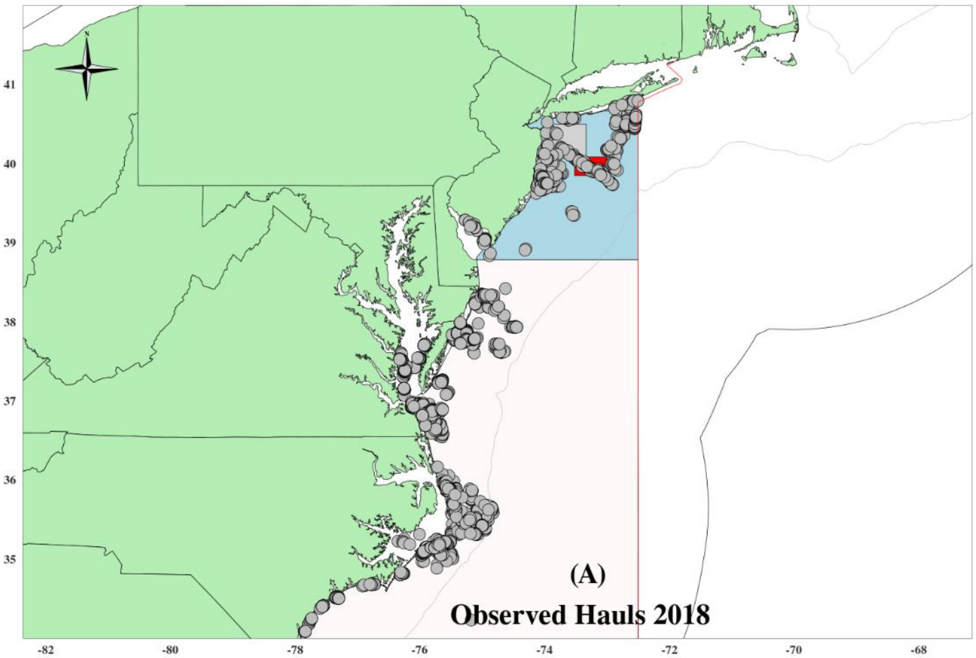


Figure 7. 2018 Mid-Atlantic gillnet observed hauls (A) and observed takes (B).



Harbor porpoise Take Reduction Plan management areas:
Southern mid-Atlantic waters | New Jersey Mudhole | Mudhole South | waters off New Jersey

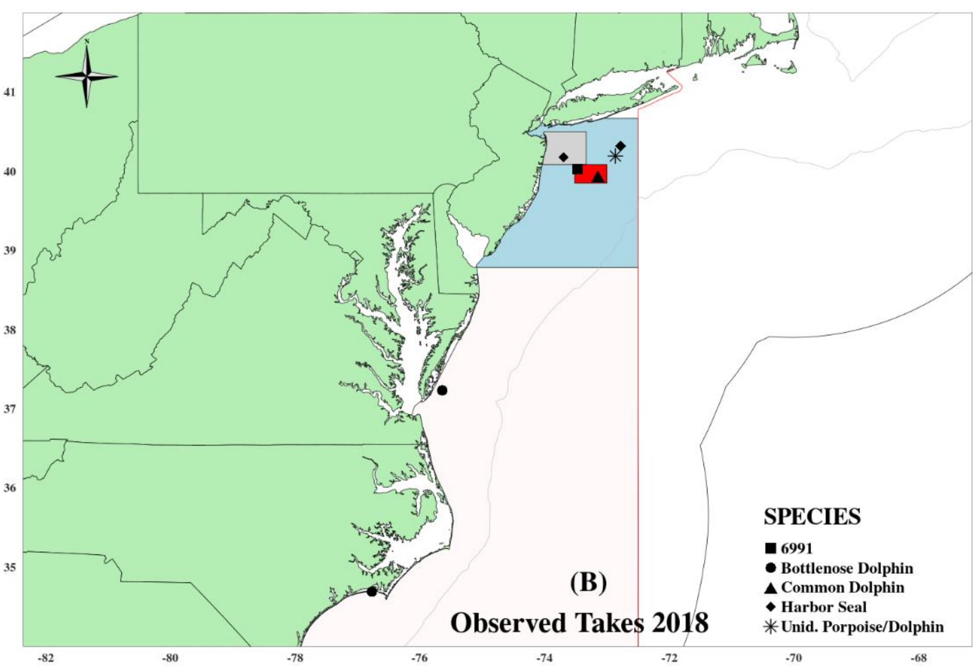
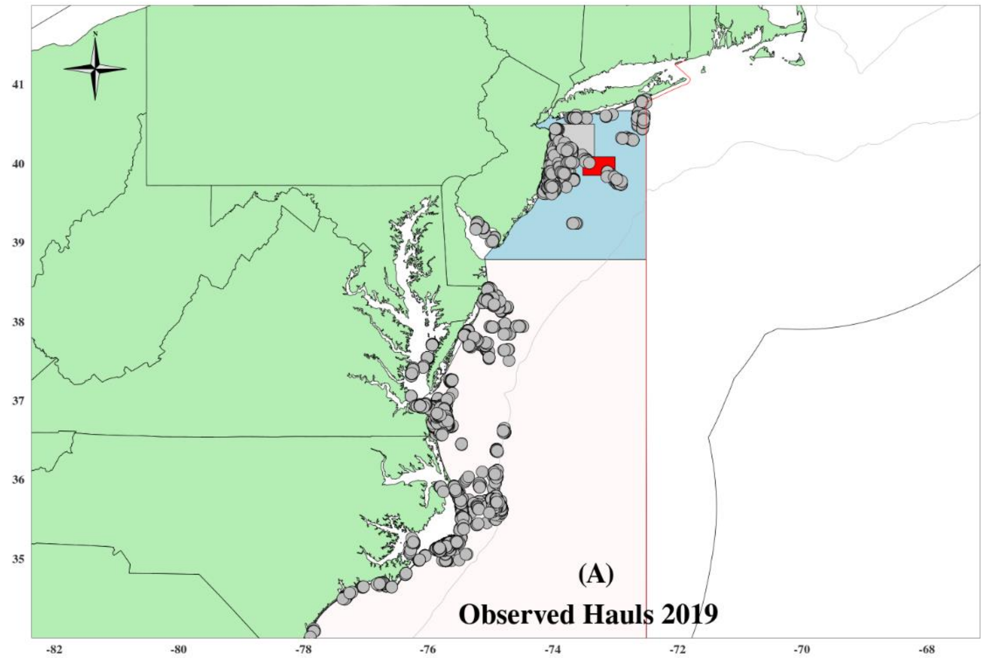


Figure 8. 2019 Mid-Atlantic gillnet observed hauls (A) and observed takes (B).



Harbor porpoise Take Reduction Plan management areas:
Southern mid-Atlantic waters | New Jersey Mudhole | Mudhole South | waters off New Jersey

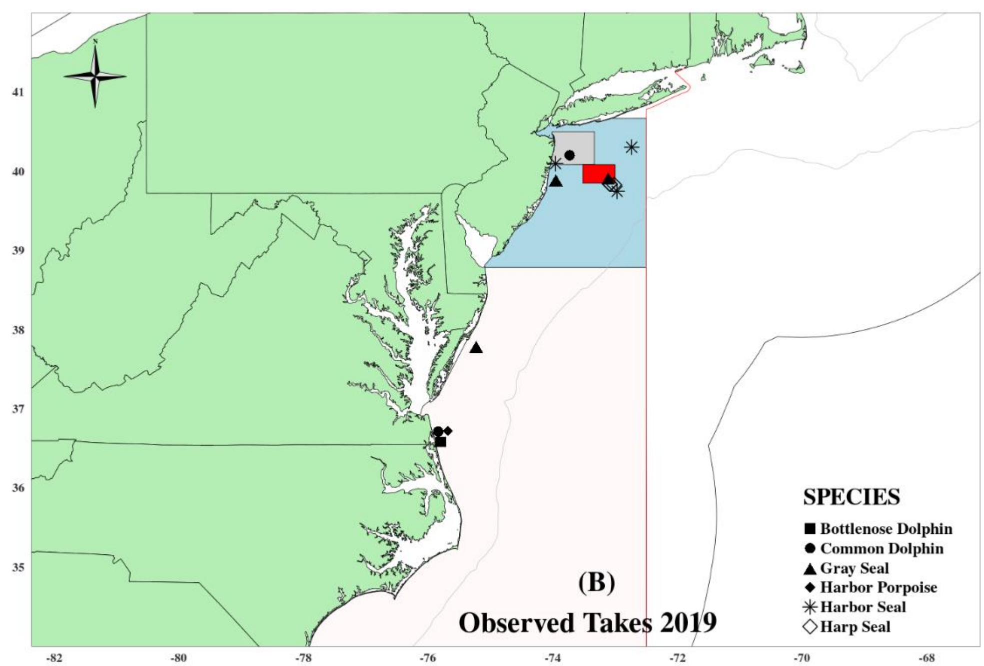
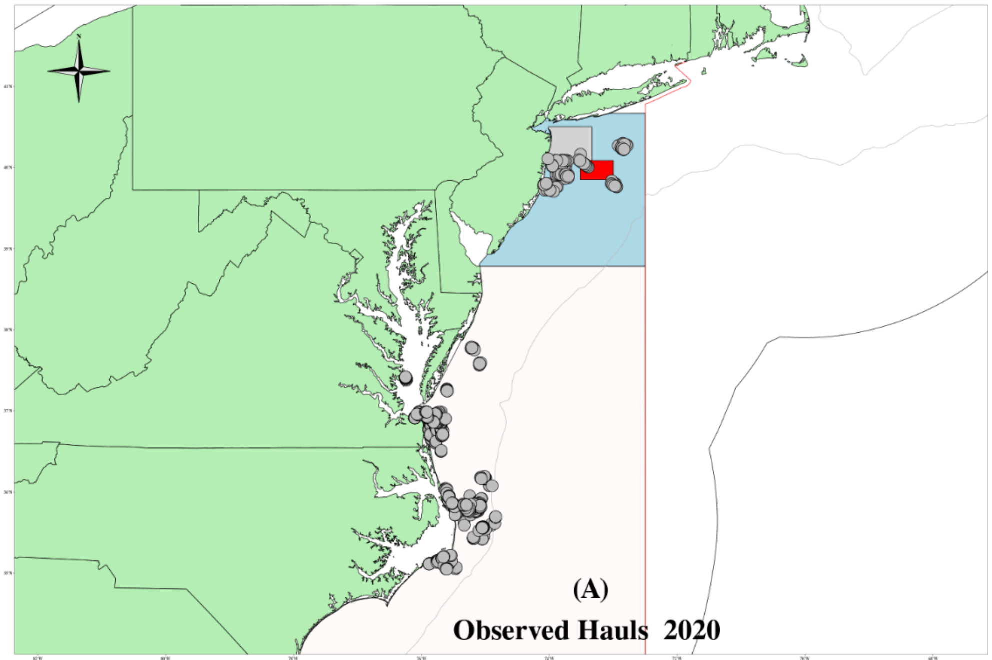


Figure 9. 2020 Mid-Atlantic gillnet observed hauls (A) and observed takes (B).



Harbor porpoise Take Reduction Plan management areas:

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

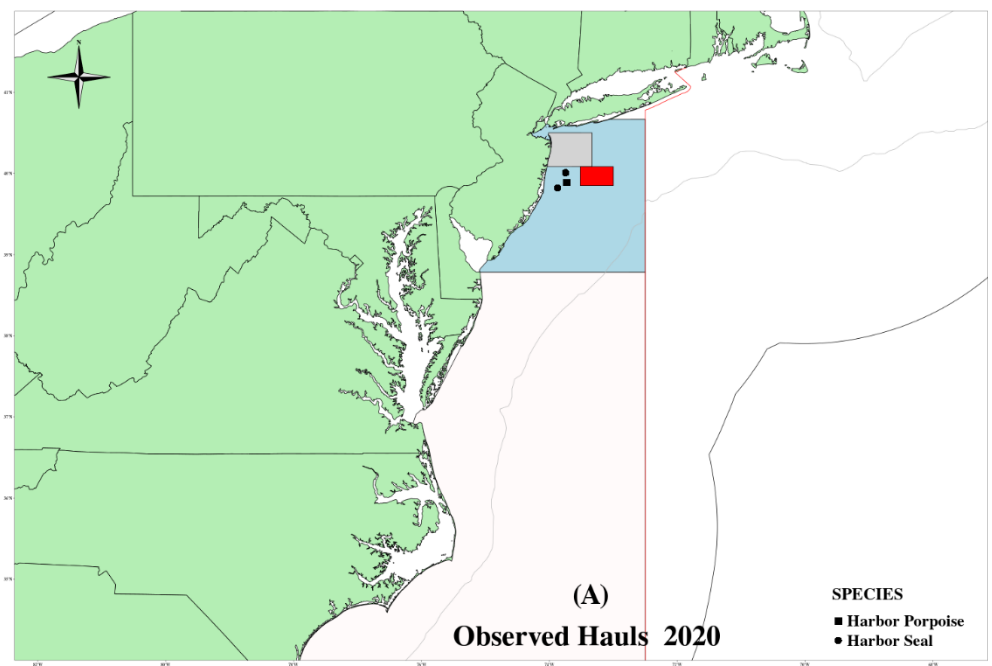
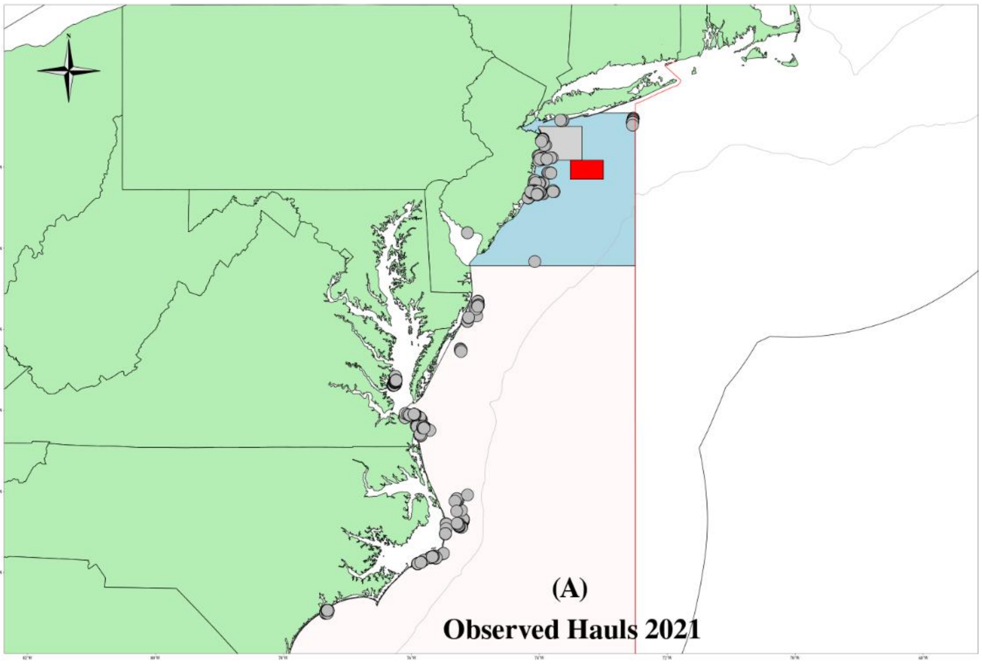


Figure 10. 2021 Mid-Atlantic gillnet observed hauls (A) and observed takes (B).



Harbor porpoise Take Reduction Plan management areas:
Southern mid-Atlantic waters | New Jersey Mudhole | Mudhole South | waters off New Jersey

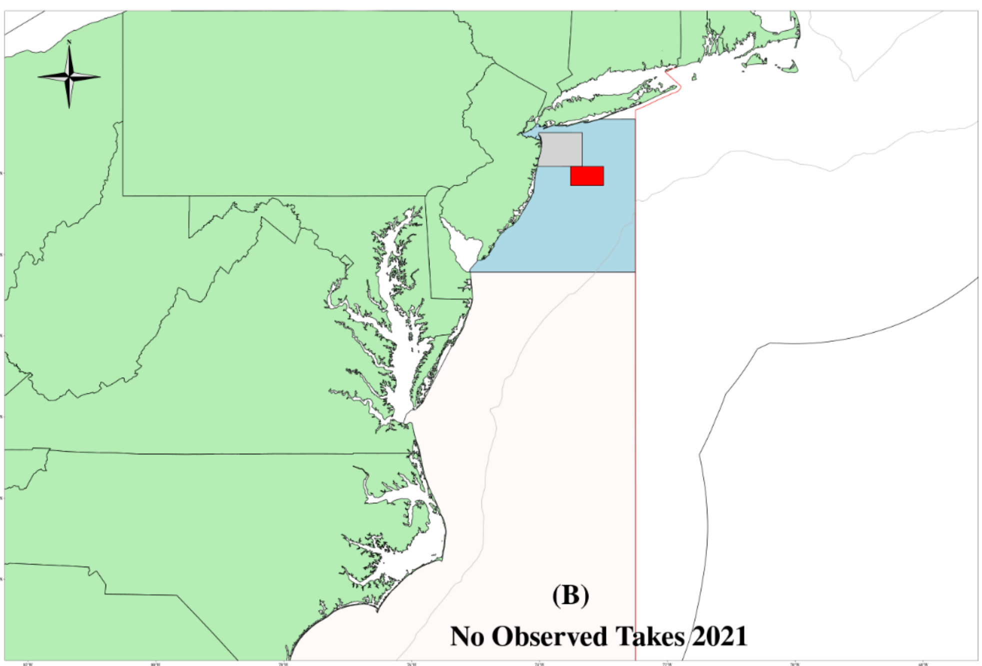
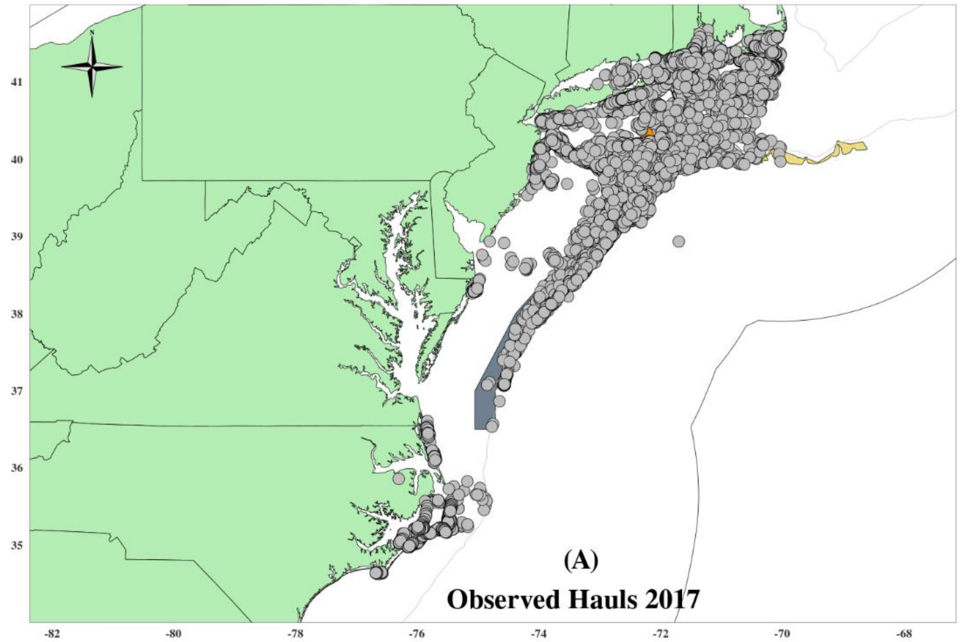


Figure 11. 2017 Mid-Atlantic bottom trawl observed tows (A) and observed takes (B).



Southern Gear Restricted Area Northern Gear Restricted Area

Restricted Area 2 Restricted Area 3 Restricted Area 4

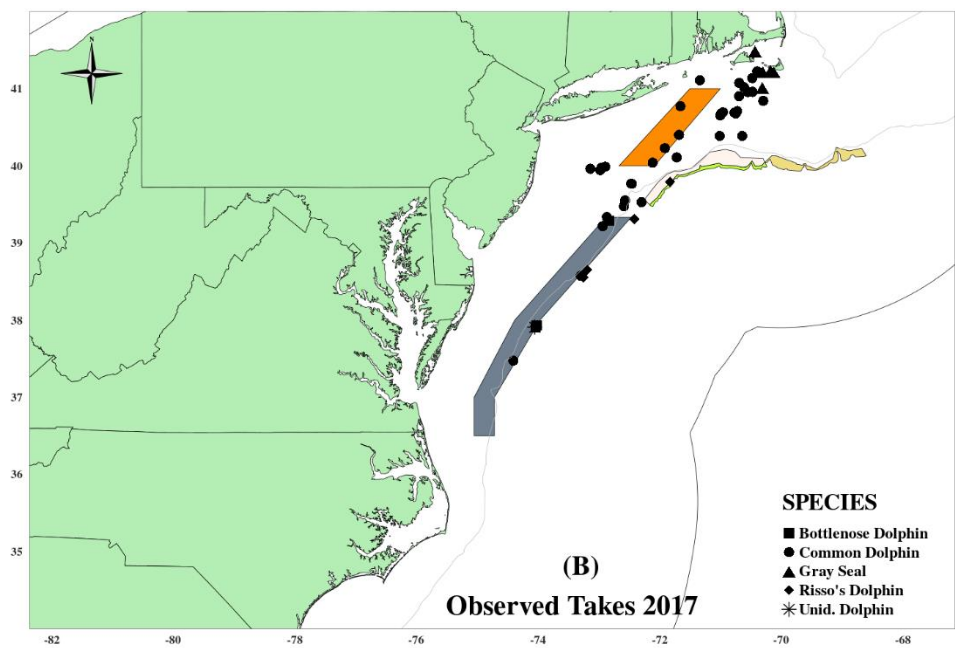


Figure 12. 2018 Mid-Atlantic bottom trawl observed tows (A) and observed takes (B).

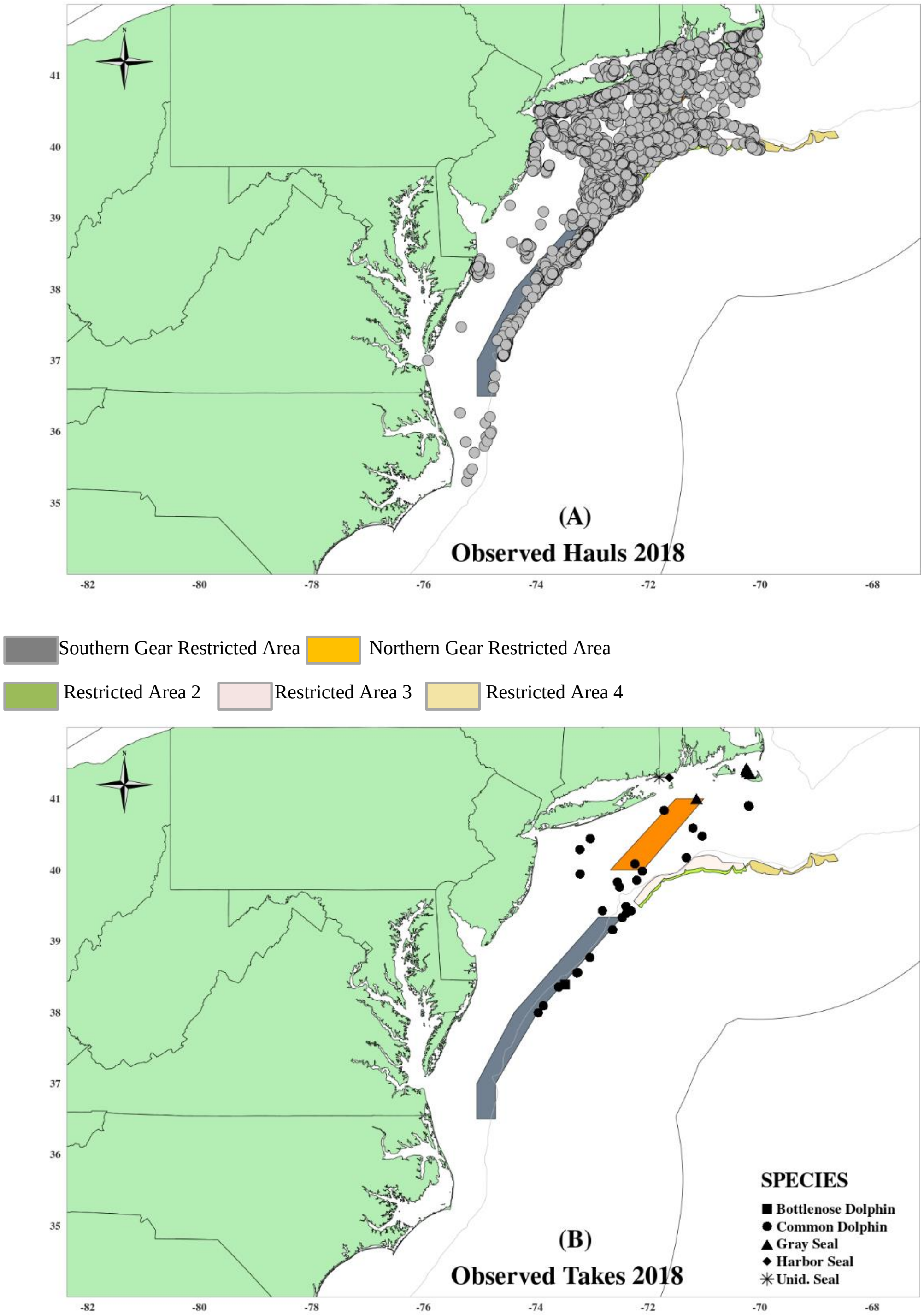


Figure 13. 2019 Mid-Atlantic bottom trawl observed tows (A) and observed takes (B).

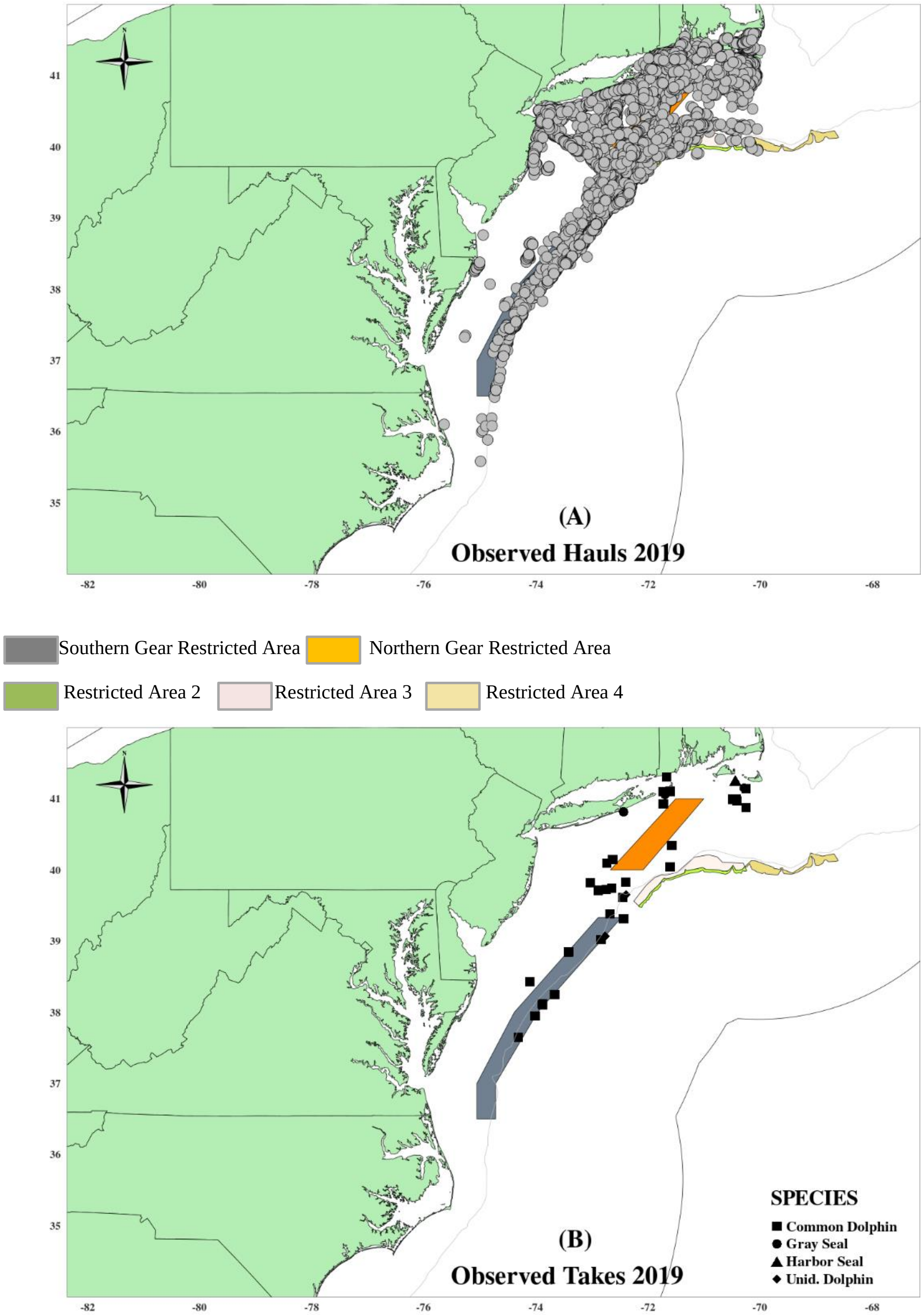
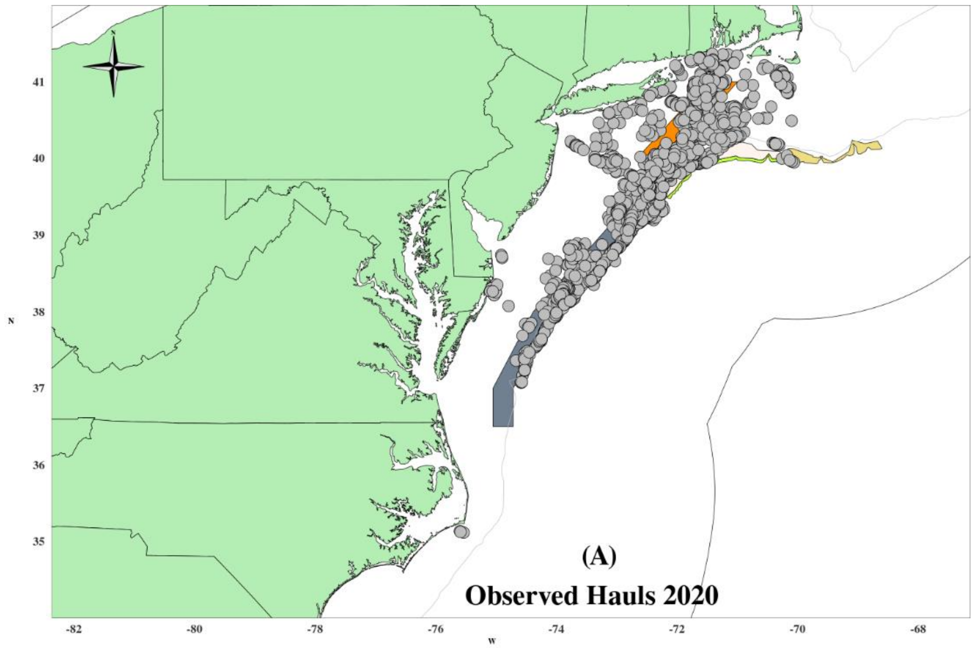


Figure 14. 2020 Mid-Atlantic bottom trawl observed tows (A) and observed takes (B).



Southern Gear Restricted Area Northern Gear Restricted Area

Restricted Area 2 Restricted Area 3 Restricted Area 4

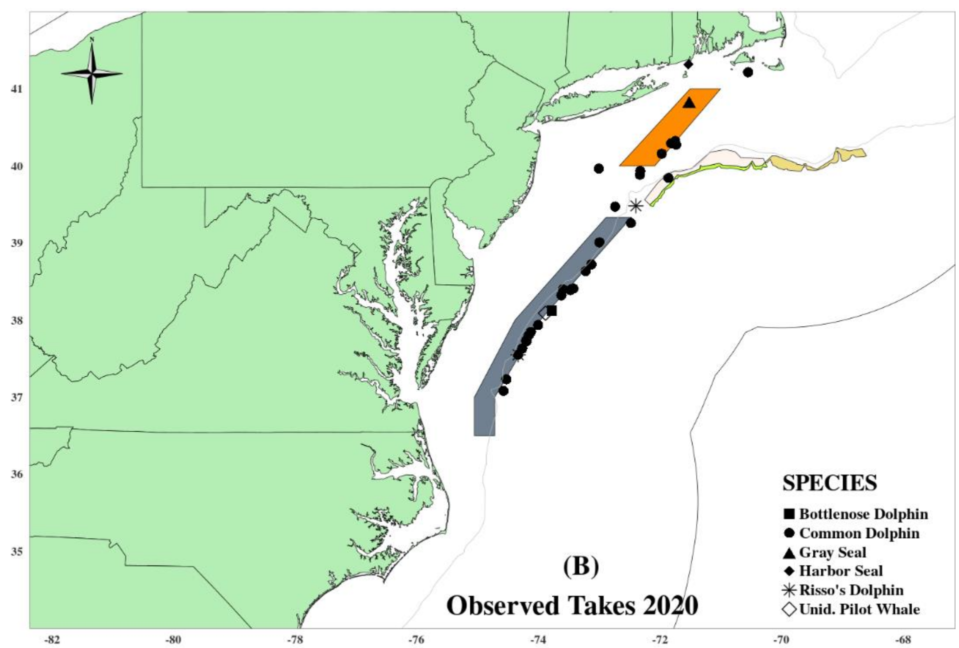


Figure 15. 2021 Mid-Atlantic bottom trawl observed tows (A) and observed takes (B).

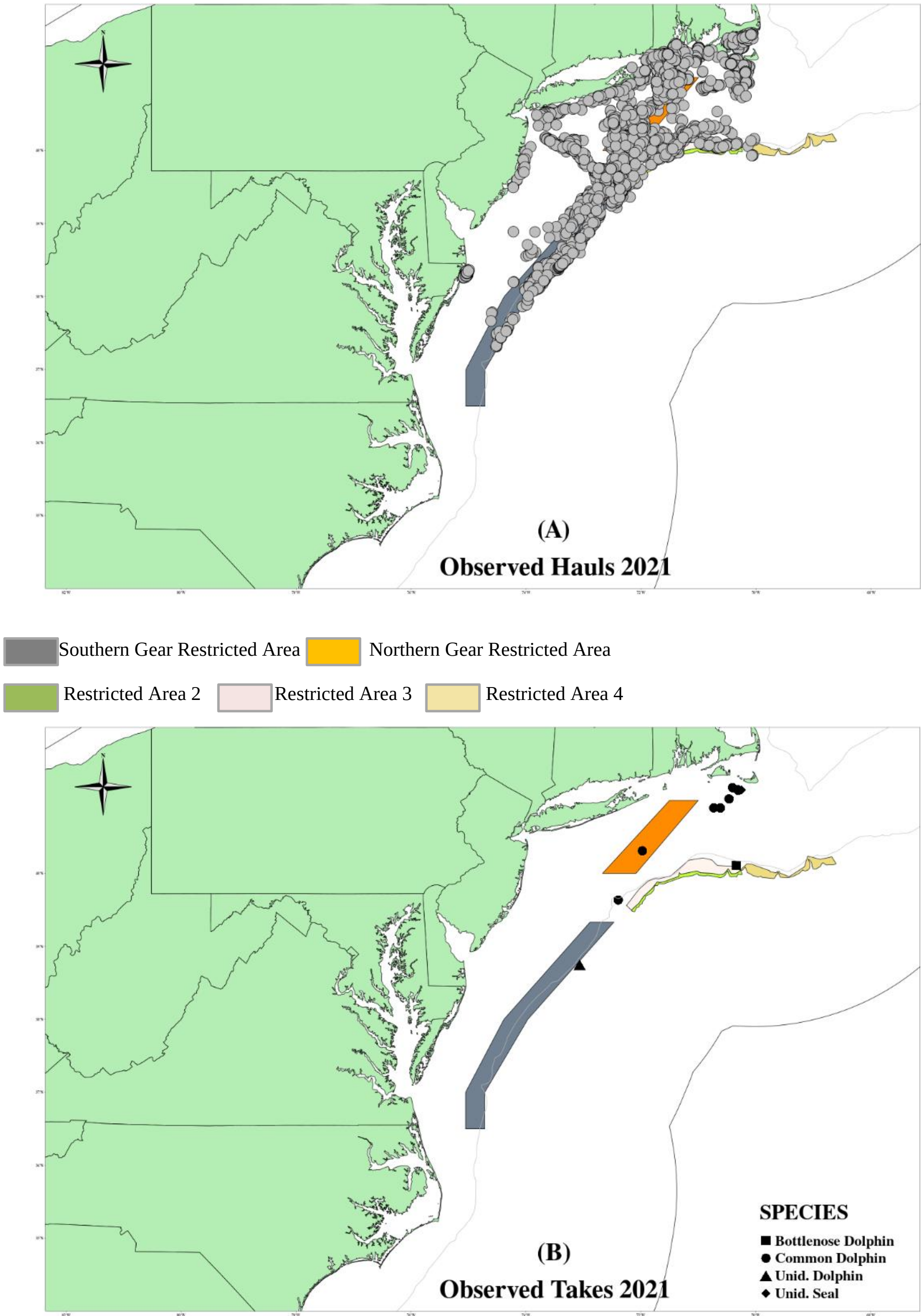
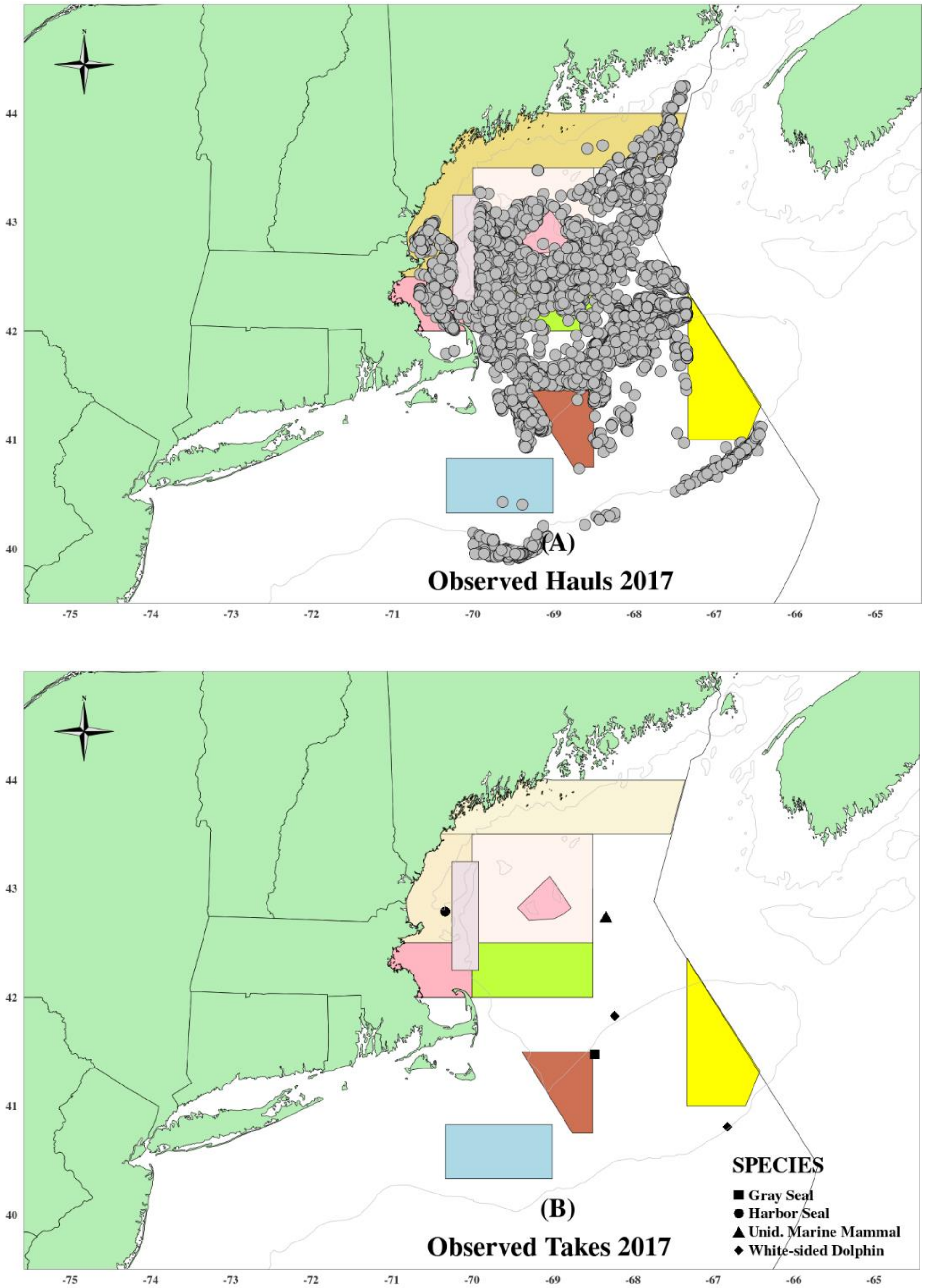


Figure 16. 2017 Northeast bottom trawl observed tows (A) and observed takes (B).



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. 2018 Northeast bottom trawl observed tows (A) and observed takes (B).

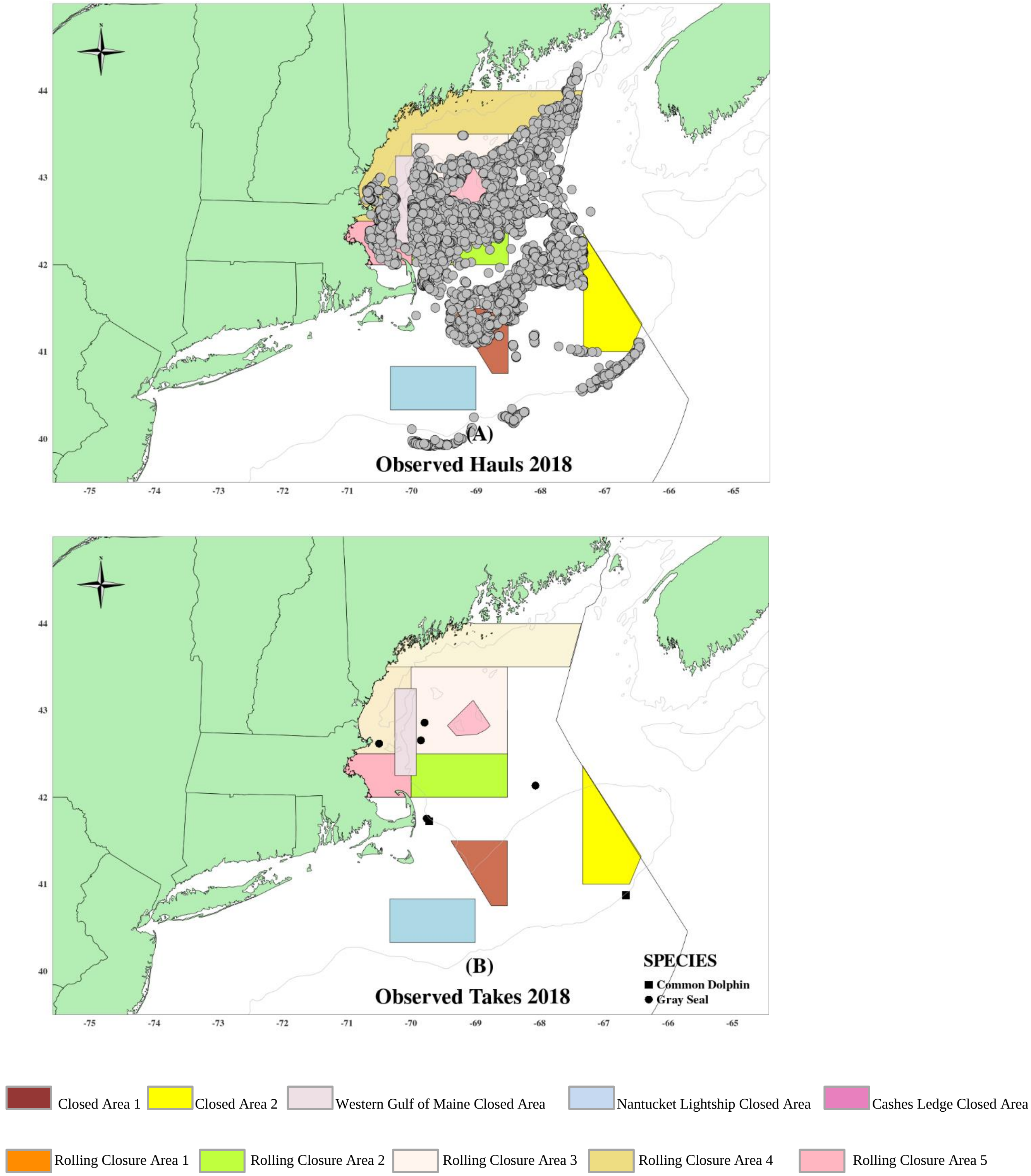


Figure 18. 2019 Northeast bottom trawl observed tows (A) and observed takes (B).

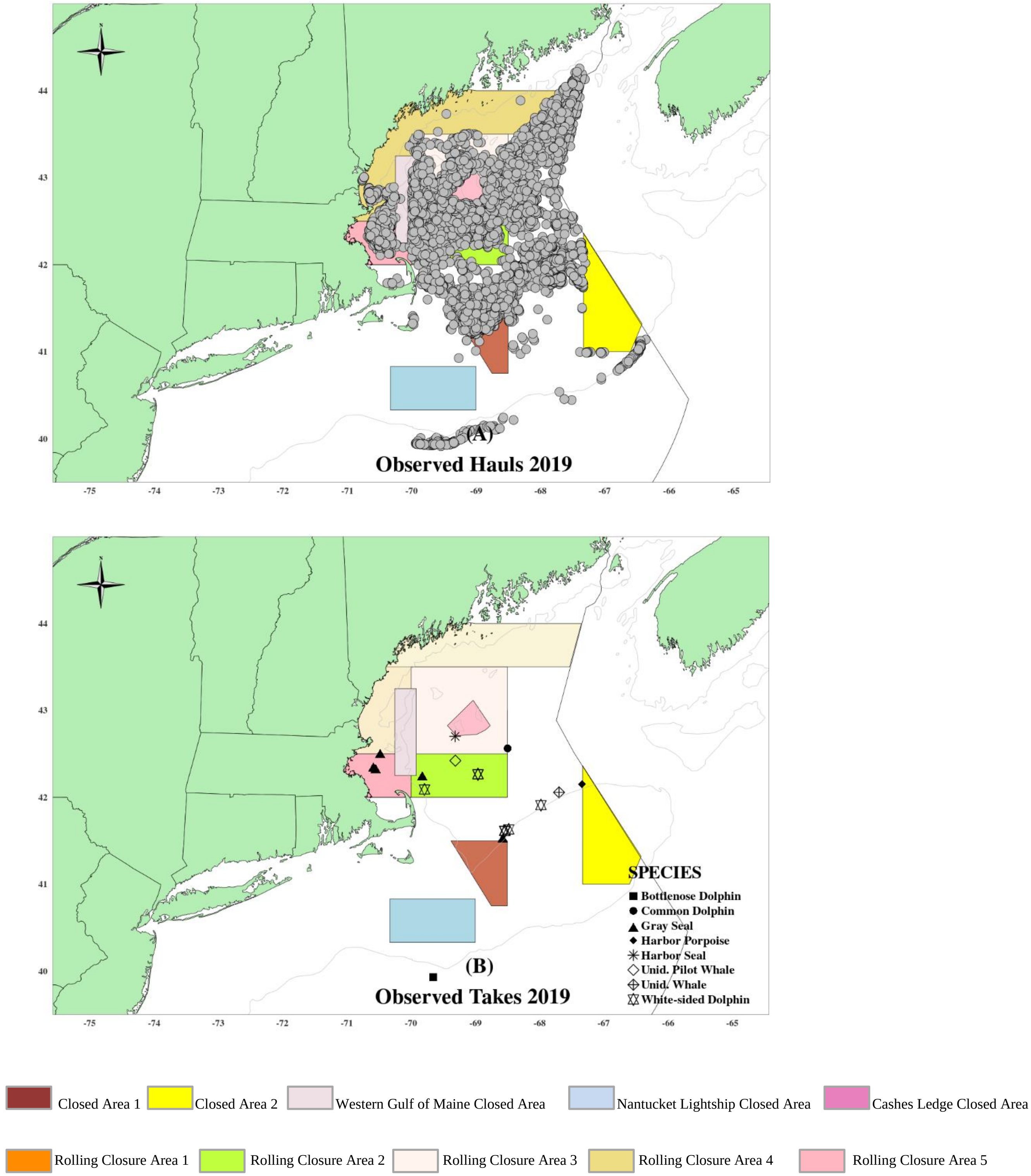
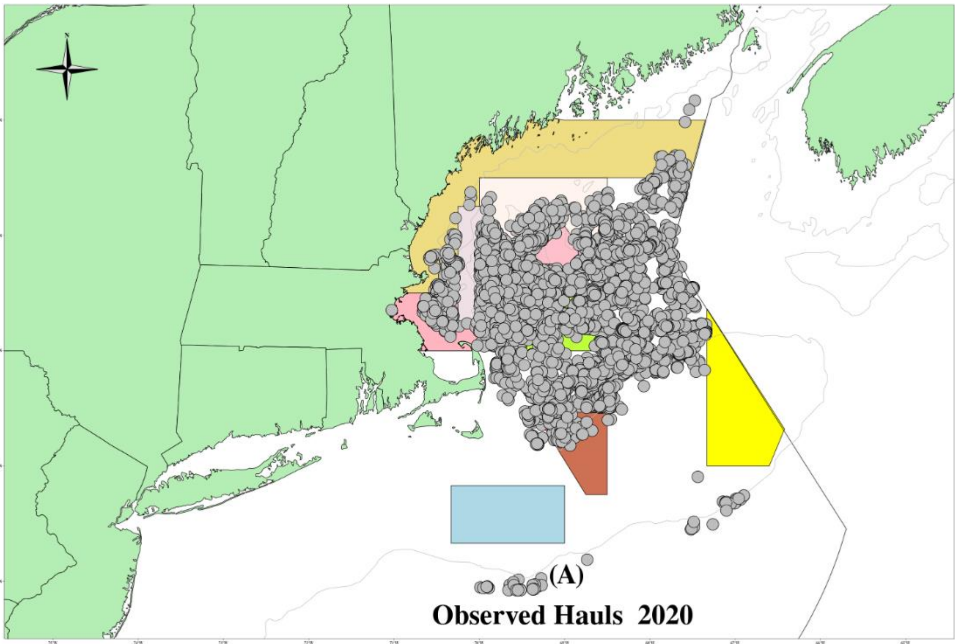
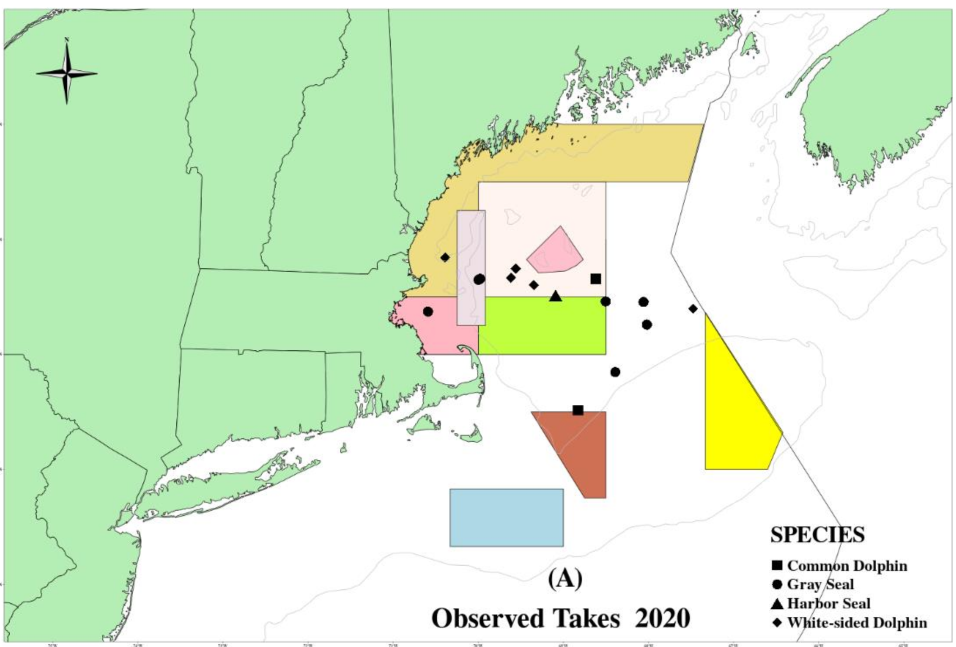


Figure 19. 2020 Northeast bottom trawl observed tows (A) and observed takes (B).





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 20. 2021 Northeast bottom trawl observed tows (A) and observed takes (B).

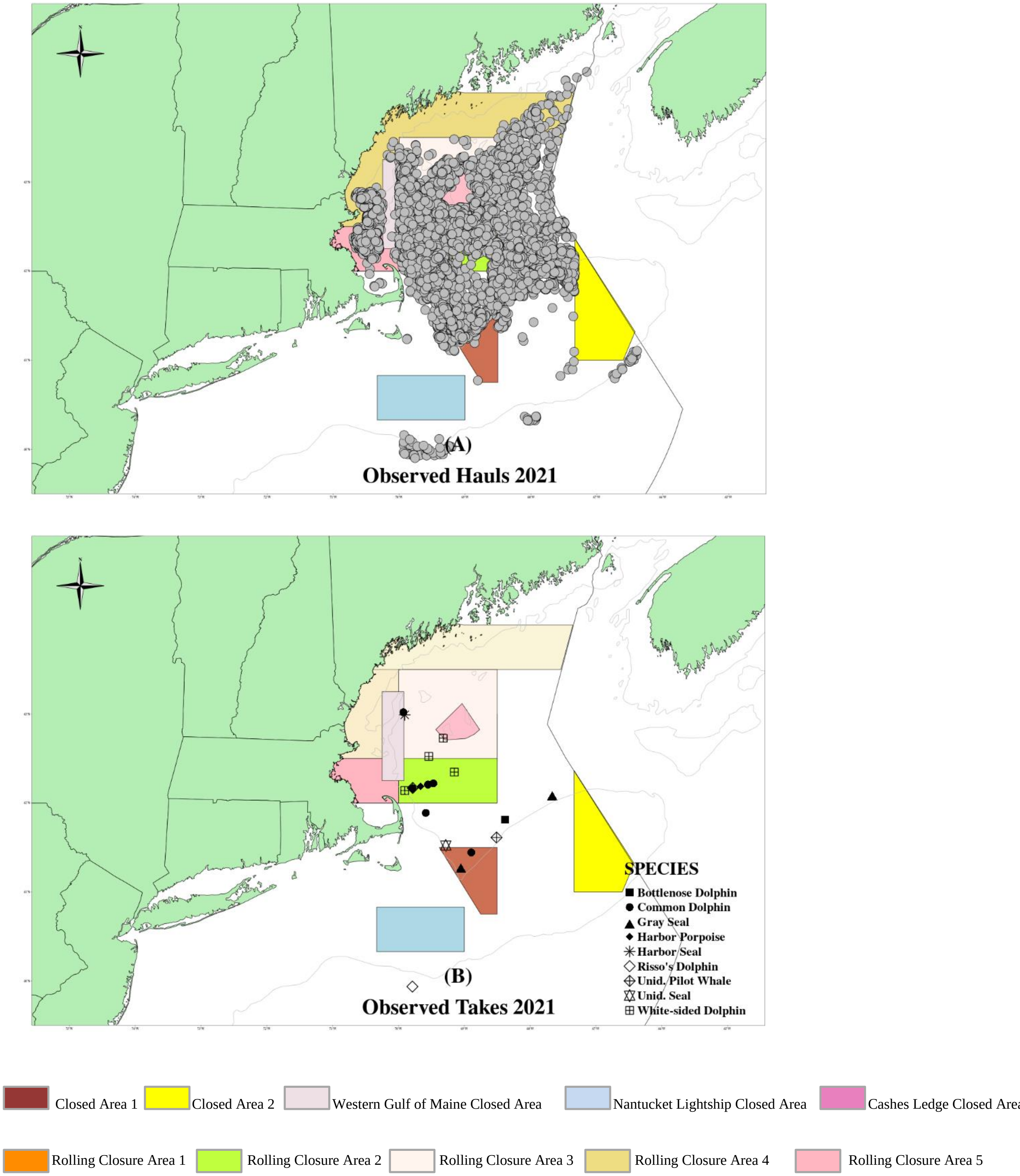
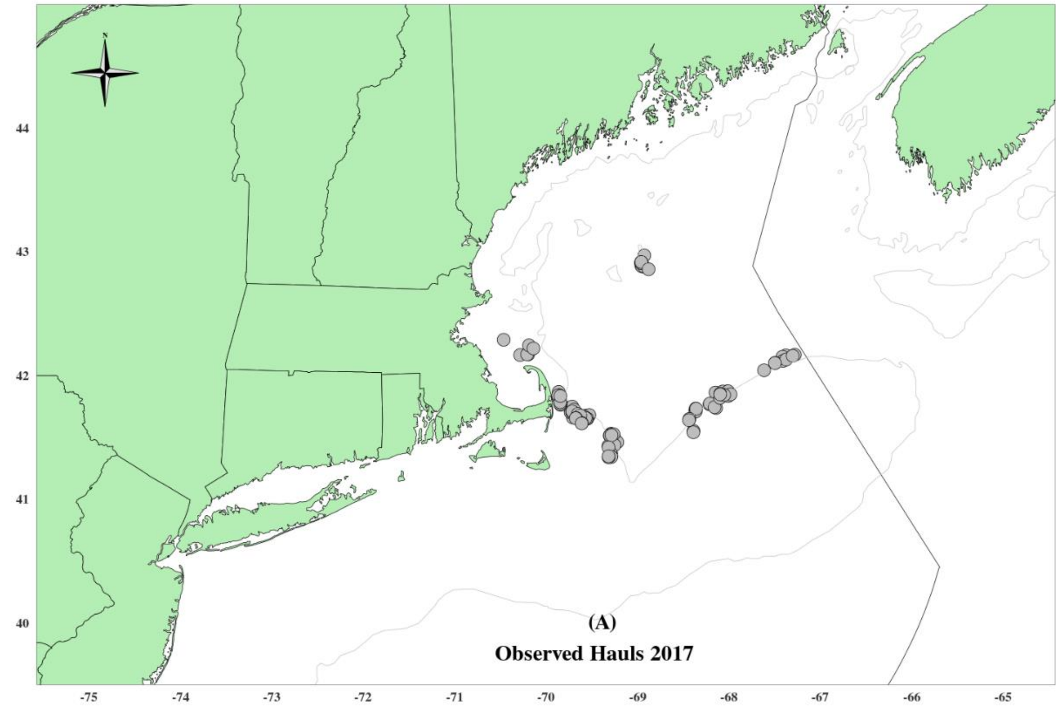


Figure 21. 2017 Northeast mid-water trawl observed tows (A) and observed takes (B).



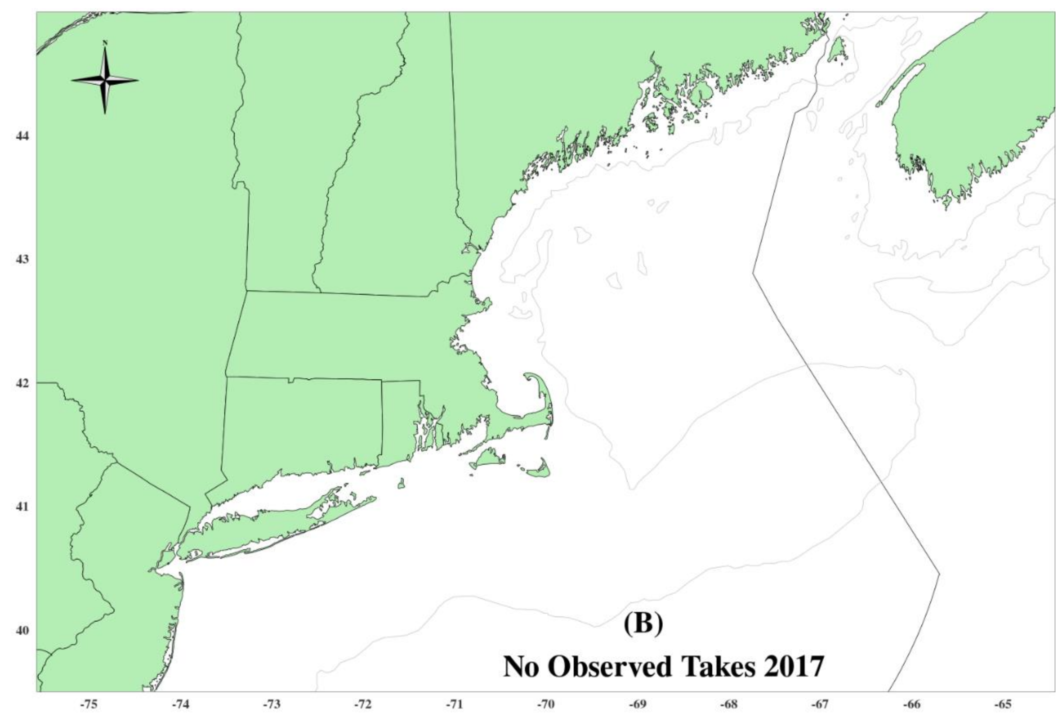


Figure 22. 2018 Northeast mid-water trawl observed tows (A) and observed takes (B).

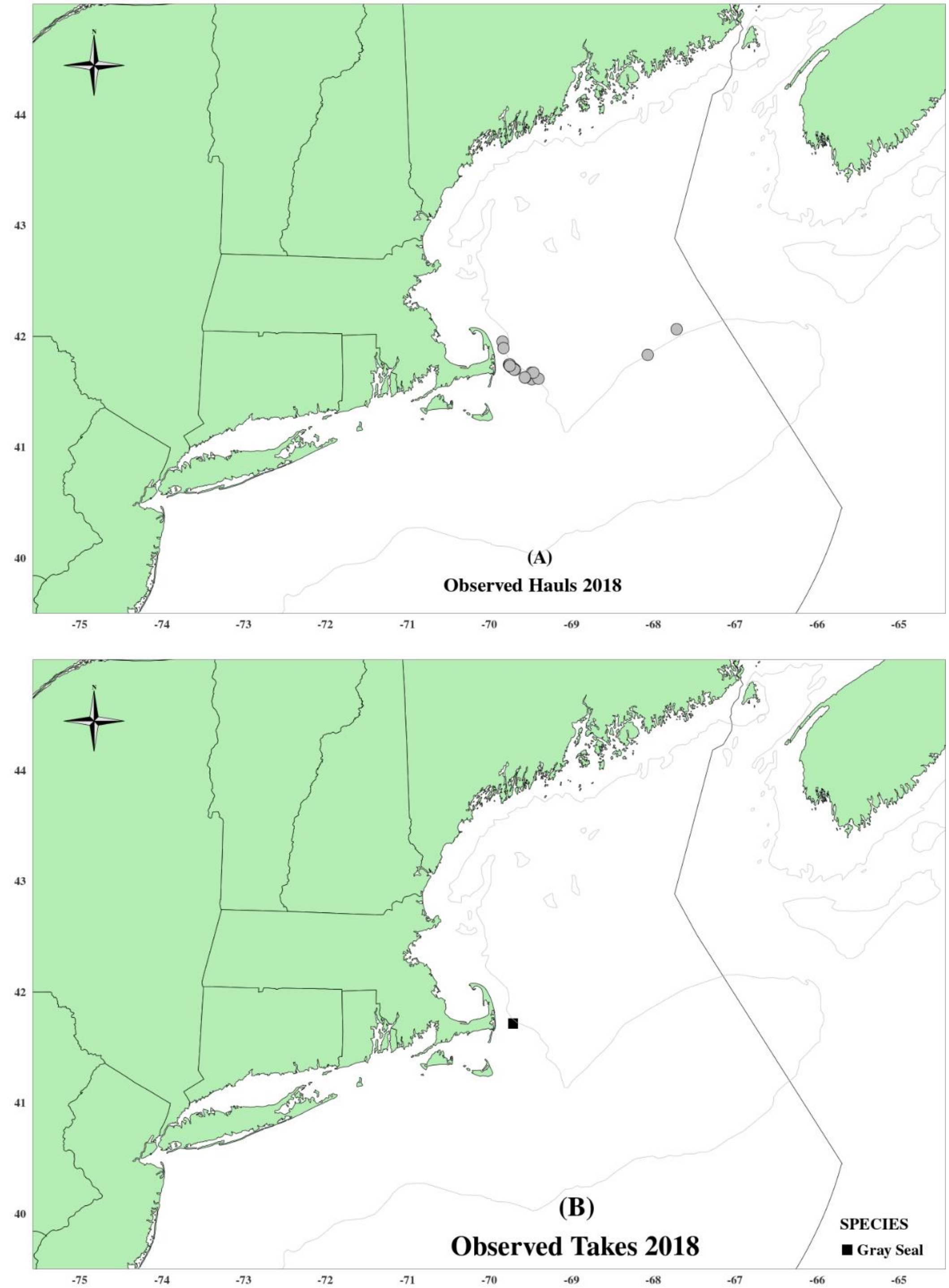
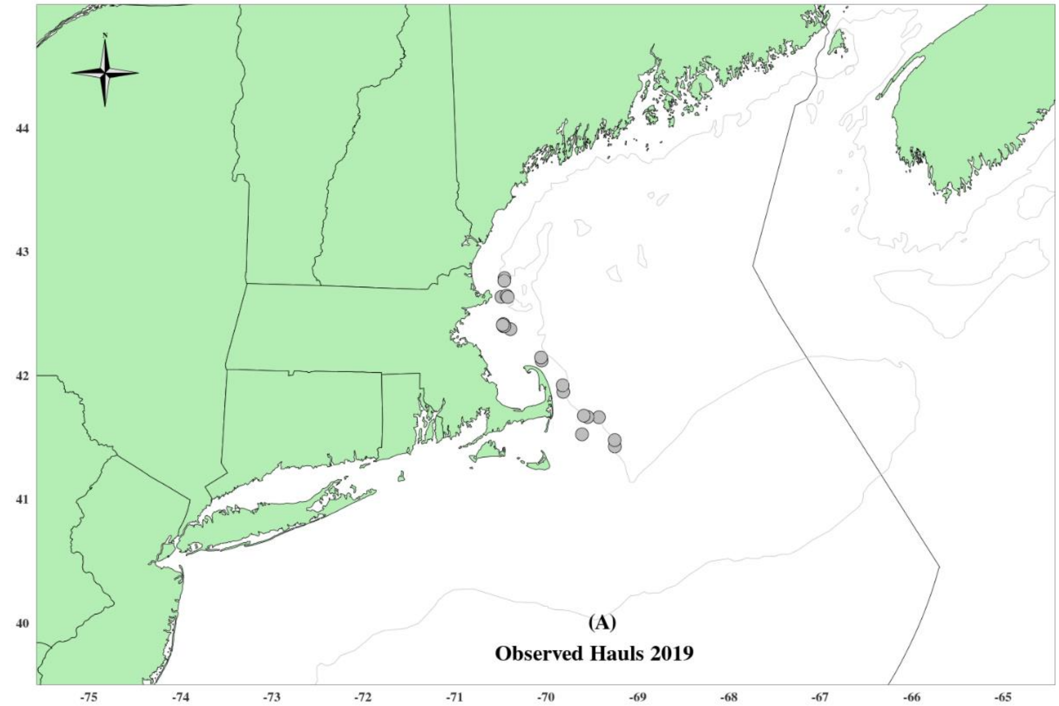


Figure 23. 2019 Northeast mid-water trawl observed tows (A) and observed takes (B).



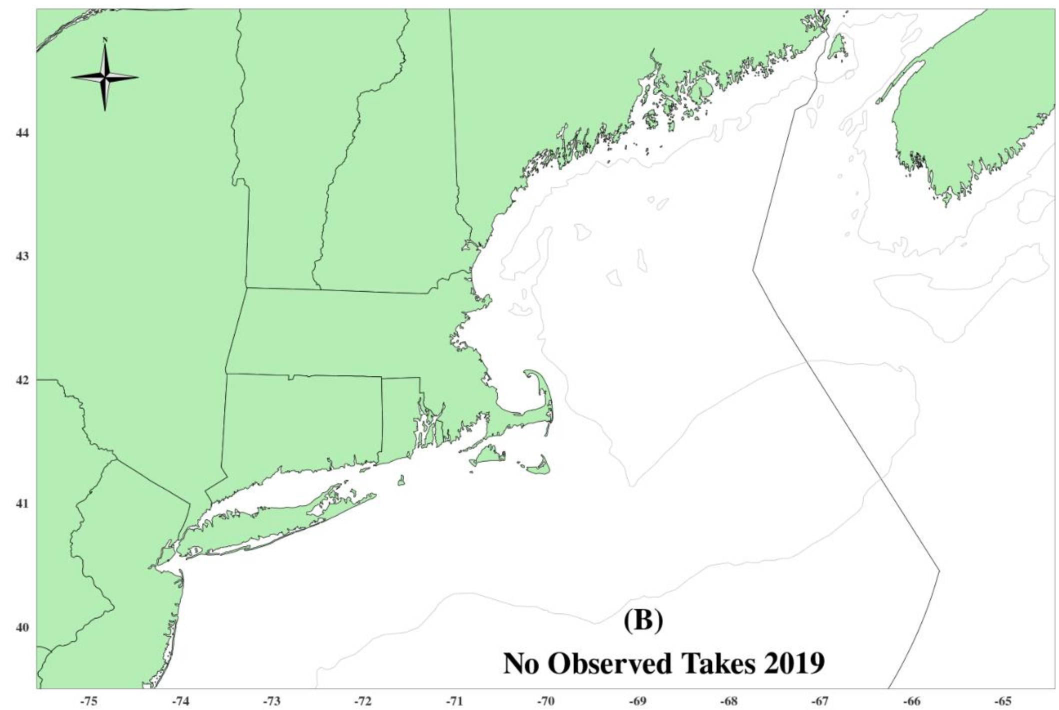
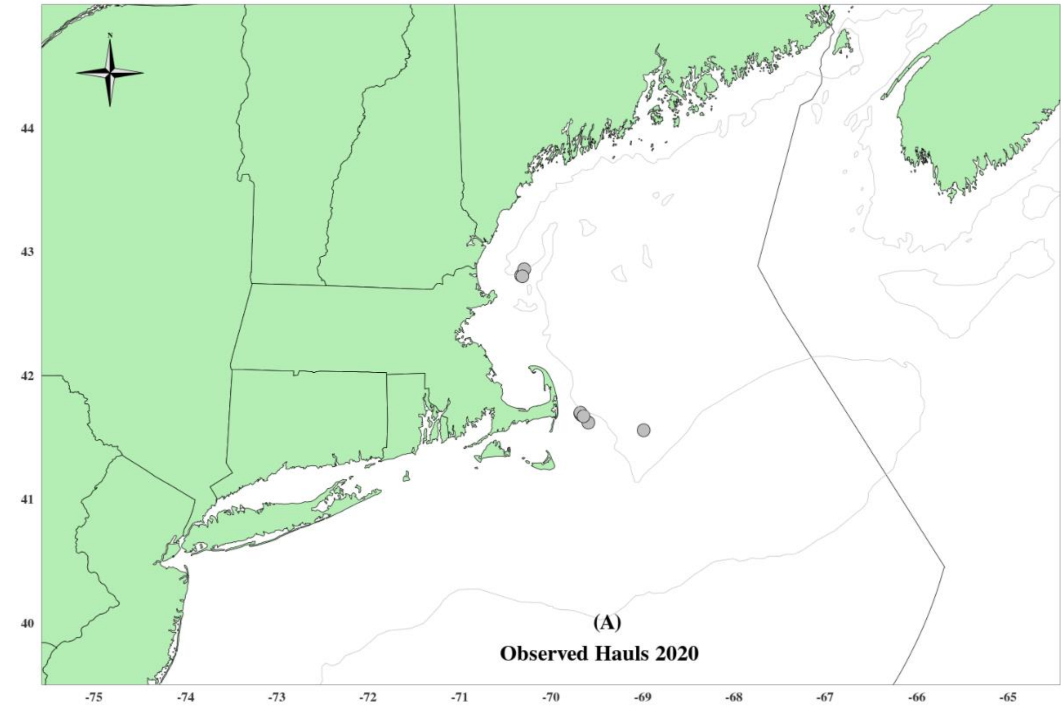


Figure 24. 2020 Northeast mid-water trawl observed tows (A) and observed takes (B).



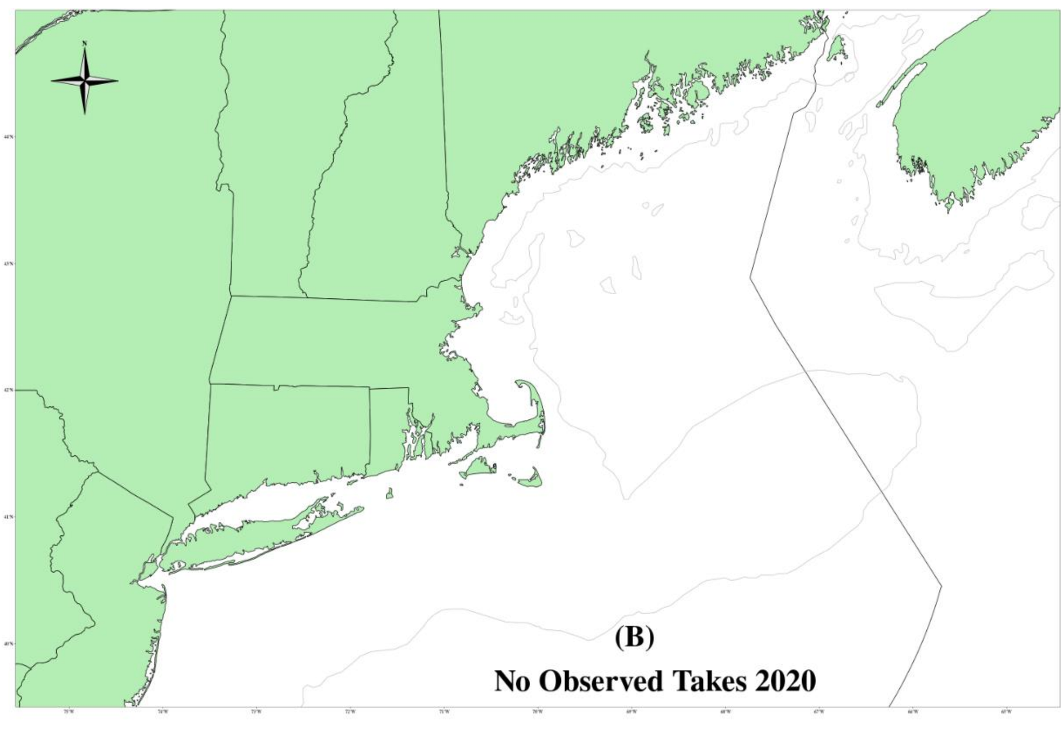
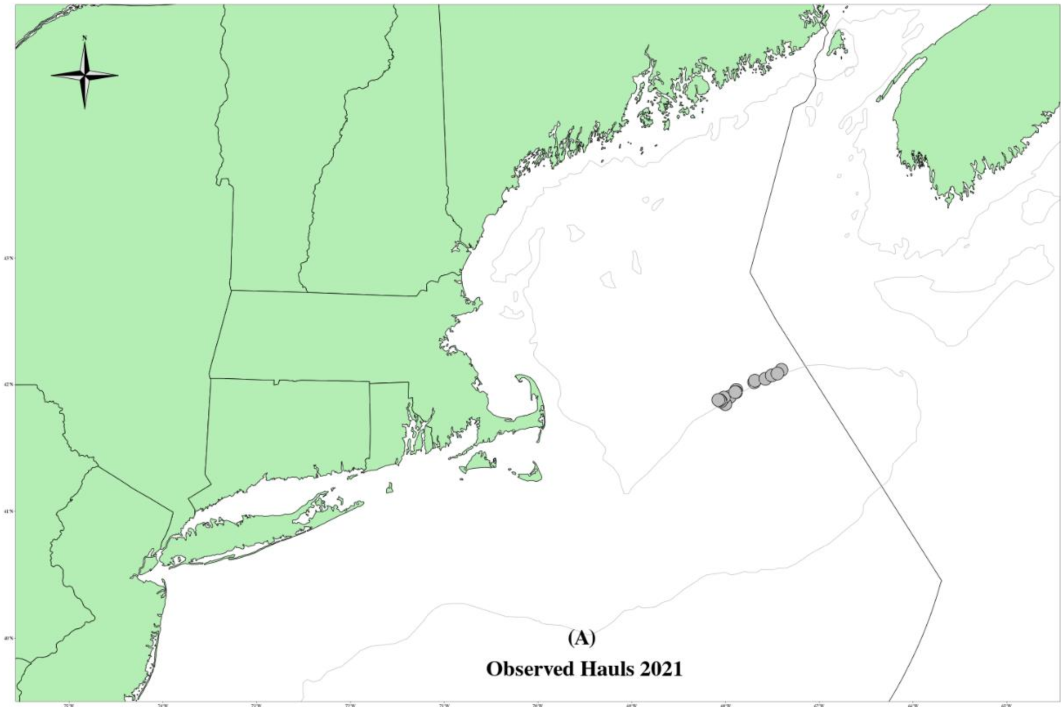


Figure 25. 2021 Northeast mid-water trawl observed tows (A) and observed takes (B).



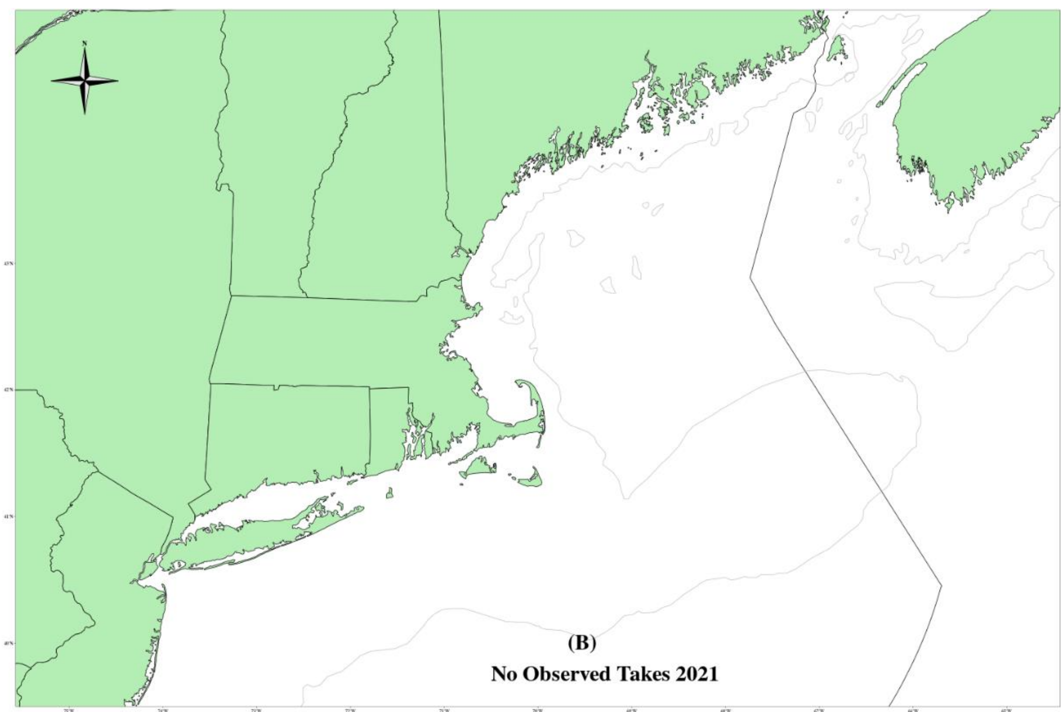


Figure 26. 2017 Mid-Atlantic mid-water trawl observed tows (A) and observed takes (B).

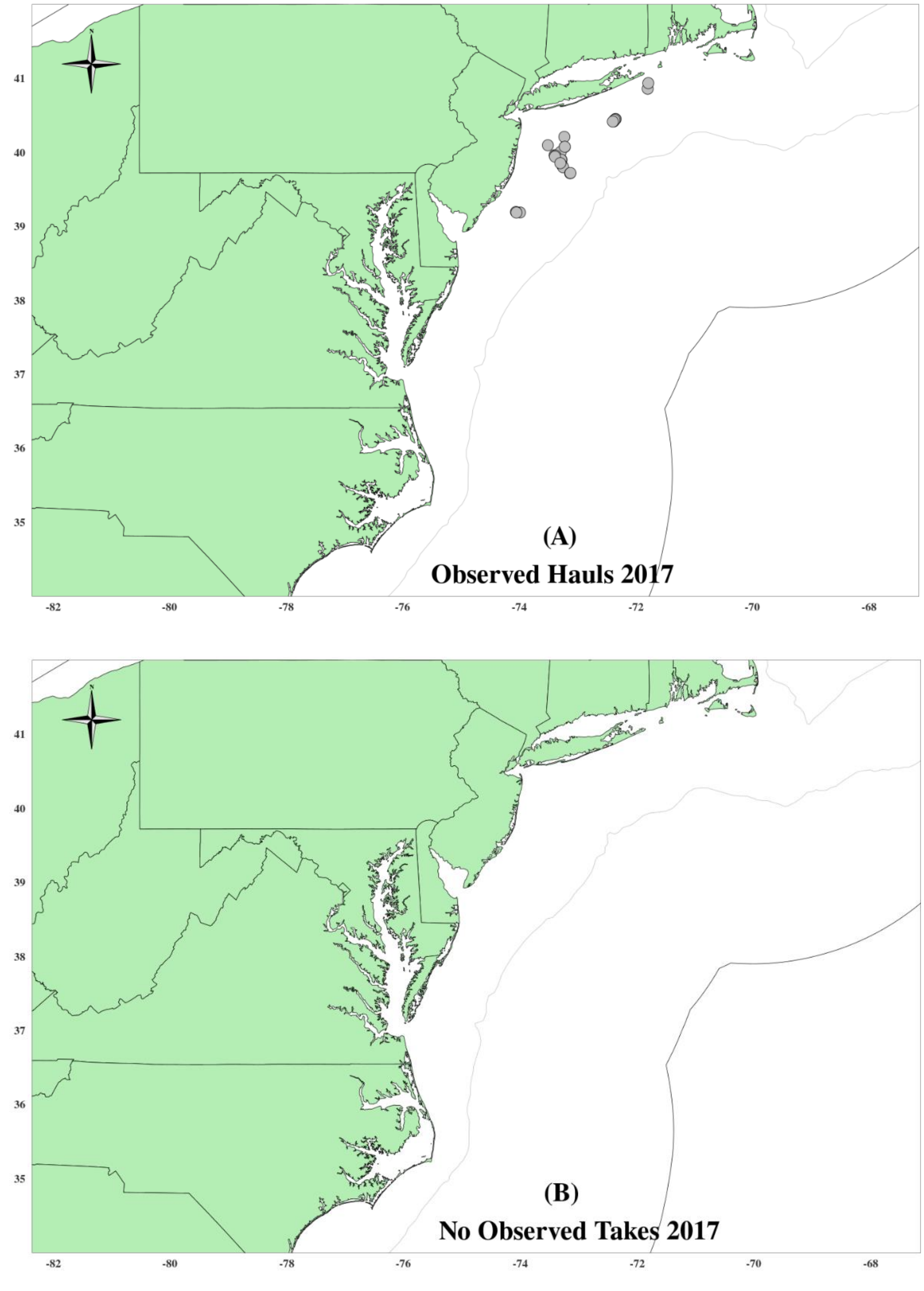
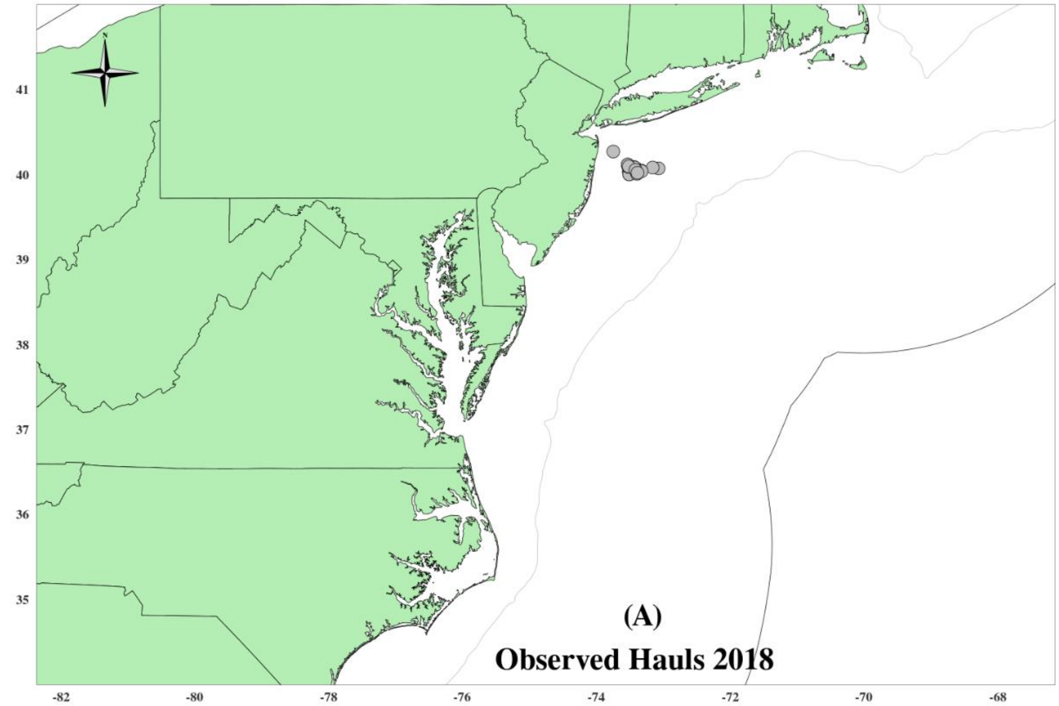


Figure 27. 2018 Mid-Atlantic mid-water trawl observed tows (A) and observed takes (B).



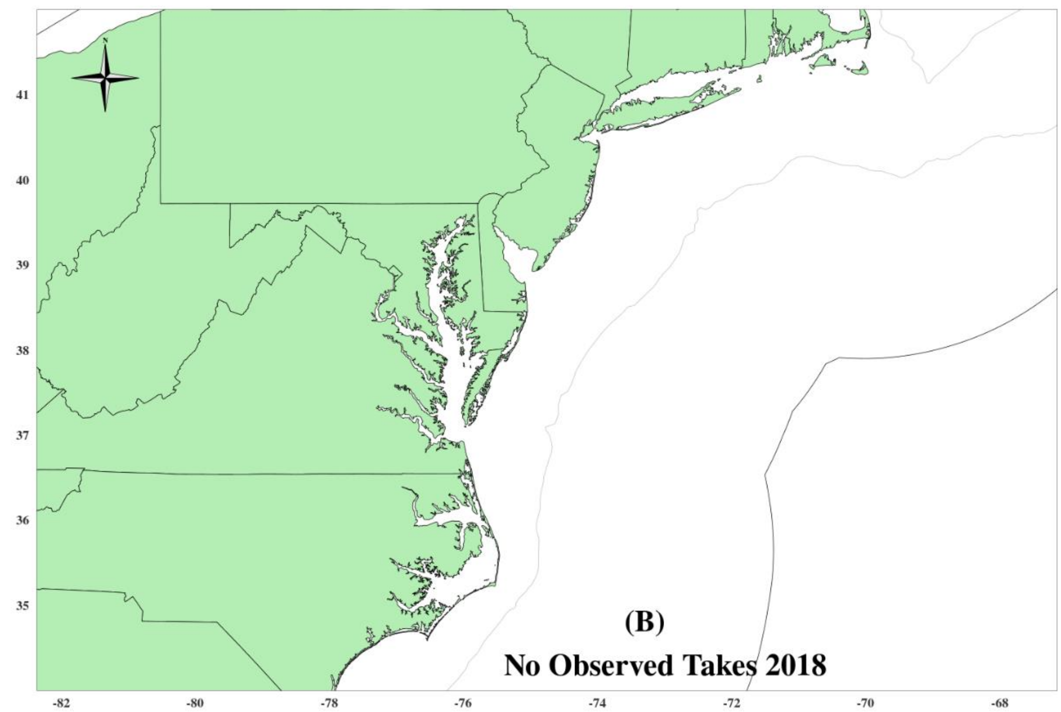
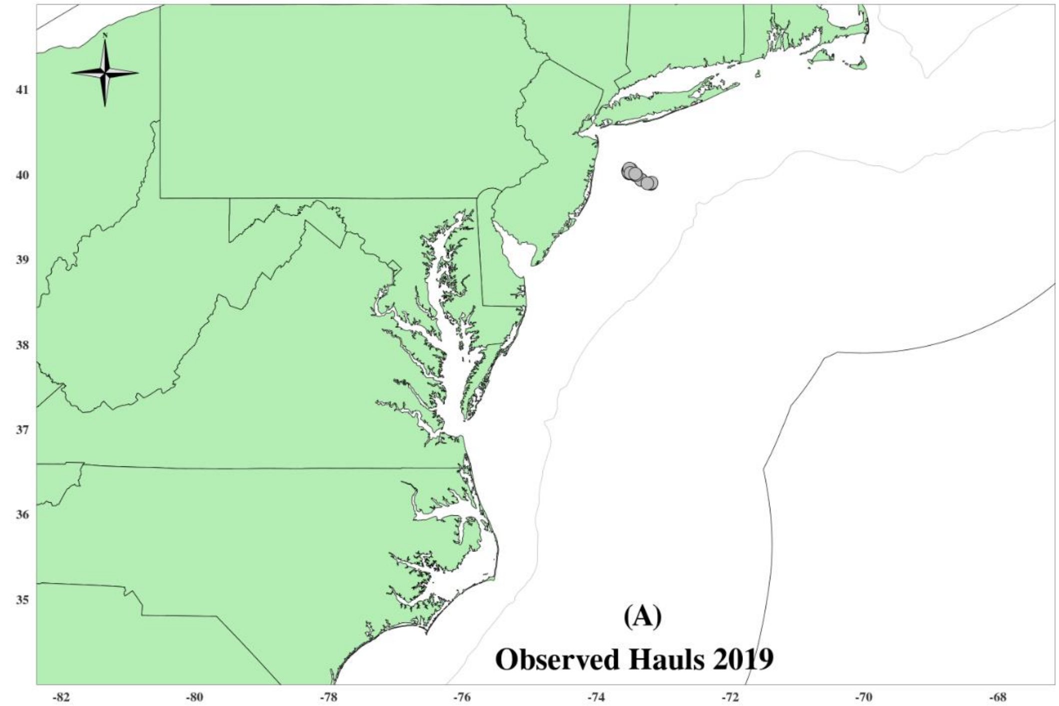


Figure 28. 2019 Mid-Atlantic mid-water trawl observed tows (A) and observed takes (B).



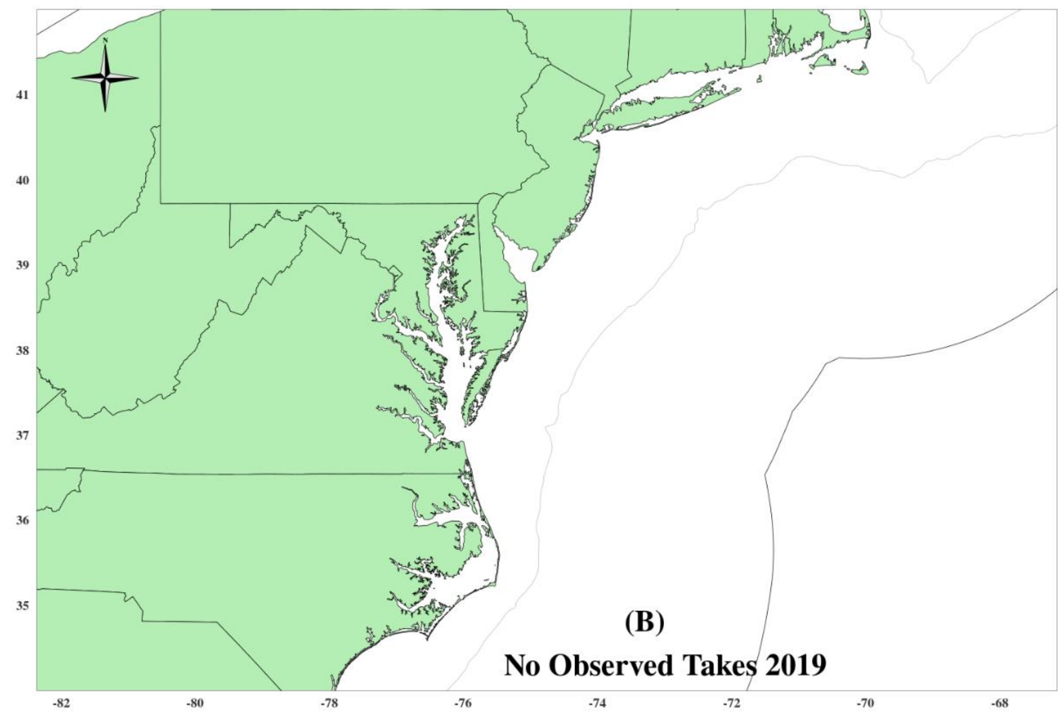
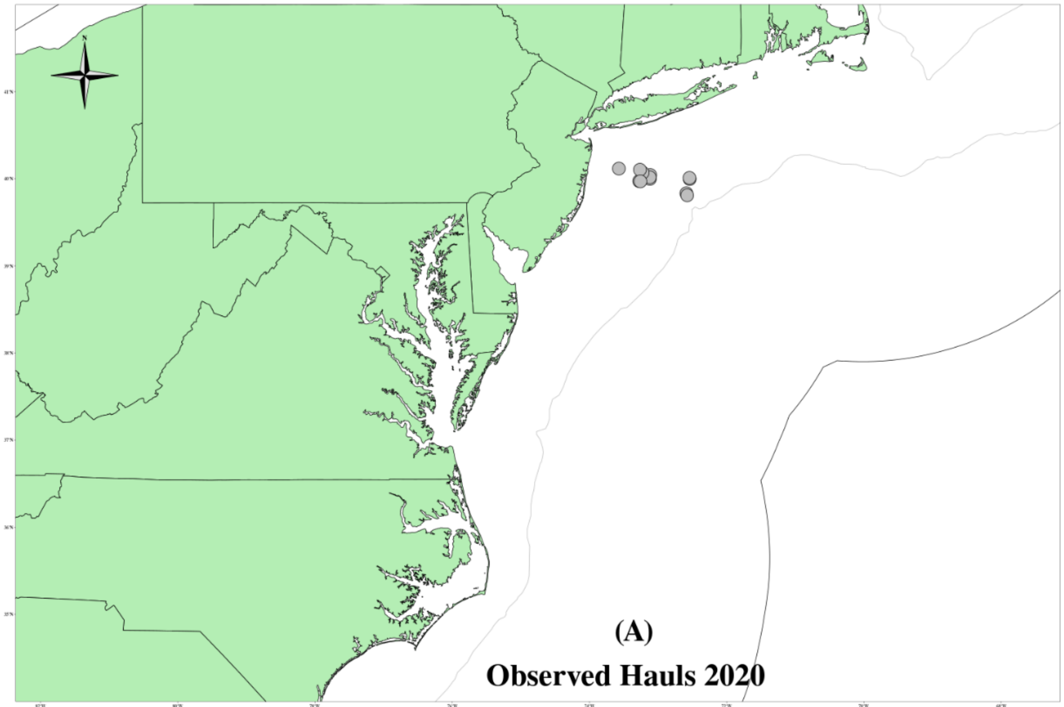


Figure 29. 2020 Mid-Atlantic mid-water trawl observed tows (A) and observed takes (B).



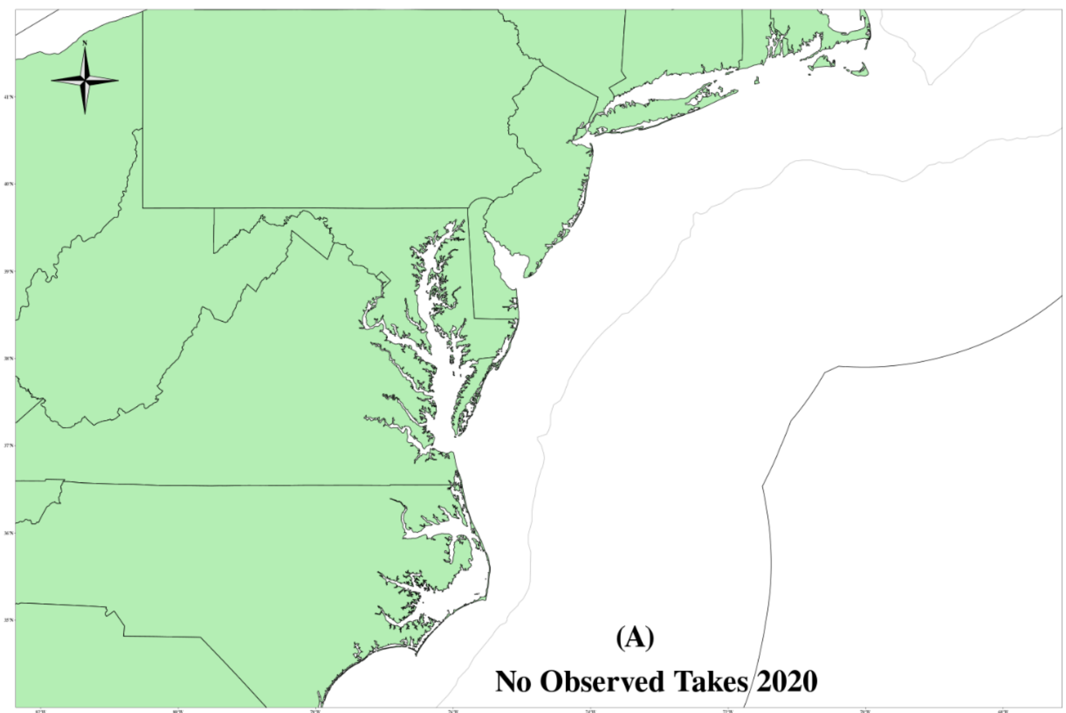
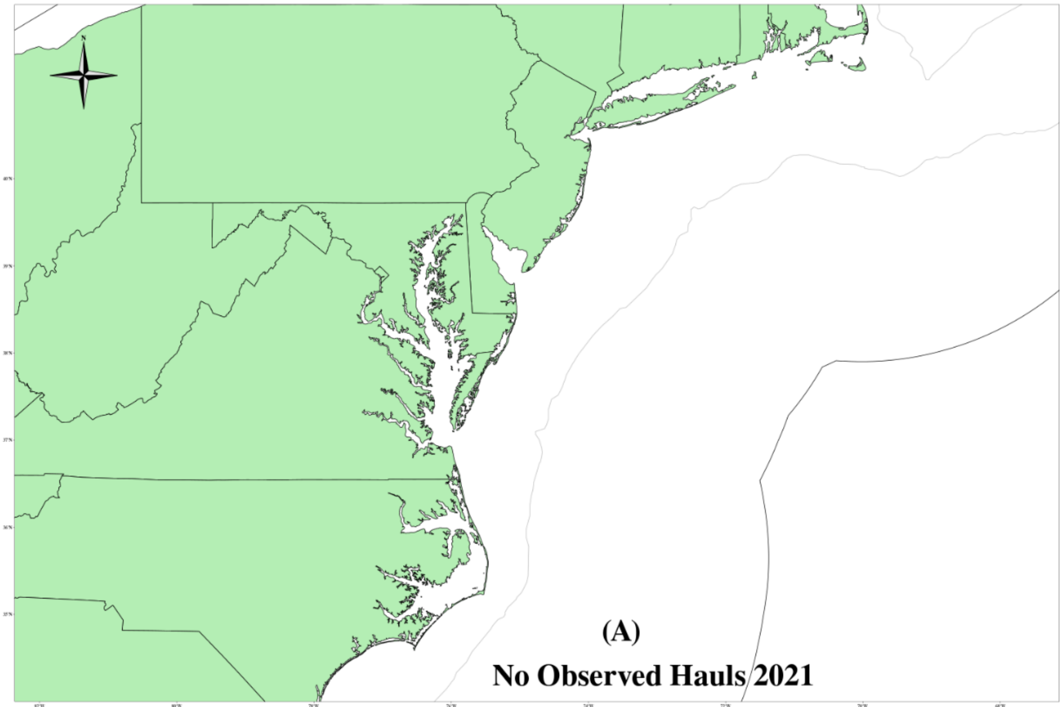


Figure 30. 2021 Mid-Atlantic mid-water trawl observed tows (A) and observed takes (B).



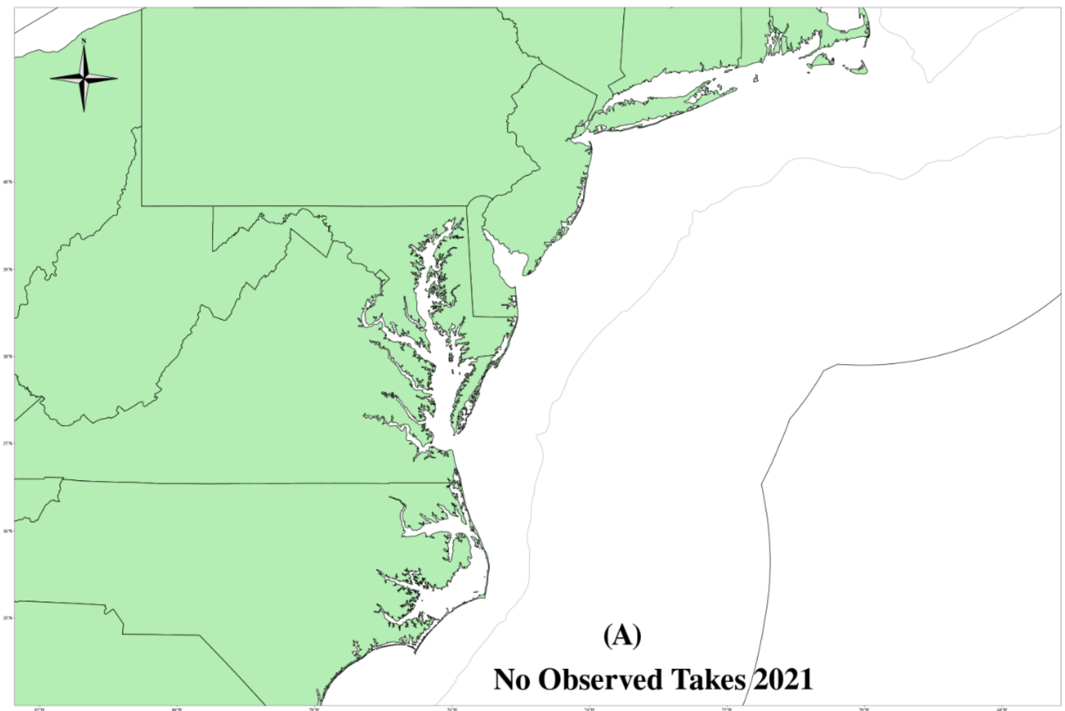
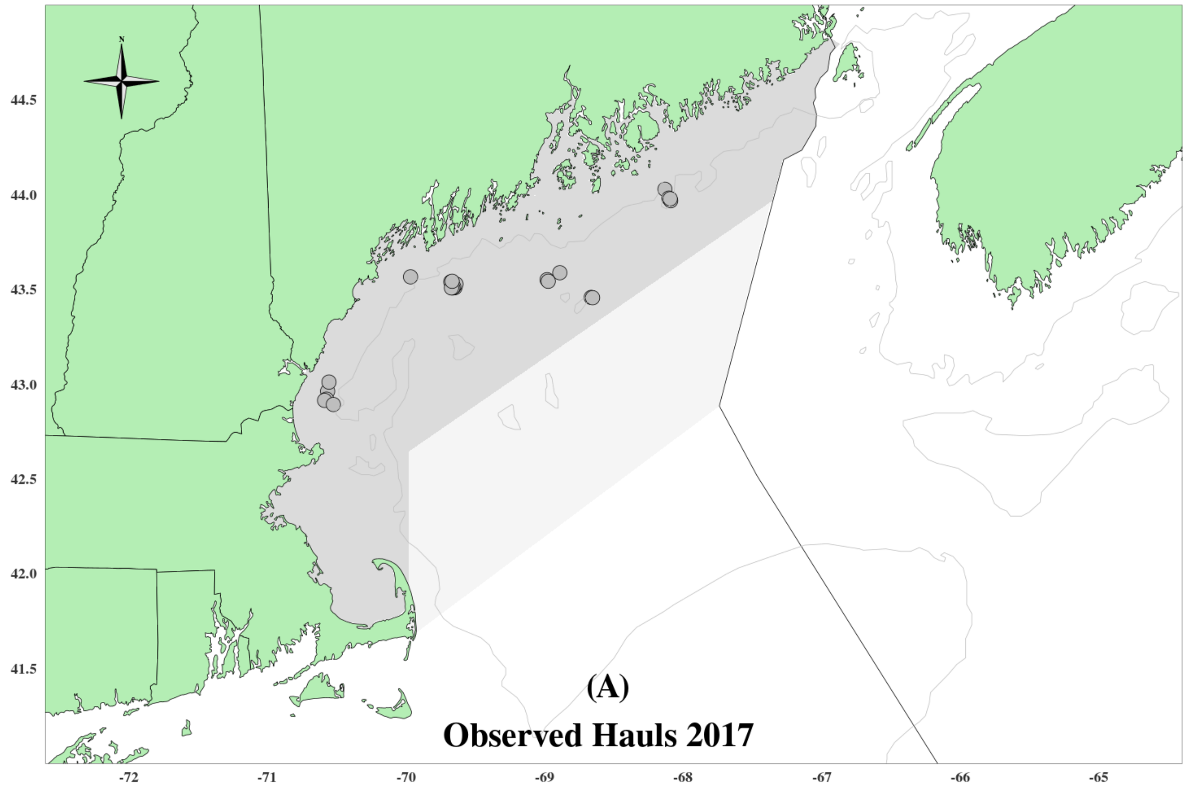


Figure 31. 2017 Herring Purse Seine observed hauls (A) and observed takes (B).



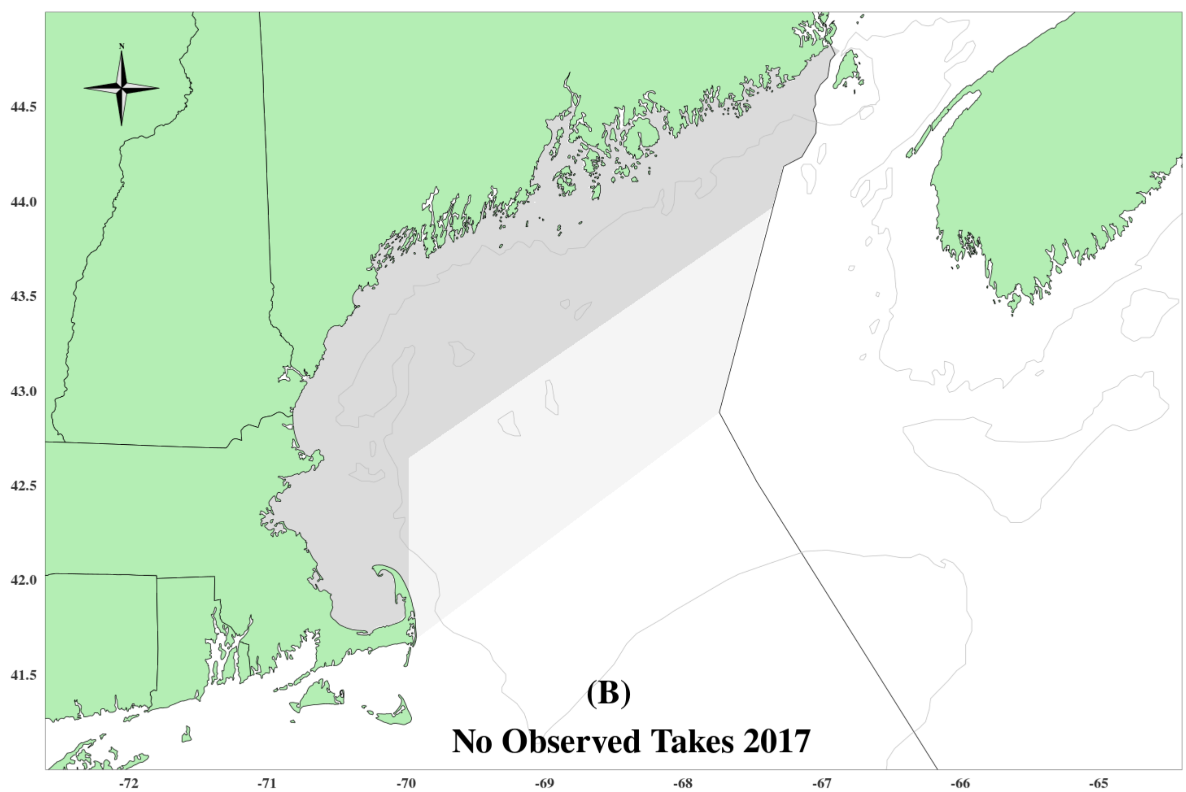


Figure 32. 2018 Herring Purse Seine observed hauls (A) and observed takes (B).

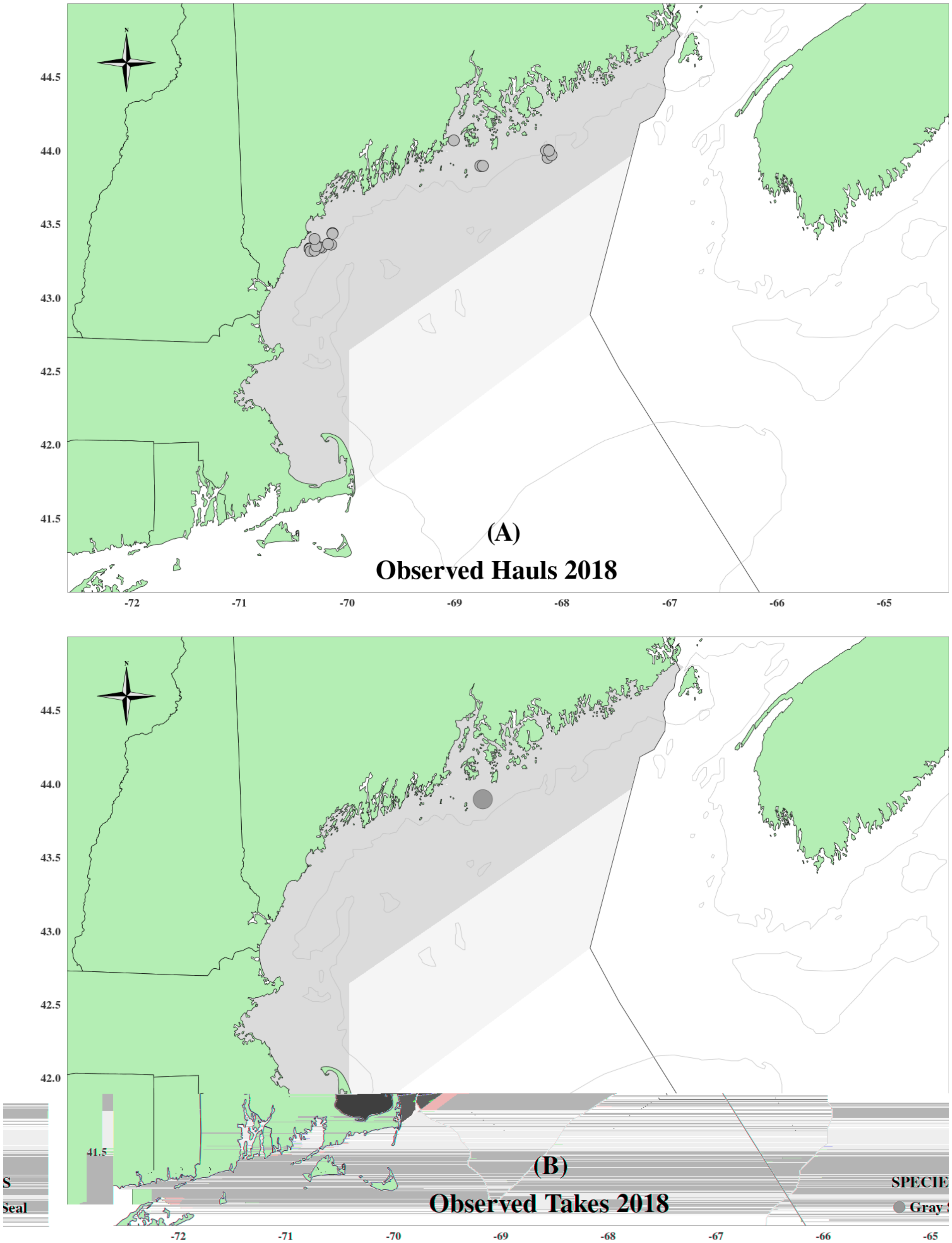
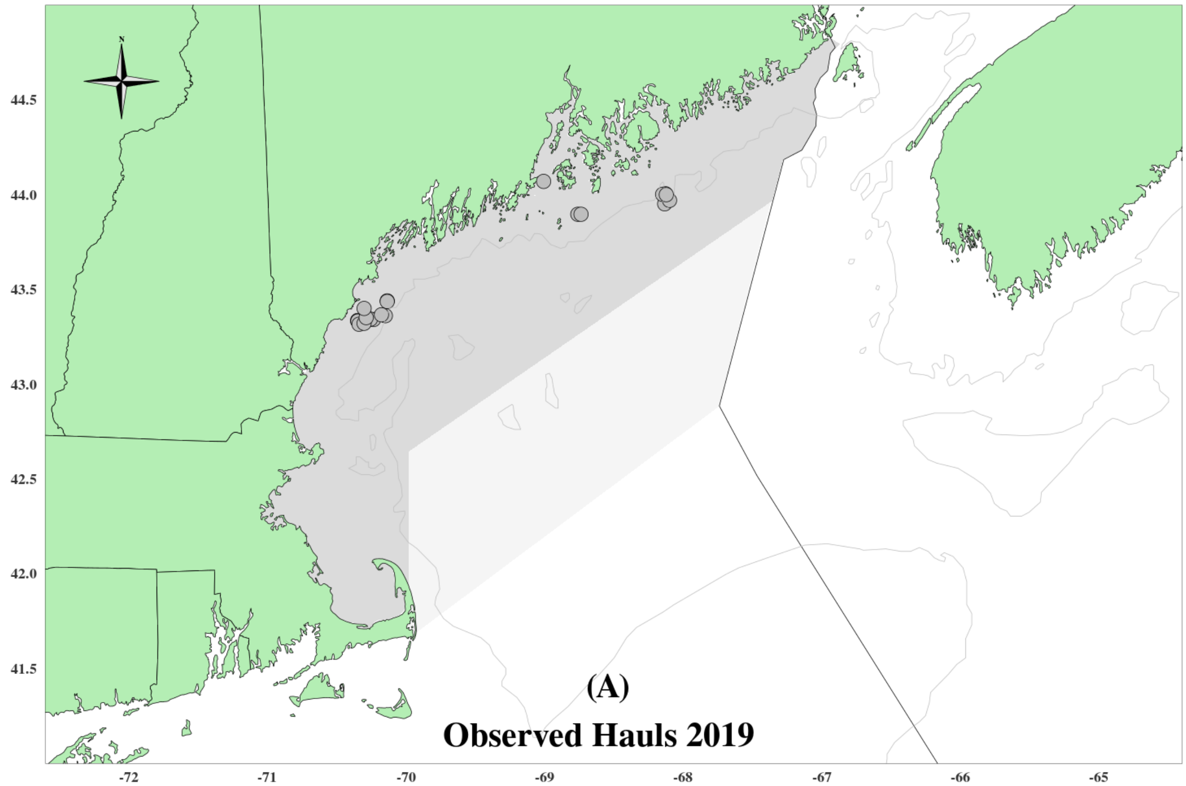


Figure 33. 2019 Herring Purse Seine observed hauls (A) and observed takes (B).



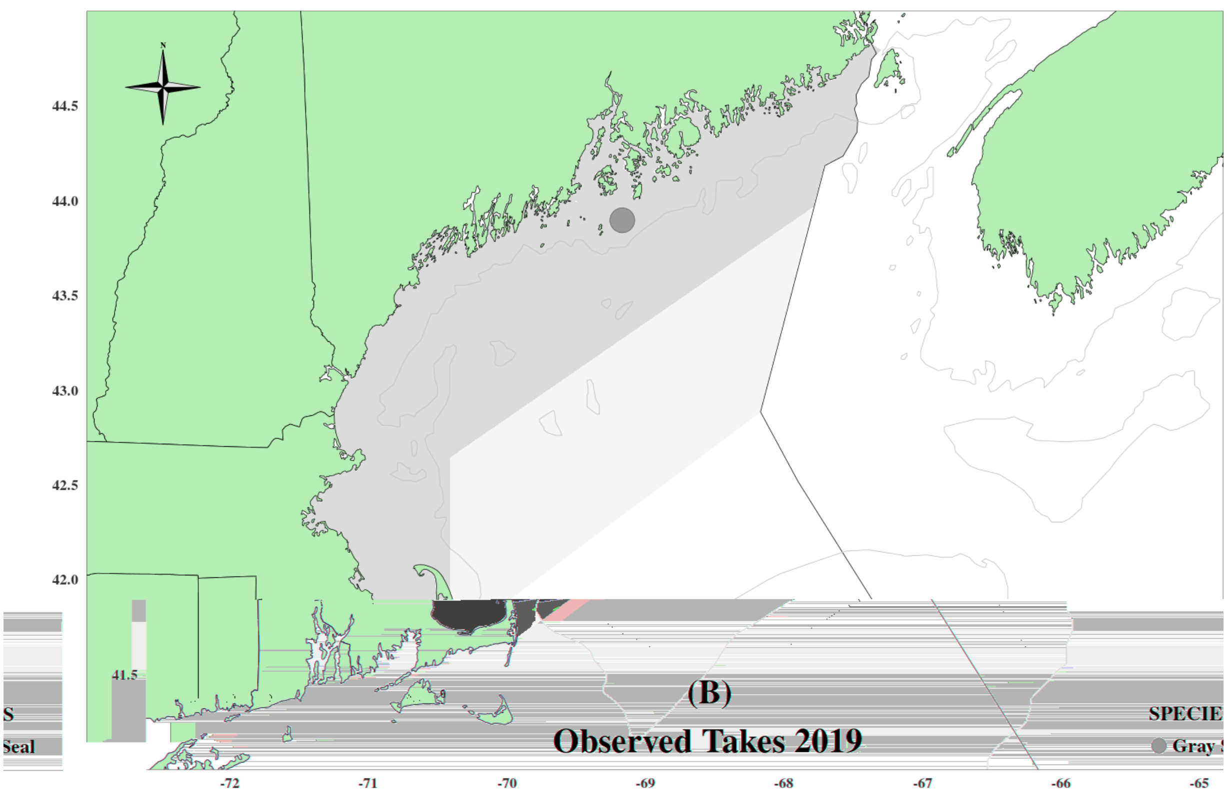
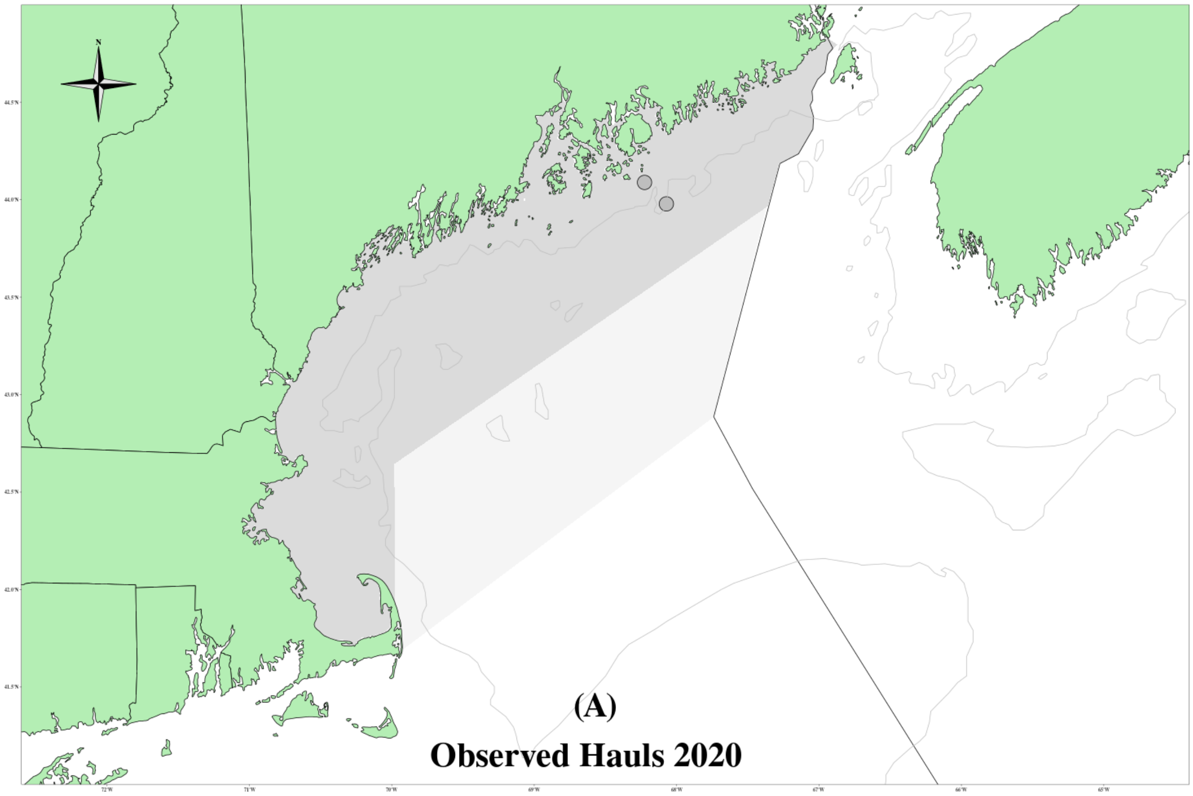


Figure 34. 2020 Herring Purse Seine observed hauls (A) and observed takes (B).



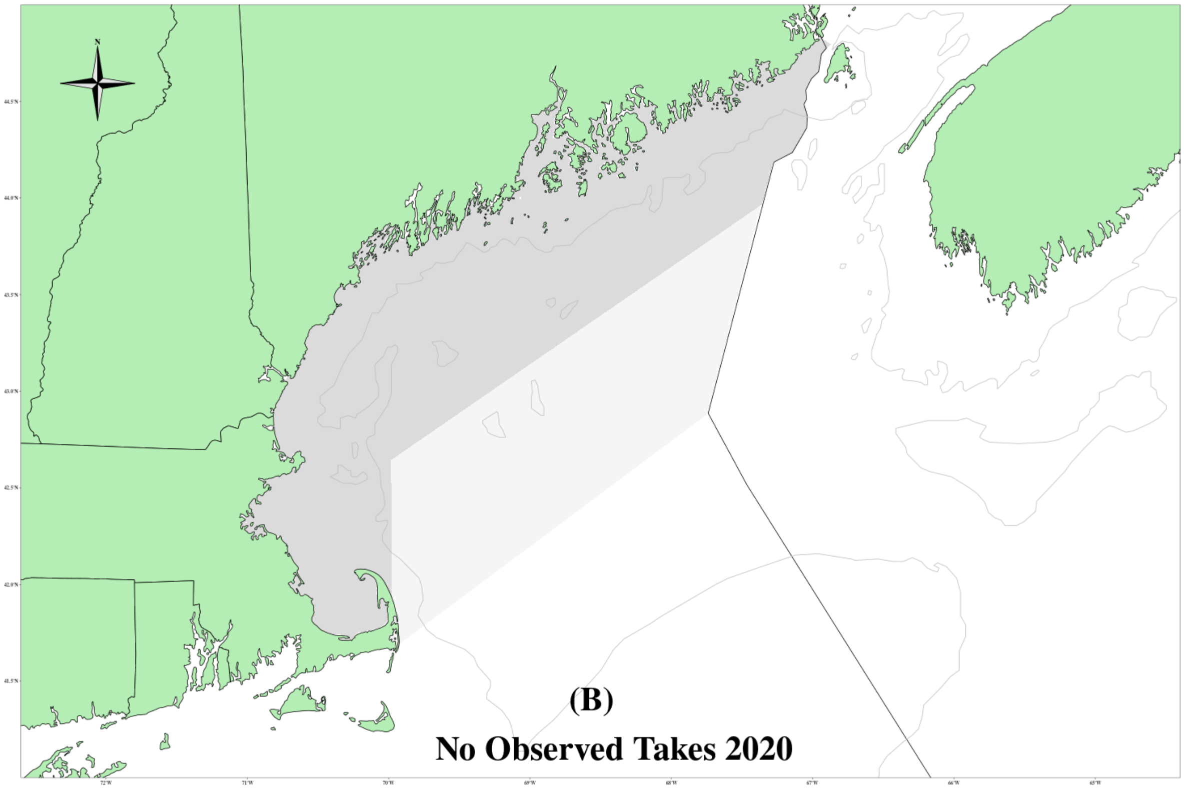
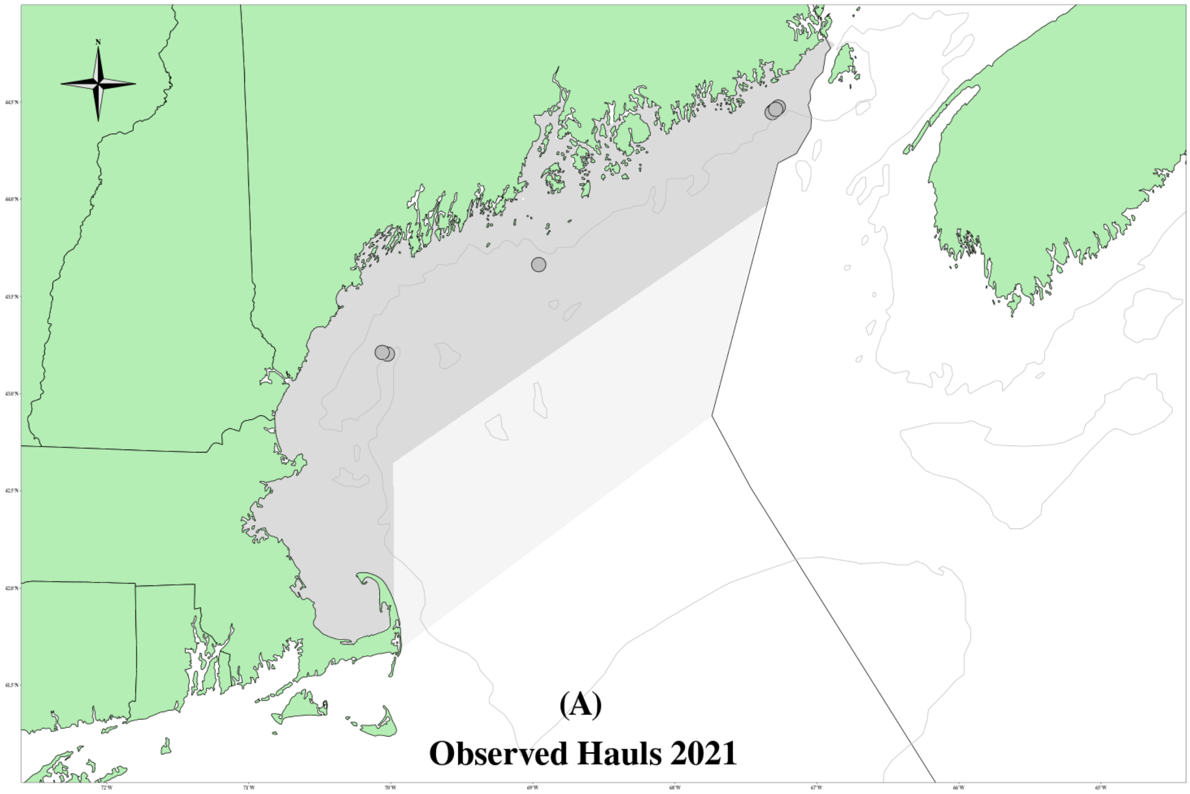


Figure 35. 2021 Herring Purse Seine observed hauls (A) and observed takes (B).



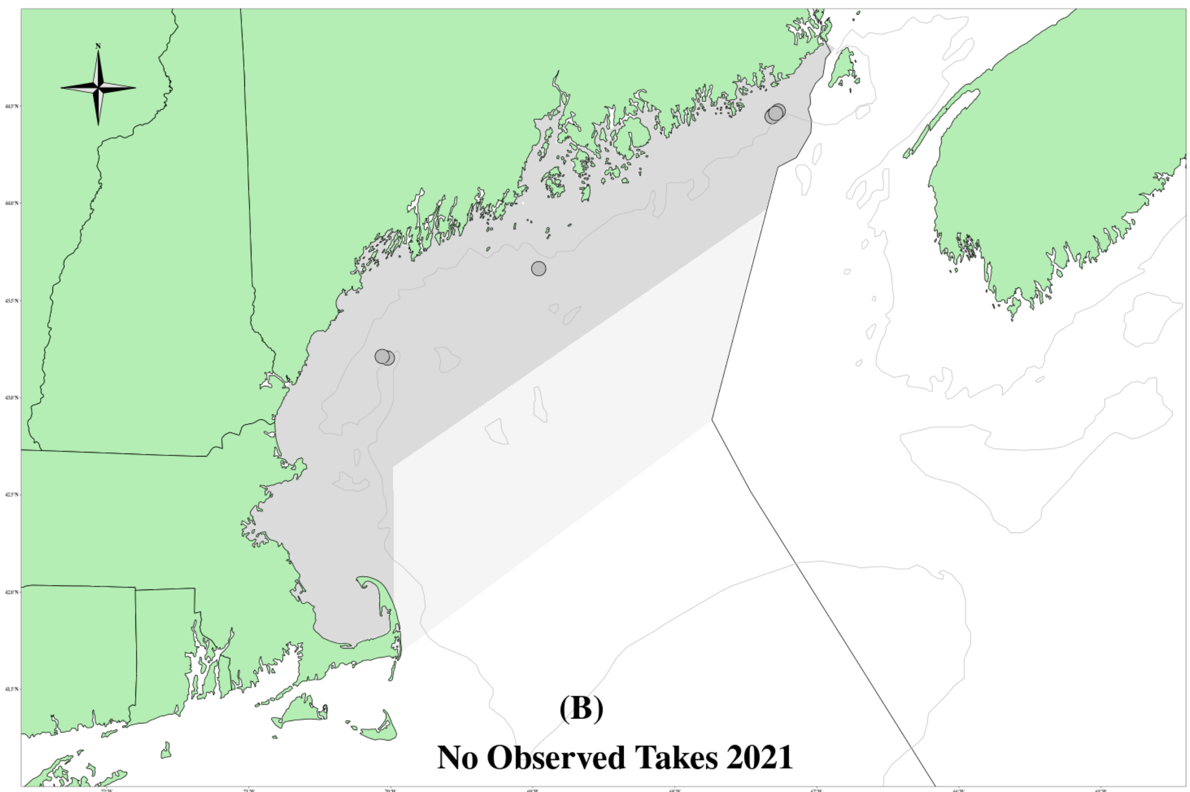


Figure 36. The location of marine mammal interactions and the number of observed sets in 2 degree latitude x 2 degree longitude grid cells in the U.S. Atlantic and Gulf of Mexico Pelagic Longline fishery during 2017. The boundary of the US Exclusive Economic Zone is indicated by a dashed line.

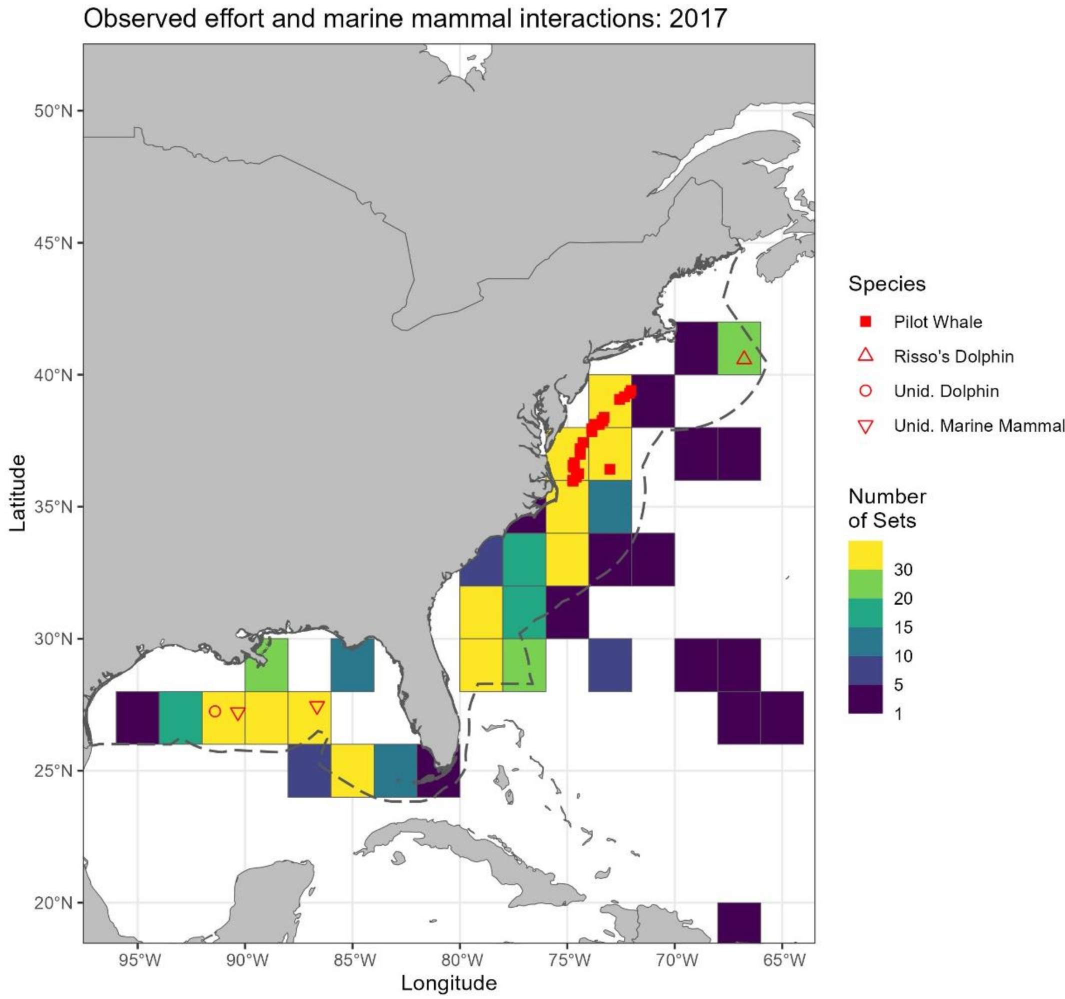


Figure 37. The location of marine mammal interactions and the number of observed sets in 2 degree latitude x 2 degree longitude grid cells in the U.S. Atlantic and Gulf of Mexico Pelagic Longline fishery during 2018. The boundary of the US Exclusive Economic Zone is indicated by a dashed line.

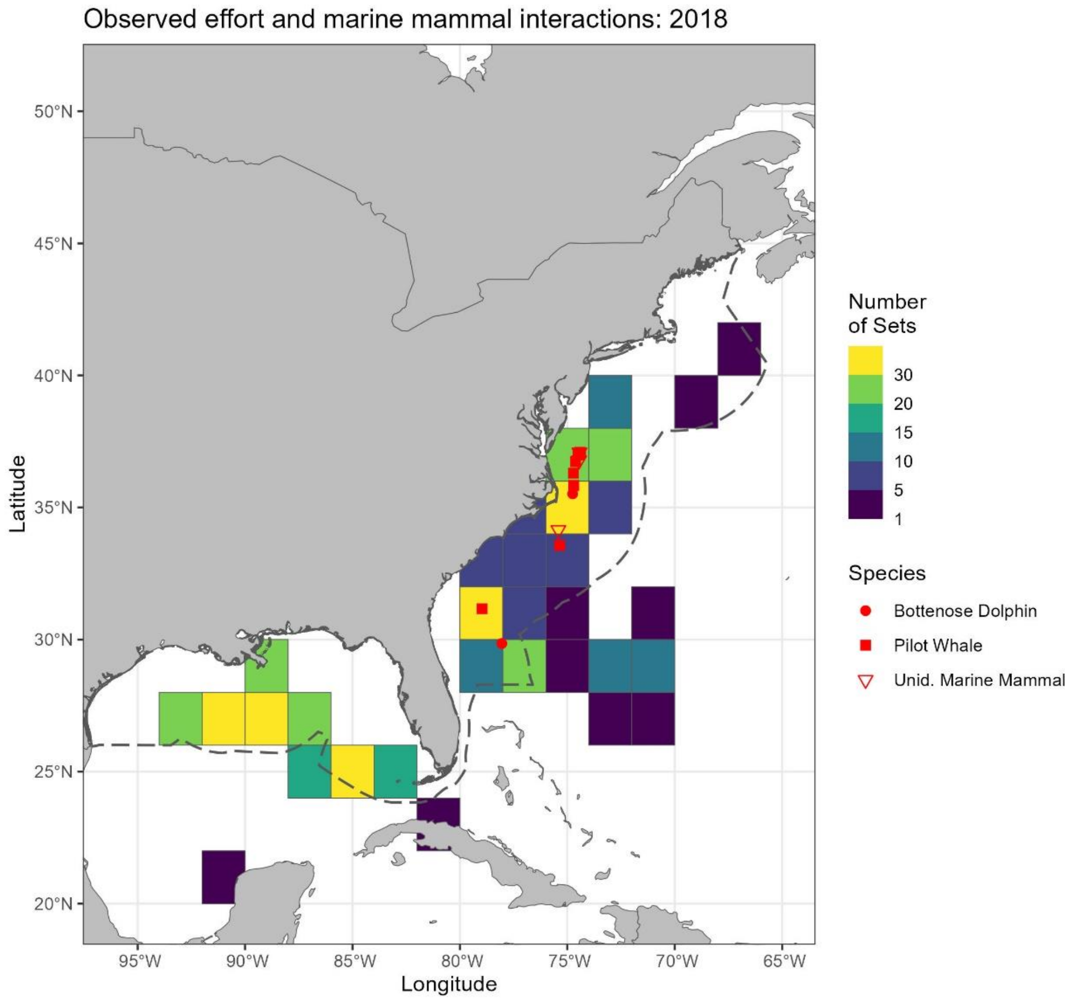


Figure 38. The location of marine mammal interactions and the number of observed sets in 2 degree latitude x 2 degree longitude grid cells in the U.S. Atlantic and Gulf of Mexico Pelagic Longline fishery during 2019. The boundary of the US Exclusive Economic Zone is indicated by a dashed line.

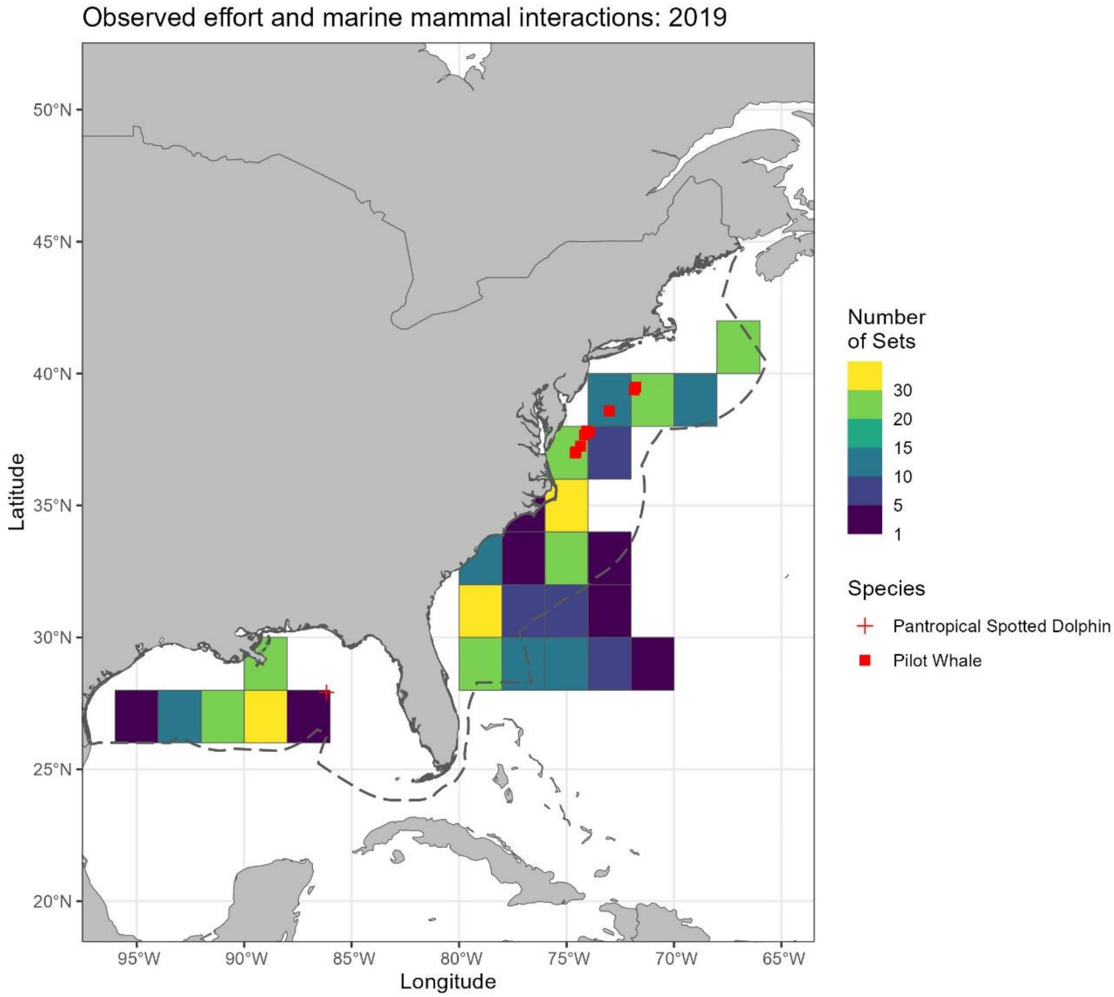


Figure 39. The location of marine mammal interactions and the number of observed sets in 2 degree latitude x 2 degree longitude grid cells in the U.S. Atlantic and Gulf of Mexico Pelagic Longline fishery during 2020. The boundary of the US Exclusive Economic Zone is indicated by a dashed line.

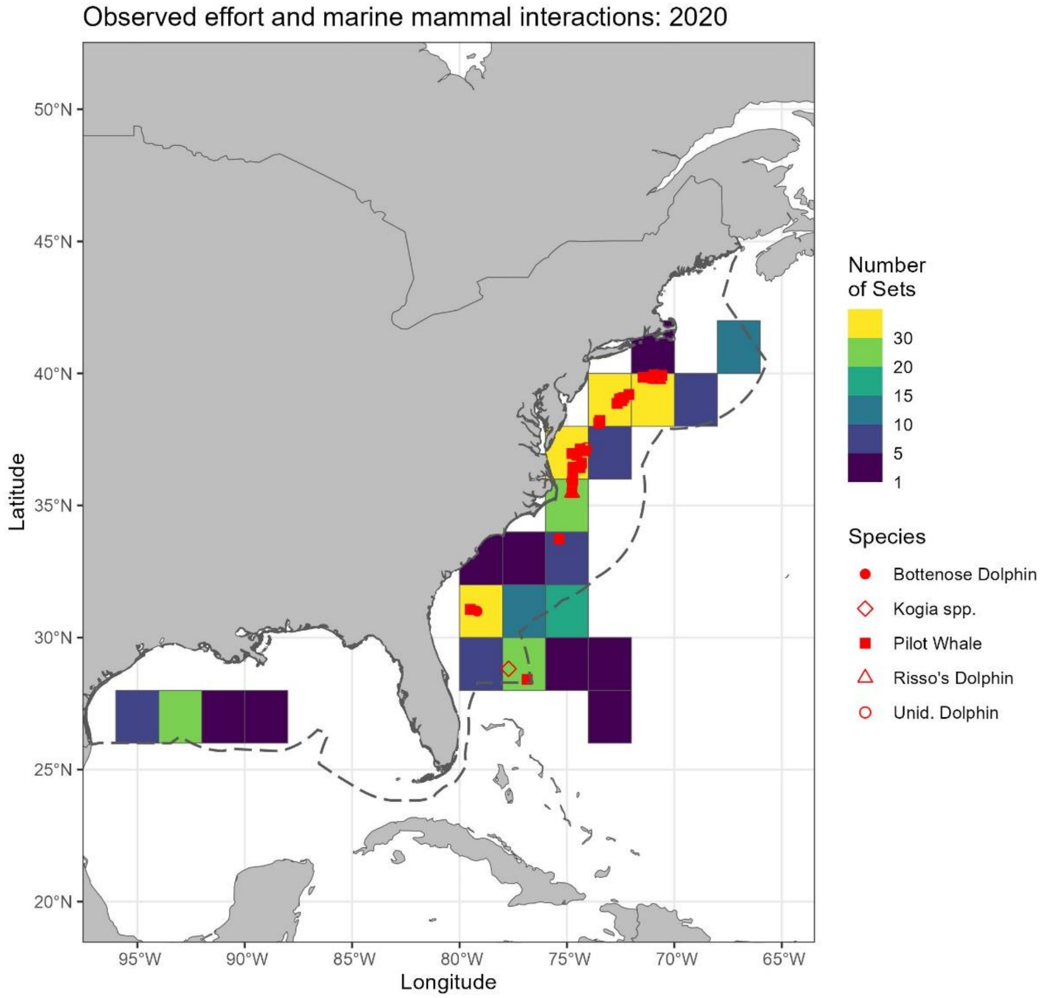
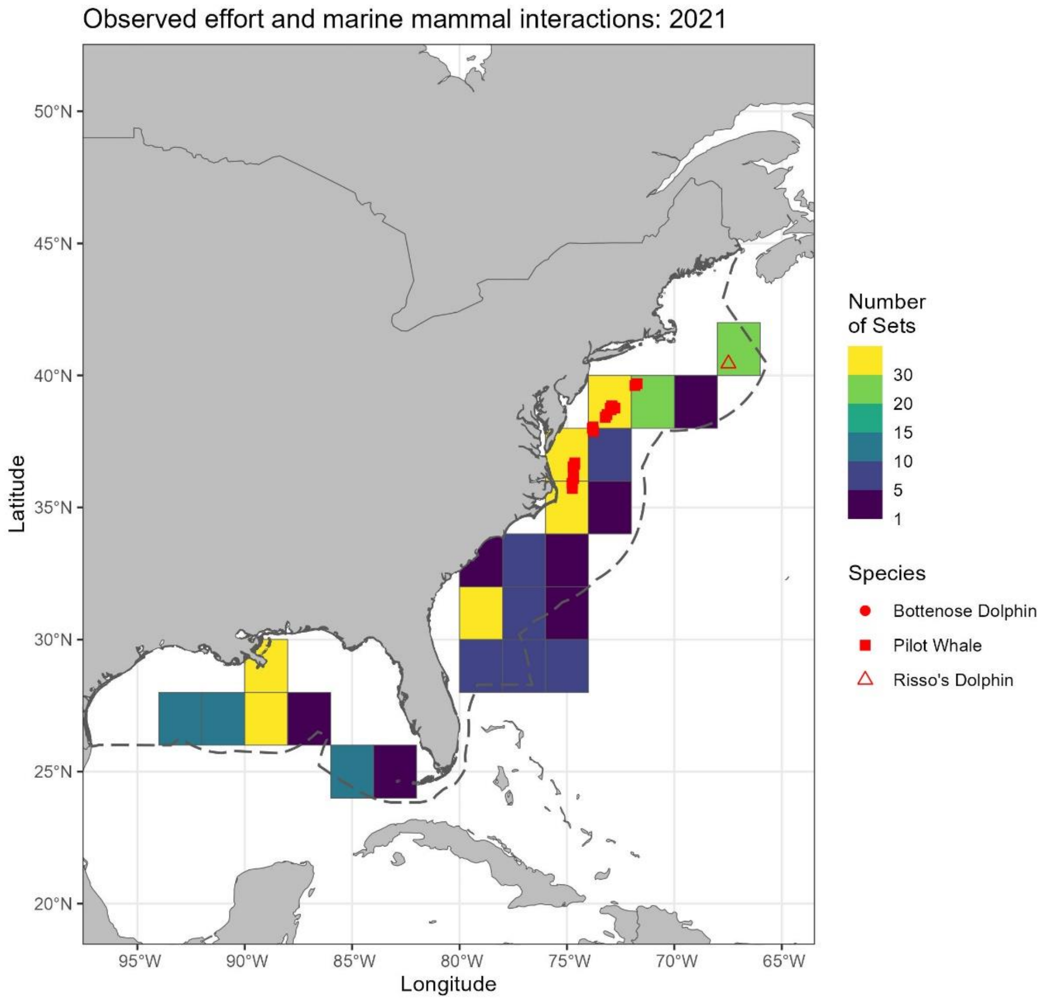


Figure 40. The location of marine mammal interactions and the number of observed sets in 2 degree latitude x 2 degree longitude grid cells in the U.S. Atlantic and Gulf of Mexico Pelagic Longline fishery during 2021. The boundary of the US Exclusive Economic Zone is indicated by a dashed line.



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
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

Survey Number	Year(s)	Time of Year	Platform	Track Line Length (km)	Area	Agency/ Program	Analysis	Corrected for g(0)	Reference(s)
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 (Twin Otter)	8,195	Georges Bank to Bay of Fundy	NEC	Ship: Tracker data analyzed by DISTANCE. Plane: Same as for plane in survey 17	Y	Palka 2005
25	2007	Jul–Aug	Plane	46,804	Nova Scotia to Newfoundland	DFO	Uncorrected counts	N	Lawson and Gosselin 2009
26	2008	Aug	Plane (Twin Otter)	6,267	New York to Maine	NEC	Same as for plane in survey 17	Y	Palka 2005
27	2001	May–Jun	Plane		Maine Coast	NEC, UM	Corrected counts	N	Gilbert et al. 2005
28	1999	Mar	Plane		Cape Cod	NEC	Uncorrected counts	N	Barlas 1999

Survey Number	Year(s)	Time of Year	Platform	Track Line Length (km)	Area	Agency/ Program	Analysis	Corrected for g(0)	Reference(s)
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		
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

Survey Number	Year(s)	Time of Year	Platform	Track Line Length (km)	Area	Agency/ Program	Analysis	Corrected for g(0)	Reference(s)
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
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

Survey Number	Year(s)	Time of Year	Platform	Track Line Length (km)	Area	Agency/ Program	Analysis	Corrected for g(0)	Reference(s)
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
52	2021	27Jun–23Aug 2021	Ship (Bigelow)	5,871 km	New Jersey to lower Bay of Fundy	NEC	Two-independent teams. Analyzed using DISTANCE, the independent observer option assuming point independence.	Y	NEFSC and SEFSC 2022
53	2021	12Jun–5Sep 2021	Ship (Gunter)	5,659 km	central Florida to New Jersey	SEC	Two-independent teams. Analyzed using DISTANCE, the independent observer option assuming point independence.	Y	NEFSC and SEFSC 2022

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
		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**	
		2021	2,240	0.39	52	Palka 2023
		2021	12	1.02	53	Garrison and Dias 2023
		2021	**2,252**	**0.39**	**52+53**	Garrison and Dias 2023; Palka 2023; Estimate summed from north and south surveys
Sei Whale	Nova Scotia Stock	1977	1,393–2,248			Based on tag-recapture data (Mitchell and Chapman 1977)
		1977	870			Based on census data (Mitchell and Chapman 1977)
		1982	280		1	CETAP 1982
		2002	71	1.01	18	Palka 2006
		2004	386	0.85	22	Palka 2006
		2006	207	0.62	23	Palka 2005
		2011	357	0.52	40+41	Palka 2012
		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

Species	Stock	Year	Nest	CV	Survey Number	Notes
						(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
		2021	34	0.99	52	Palka 2023
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**	
		2021	5,630	0.58	52	Palka 2023
Sperm Whale	North Atlantic	1982	219	0.36	1	CETAP 1982
		1990	338	0.31	2	
		1991	736	0.33	7	Waring et al.1992, Warring 1998
		1991	705	0.66	6	
		1991	337	0.5	5	
		1993	116	0.4	9	
		1994	623	0.52	10	
		1995	2,698	0.67	12	Palka 1996
		1998	2,848	0.49	16	
		1998	1,181	0.51	14	Mullin and Fulling 2003
		2004	2,607	0.57	22	Palka 2006
		2004	2,197	0.47	21	Garrison et al. 2010
		2004	4,804	0.38	21+22	Estimate summed from north and south surveys
		2011	1,593	0.36	40+41	Palka 2012
		2011	695	0.39	42	
		2011	2,288	0.28	40+41+42	Estimate summed from north and south surveys

Species	Stock	Year	Nest	CV	Survey Number	Notes
		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
		2021	3,789	0.38	52	Palka 2023
		2021	2,106	0.44	53	Garrison and Dias 2023
		2012	**5,895**	**0.29**	**52+53**	Garrison and Dias 2023; Palka 2023; 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
		2021	**9,474**	**0.36**	**52+53**	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
		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

Species	Stock	Year	Nest	CV	Survey Number	Notes
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
		2021	**1,742**	**0.39**	**52**	Palka 2023
		2021	**2,928**	**0.31**	**53**	Garrison and Dias 2023
		2021	**4,670**	**0.24**	**52+53**	Garrison and Dias 2023; Palka 2023; 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**	
		2021	**39,612**	**0.5**	**52**	Palka 2023
		2021	**4,455**	**0.45**	**53**	Garrison and Dias 2023
		2021	**44,067**	**0.45**	**52+53**	Garrison and Dias 2023; Palka 2023; Estimate summed from north and south surveys
Pilot Whale	Western North Atlantic	1951	50,000			Derived from catch data from 1951–1961 drive fishery (Mitchell 1974)
		1975	43,000–96,000			Derived from population models (Mercer 1975)
		1982	11,120	0.29	1	CETAP 1982
		1991	3,636	0.36	7	Waring et al.1992, Warring 1998
		1991	3,368	0.28	5	

Species	Stock	Year	Nest	CV	Survey Number	Notes
		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
		2021	**18,726**	**0.33**	**52+53**	Best estimate for short-finned pilot whales alone; Estimate summed from north and south surveys
		2021	**5,711**	**0.62**	**52+53**	Best estimate for long-finned pilot whales alone; Estimate summed from north and south surveys
Atlantic White-sided Dolphin	Western North Atlantic	1982	28,600	0.21	1	
		1992	20,400	0.63	2+7	
		1993	729	0.47	9	
		1995	27,200	0.43	12	Palka 1996
		1995	11,750	0.47	11	
		1996	560	0.89	13	
		1999	51,640	0.38	17	
		2002	109,141	0.3	18	Palka 2006
		2004	2,330	0.8	22	Palka 2006
		2006	17,594	0.3	23	

Species	Stock	Year	Nest	CV	Survey Number	Notes
		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**	
		2021	4,632	0.55	52	Palka 2023; less of the range covered than in 2016
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
		2021	**3,722**	**0.67**	**52**	Palka 2023
		2021	**15,004**	**0.38**	**53**	Garrison and Dias 2023
		2021	**18,726**	**0.33**	**52+53**	Garrison and Dias 2023; Palka 2023; Estimate summed from north and south surveys
	Western North Atlantic	1982	6,107	0.27	1	CETAP 1982
		1995	4,772	1.27	12	Palka 1996

Species	Stock	Year	Nest	CV	Survey Number	Notes
Atlantic Spotted Dolphin		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
		2021	**31,506**	**0.28**	**52+53**	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
		2021	**2,757**	**0.50**	**52+53**	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
	Western North Atlantic	2011	0	0	40+41	Palka 2012
		2011	271	1	42	

Species	Stock	Year	Nest	CV	Survey Number	Notes
Rough-toothed Dolphin		**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
		2021	**64,587**	**0.24**	**52+53**	Estimate summed from north and south surveys
Bottlenose Dolphin	Western North Atlantic Northern Migratory Coastal	2016	6,639	0.41	48	Garrison et al. 2017
Bottlenose Dolphin	Western North Atlantic Southern Migratory Coastal	2016	3,751	0.60	48	Garrison et al. 2017
Bottlenose Dolphin	Western North Atlantic South Carolina/Georgia Coastal	2016	6,027	0.34	48	Garrison et al. 2017
Bottlenose Dolphin	Western North Atlantic Northern Florida Coastal	2016	877	0.49	48	Garrison et al. 2017
Bottlenose Dolphin	Western North Atlantic Central Florida Coastal	2016	1,218	0.35	48	Garrison et al. 2017
Bottlenose Dolphin	Western North Atlantic Bay, Sound and Estuarine (10 stocks)	Northern North Carolina Estuarine System (2013)	823	0.06		Gorgone et al. 2014
		Southern North Carolina Estuarine System (2006)	188	0.19		Urian et al. 2013
		Northern South Carolina Estuarine System (2016)	453	0.28		Silva et al. 2019
		Charleston Estuarine System (2005–2006)	289	0.03		Speakman et al. 2010
		Northern Georgia/Southern South Carolina Estuarine System	unknown	-		

Species	Stock	Year	Nest	CV	Survey Number	Notes
		Central Georgia Estuarine System (2008–2009)	unknown	-		
		Southern Georgia Estuarine System (2008–2009)	unknown	-		
		Jacksonville Estuarine System	unknown	-		
		Indian River Lagoon Estuarine System (2016–2017)	1,032	0.03		Durden et al. 2021
		Biscayne Bay	unknown	-		
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
		2021	**85,765**	**0.53**	**52**	Palka 2023
Harbor Seal	Western North Atlantic	2001	99,340	0.097	27	Gilbert et al. 2005
		2012	**75,834**	**0.15**	**43**	Waring et al. 2015
Gray Seal	Western North Atlantic	1999	5,611		28	Barlas 1999
		2001	1,731		27	Gilbert et al. 2005
		2004	52,500	0.15	37	Gulf of St Lawrence and Nova Scotia Eastern Shore
		2004	208,720–223,220	0.08–0.14	36	Sable Island
		2012	331,000	95%CI= 263,000-458,000		Gulf of St Lawrence + Nova Scotia Eastern Shore + Sable Island (DFO 2013)
		2014	505,000	95%CI= 329,000–682,000		Gulf of St Lawrence + Nova Scotia Eastern Shore + Sable Island (DFO 2014)
		2016	424,300	95%CI= 263,600–578,300		Gulf of St Lawrence + Nova Scotia Eastern Shore + Sable Island (DFO 2017)
		2016	27,131	95%CI= 18,768–39,221		Derived from total population size to pup ratios in Canada applied to U.S. pup counts
		2021	**366,400**	**317,800-409,400**		Gulf of St. Lawrence and Scotian Shelf; DFO 2022

Species	Stock	Year	Nest	CV	Survey Number	Notes
		2021	**27,911**	**NA**		US portion of the population; Wood et al. 2022
Rice'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
		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	

Species	Stock	Year	Nest	CV	Survey Number	Notes
		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	
		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
	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

Species	Stock	Year	Nest	CV	Survey Number	Notes
Atlantic Spotted Dolphin		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
Bottlenose Dolphin	Northern Gulf of Mexico: Coastal (3 stocks)	Eastern (1994)	9,912	0.12	32	
		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)

Species	Stock	Year	Nest	CV	Survey Number	Notes
		Western (2017–2018)	**20,759**	**0.13**	**51**	Garrison et al. 2021
Bottlenose Dolphin	Northern Gulf of Mexico: Bay, Sound and Estuarine (32 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
		Florida Bay	unk	-		
		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	-	3	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
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	195	0.22	2	0.99	0	0	5	0.68	5	0.7	0	0	316	0.15	2019	0.17	163	0.19
2020	121	0.22	1	0.99	0	0	50	0.25	2	1.01	0	0	261	0.14	1357	0.14	72	0.22
2021	111	0.19	1.4	0.99	2	0	39	0.24	3	0	0	0	241	0.13	1027	0.14	66	0.24

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		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**
1994	0	0	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	0	0
1996	311	0.31	64	0.83	0	0	43	0.79	0	0	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	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	0	0
1999	53	0.49	0	0	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	0	0
2001	26	0.95	na	na	0	0	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	0	0
2003	76	1.13	0	0	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	0	0
2005	470	0.51	1[a]	na	0	0	0	0	0	0	0	0	63	0.67	0	0	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	0	0
2007	58	1.03	0	0	0	0	0	0	34	0.73	0	0	0	0	0	0	38	0.9	0	0
2008	350	0.75	0	0	0	0	0	0	0	0	0	0	88	0.74	0	0	176	0.74	0	0
2009	201	0.55	0	0	0	0	0	0	0	0	0	0	47	0.68	0	0	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	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	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	0	0
2013	19	1.06	26	0.95	0	0	62	0.67	0	0	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	0	0
2015	60	1.16	0	0	0	0	30	0.55	0	0	0	0	48	0.52	15	1.04	0	0	0	0
2016	23	0.64	0	0	0	0	7	0.97	0	0	0	0	18	0.95	7	0.93	0	0	0	0
2017	9	0.95	0	0	0	0	22	0.71	0	0	0	0	3	0.62	0	0	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	0	0
2019	13	0.51	0	0	0	0	20	0.56	0	0	0	0	17	0.35	18	0.4	29	.84	0.2[a]	na
2020	16	0.63	0	0	0	0	30	0.55	0	0	0	0	9	0.43	9	0.72	2	1.01	0	0
2021	10	0.65	0	0	0	0	24	0.33	0	0	0	0	9	0.40	7	0.69	2	1.01	0	0

Note: This table only includes observed bycatch. For a complete list of marine mammal species interactions with this fishery, please see https://www.fisheries.noaa.gov/national/marine-mammal-protection/mid-atlantic-gillnet-fishery-mmpa-list-fisheries. For bottlenose dolphin stocks not listed in this table (Northern Migratory Coastal Stock, Southern Migratory Coastal Stock, Northern NC Estuarine Stock, Southern NC Estuarine Stock), see Lyssikatos & Garrison 2018 and Lyssikatos 2021.

[a] Unextrapolated mortalities

na=not applicable; unk= observer coverage was absent or too low to detect bycatch, or no estimate generated; tbd= to be determined

New England/North Atlantic Bottom Trawl

	Harbor Porpoise		Bottlenose Dolphin, Atlantic Offshore Stock		White-sided Dolphin		Common Dolphin		Risso's Dolphin, Atlantic		Pilot Whale, Unidentified		Long-finned Pilot Whale		Harbor Seal		Gray Seal		Harp Seal		Minke Whale	
Year	**M/SI**	**CV**	**M/SI**	**CV**	**M/SI**	**CV**	**M/SI**	**CV**	**M/SI**	**CV**	**M/SI**	**CV**	**M/SI**	**CV**	**M/SI**	**CV**	**M/SI**	**CV**	**M/SI**	**CV**	**M/SI**	**CV**
1990	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
1991	0	0	91	0.97	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
1992	0	0	0	0	110	0.97	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
1993	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
1994	0	0	0	0	182	0.71	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
1995	0	0	0	0	0	0	142	0.77	0	0	0	0	0	0	0	0	0	0	0	0	0	0
1996	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
1997	0	0	0	0	0	0	93	1.06	0	0	0	0	0	0	0	0	0	0	0	0	0	0
1998	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
1999	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
2000	0	0	0	0	137	0.34	27	0.29	0	0	0	0	0	0	0	0	0	0	0	0	0	0
2001	0	0	0	0	161	0.34	30	0.3	0	0	21	0.27	0	0	0	0	0	0	49	1.1	0	0
2002	0	0	0	0	70	0.32	26	0.29	0	0	22	0.26	0	0	0	0	0	0	0	0	0	0
2003	*	*	0	0	216	0.27	26	0.29	0	0	20	0.26	0	0	0	0	0	0	0	0	0	0
2004	0	0	0	0	200	0.30	26	0.29	0	0	15	0.29	0	0	0	0	0	0	0	0	0	0
2005	7.2	0.48	0	0	213	0.28	32	0.28	0	0	15	0.30	0	0	0	0	unk	unk	unk	unk	0	0
2006	6.5	0.49	0	0	40	0.50	25	0.28	0	0	14	0.28	0	0	0	0	0	0	0	0	0	0
2007	5.6	0.46	48	0.95	29	0.66	24	0.28	3	0.52	0	0	0	0	0	0	unk	unk	0	0	0	0
2008	5.6	0.97	19	0.88	13	0.57	6	0.99	2	0.56	0	0	21	0.51	0	0	16	0.52	0	0	2.9	0.73
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	10.8	0.63	5.6	0.92	79	0.28	10	0.62	0	0	5.4	0.88	6.9	0.51	5.4	0.88	30	0.37	5.4	0.89	0	0
2020	3.6	0.63	1.9	0.92	31	0.26	50	0.25	0	0	1.8	0.62	0	0	4.6	0.68	25.8	0.26	1.8	0.89	0	0
2021	5.0	0.92	3.7	0.86	15	0.52	43	0.42	3.8	0.88	0	0	0	0	0	0	7	0.60	0	0	0	0

Note: This table only includes observed bycatch. For a complete list of marine mammal species interactions with this fishery, please see https://www.fisheries.noaa.gov/national/marine-mammal-protection/northeast-bottom-trawl-fishery-mmpa-list-fisheries
na=not applicable; unk= observer coverage was absent or too low to detect bycatch, or no estimate generated; tbd= to be determined
[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	0	0	0	0	395	0.23	0	0	0	0	7.3	0.93	22	0.53
2020	9.5	0.55	0	0	333	0.14	18.4	0.51	0	0	4.3	0.67	34.7	0.35
2021	37.9	1.03	0	0	230	0.57	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-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
2020	0	0	0	0	0	0	0	0	0	0	0	0
2021	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
2020	0	0	0	0	0	0	0	0	0	0
2021	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
2019	12.9	1	0	0	0	0	0	0	0	0	0.4	1	131	0.37	0.3	1
2020	1	1	10.2	0.73	0	0	12.2	0.71	0	0	5.7	0.44	371	0.45	0	0
2021	0	0	15.8	1.00	0	0	0	0	0	0	2.8	0.66	355	0.31	0	0

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	0.52		
2018	0	0	0	0	92	0.23		
2019	13	0.51	10.8	0.63	195	0.22		
2020	16	0.63	3.6	0.63	121	0.22		
2021	10	0.65	5.0	0.92	111	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

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
2020	9.5	0.55	0	0	1.9	0.92	2	0.99			10.2	0.73
2021	37.9	1.03	0	0	3.7	0.86	1.4	0.99			15.8	1.00

Note: This table only includes observed bycatch.

[a] Unextrapolated mortalities

na = not applicable; unk = observer coverage was absent or too low to detect bycatch, or no estimate generated; tbd = to be determined

White-sided Dolphin

	Mid-Atlantic Bottom Trawl		Mid-Atlantic Gillnet		Mid-Atlantic Midwater Trawl		North Atlantic Bottom Trawl		NE Sink Gillnet		Northeast Midwater Trawl		Pelagic Drift Gillnet	
Year	**M/SI (est)**	**CV**	**M/SI (est)**	**CV**	**M/SI (est)**	**CV**	**M/SI (est)**	**CV**	**M/SI (est)**	**CV**	**M/SI (est)**	**CV**	**M/SI (est)**	**CV**
1990	na	na	na	na	na	na	0	0	0	0	na	na		
1991	na	na	na	na	na	na	0	0	49	0.46	na	na	0	0
1992	na	na	na	na	na	na	110	0.97	154	0.35	na	na	110	0.97
1993	na	na	na	na	na	na	0	0	205	0.31	na	na	0	0
1994	na	na	0	0	na	na	182	0.71	240	0.51	na	na	182	0.71
1995	na	na	0	0	na	na	0	0	80	1.16	na	na	0	0
1996	na	na	0	0	na	na	0	0	114	0.61	na	na		
1997	161	1.58	45	0.82	na	na	0	0	140	0.61	na	na		
1998	0	0	0	0	na	na	0	0	34	0.92	na	na		
1999	0	0	0	0	0	0	0	0	69	0.7	0	0		
2000	27	0.17	0	0	0	0	137	0.34	26	1	0	0		
2001	27	0.19	0	0	unk	na	161	0.34	26	1	unk	na		
2002	25	0.17	0	0	unk	na	70	0.32	30	0.74	unk	na		
2003	31	0.25	0	0	0	0	216	0.27	31	0.93	22	0.97		
2004	26	0.2	0	0	22	0.99	200	0.3	7	0.98	0	0		
2005	38	0.29	0	0	58	1.02	213	0.28	59	0.49	9.4	1.03		
2006	3	0.53	0	0	29	0.74	40	0.5	41	0.71	0	0		
2007	2	1.03	0	0	12	0.98	29	0.66	0	0	0	0		
2008	0	0	0	0	15	0.73	13	0.57	81	0.57	0	0		
2009	0	0	0	0	4	0.92	171	0.28	0	0	0	0		
2010	0	0	0	0	0	0	37	0.32	66	0.9	0	0		
2011	0	0	0	0	0	0	141	0.24	18	0.43	0	0		
2012	0	0	0	0	0	0	27	0.47	9	0.92	0	0		
2013	0	0	0	0	0	0	33	0.31	4	1.03	0	0		
2014	9.7	0.94	0	0	0	0	16	0.50	10	0.66	0	0		
2015	0	0	0	0	0	0	15	0.52	0	0	0	0		
2016	0	0	0	0	0	0	28	0.46	0	0	0	0		
2017	0	0	0	0	0	0	15	0.64	0	0	0	0		
2018	0	0	0	0	0	0	0	0	0	0	0	0	0	0
2019	0	0	0	0	0	0	79	0.28	0	0	0	0	0	0
2020	0	0	0	0	0	0	31	0.26	2	na	0	0		
2021	0	0	0	0	0	0	15	0.52	15	0.52	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
2020	18.4	0.51	0	0	0	0	2	1.01	12.2	0.7
2021	0	0	0	0	3.8	.88	3	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

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
2020	0	0	0	0	1.8	0.88	0	0	0	0	5.7	0.44
2021	0	0	0	0	0	0	0	0	0	0	2.8	0.66

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
2020	371	0.45
2021	355	0.31

Note: This table only includes observed bycatch.
na=not applicable; unk= observer coverage was absent or too low to detect bycatch, or no estimate generated; tbd= to be determined

Common Dolphin, Western North Atlantic Stock

	Mid-Atlantic Bottom Trawl		Mid-Atlantic Gillnet		North Atlantic Bottom Trawl		NE Sink Gillnet		Northeast Midwater Trawl		Pelagic Drift Gillnet		Pelagic Longline	
Year	**M/SI (est)**	**CV**	**M/SI (est)**	**CV**	**M/SI (est)**	**CV**	**M/SI (est)**	**CV**	**M/SI (est)**	**CV**	**M/SI (est)**	**CV**	**M/SI (est)**	**CV**
1990	na	na	na	na	0	0	0	0	na	na			na	na
1991	na	na	na	na	0	0	0	0	na	na	223	0.12	na	na
1992	na	na	na	na	0	0	0	0	na	na	227	0.09	0	0
1993	na	na	na	na	0	0	0	0	na	na	238	0.08	0	0
1994	na	na	0	0	0	0	0	0	na	na	163	0.02	0	0
1995	na	na	7.4	0.69	142	0.77	0	0	na	na	83	0	0	0
1996	na	na	43	0.79	0	0	63	1.39	na	na			0	0
1997	0	0	0	0	93	1.06	0	0	na	na			0	0
1998	0	0	0	0	0	0	0	0	na	na			0	0
1999	0	0	0	0	0	0	146	0.97	0	0			0	0
2000	0	0	0	0	27	0.29	0	0	0	0			0	0
2001	103	0.27	0	0	30	0.3	0	0	0	0			0	0
2002	87	0.27	0	0	26	0.29	0	0	0	0			0	0
2003	99	0.28	0	0	26	0.29	0	0	0	0			0	0
2004	159	0.3	0	0	26	0.29	0	0	0	0			0	0
2005	141	0.29	0	0	32	0.28	5	0.8	0	0			0	0
2006	131	0.28	0	0	25	0.28	20	1.05	0	0			0	0
2007	66	0.27	0	0	24	0.28	11	0.94	0	0			0	0
2008	23	1	0	0	6	0.99	34	0.77	0	0			0	0
2009	167	0.46	0	0	24	0.6	43	0.77	0	0			8.8	1
2010	21	0.96	30	0.48	114	0.32	42	0.81	1[a]	na			0	0
2011	271	0.25	29	0.53	72	0.37	64	0.71	0	0			0	0
2012	323	0.26	15	0.93	40	0.54	95	0.4	1[a]	0			61.8	.68
2013	269	0.29	62	0.67	17	0.54	104	0.46	0	0			0	0
2014	17	0.53	17	0.86	17	0.53	111	0.47	0	0			0	0
2015	250	0.32	30	0.55	22	0.45	55	0.54	0	0			9.1	1.0
2016	177	0.33	7	0.97	16	0.46	80	0.38	0	0			0	0
2017	380	0.23	22	0.71	0	0	133	0.28	0	0			4.92	1
2018	205	0.54	98	0.91	28	0.54	93	0.45	0	0			1.44	1
2019	395	0.23	20	0.56	10	0.62	5	0.68	0	0			0	0
2020	333	0.14	30	0.55	13	0.43	50	0.25	0	0			0	0
2021	230	0.57	24	0.33	43	0.42	39	0.24	0	0	0	0	0	0

Note: This table only includes observed bycatch.
[a] Unextrapolated mortalities
na=not applicable; unk= observer coverage was absent or too low to detect bycatch, or no estimate generated; tbd= to be determined

Harbor Seal

	Herring Purse Seine		Mid-Atlantic Bottom Trawl		Mid-Atlantic Gillnet		Mid-Atlantic Midwater Trawl		Northeast Bottom Trawl		NE Sink Gillnet		Northeast Midwater Trawl	
Year	**M/SI (est)**	**CV**	**M/SI (est)**	**CV**	**M/SI (est)**	**CV**	**M/SI (est)**	**CV**	**M/SI (est)**	**CV**	**M/SI (est)**	**CV**	**M/SI (est)**	**CV**
1990	na	na	na	na	na	na	na	na	0	0	602	0.68	na	na
1991	na	na	na	na	na	na	na	na	0	0	231	0.22	na	na
1992	na	na	na	na	na	na	na	na	0	0	373	0.23	na	na
1993	na	na	na	na	na	na	na	na	0	0	698	0.19	na	na
1994	na	na	na	na	na	na	na	na	0	0	1330	0.25	na	na
1995	na	na	na	na	0	0	na	na	0	0	1179	0.21	na	na
1996	na	na	na	na	0	0	na	na	0	0	911	0.27	na	na
1997	na	na	0	0	0	0	na	na	0	0	598	0.26	na	na
1998	na	na	0	0	11	0.77	na	na	0	0	332	0.33	na	na
1999	na	na	0	0	0	0	na	na	0	0	1446	0.34	0	0
2000	na	na	0	0	0	0	0	0	0	0	917	0.43	0	0
2001	na	na	0	0	0	0	0	0	0	0	1471	0.38	0	0
2002	na	na	0	0	0	0	0	0	0	0	787	0.32	0	0
2003	0	0	0	0	0	0	0	0	0	0	542	0.28	0	0
2004	0	0	0	0	15	0.86	0	0	0	0	792	0.34	0	0
2005	0	0	0	0	63	0.67	0	0	0	0	719	0.2	0	0
2006	na	na	0	0	26	0.98	0	0	0	0	87	0.58	0	0
2007	0	0	0	0	0	0	0	0	0	0	92	0.49	0	0
2008	0	0	0	0	88	0.74	0	0	0	0	242	0.41	0	0
2009	0	0	24	0.92	47	0.68	0	0	0	0	513	0.28	1.3	0.81
2010	0	0	11	1.1	89	0.39	1[a]	0	0	0	540	0.25	2	0
2011	1[a]	0	0	0	21	0.67	0	0	9	0.58	343	0.19	0	0
2012	0	0	23	1	0	0	0	0	3	1	252	0.26	1	0
2013	0	0	11	0.96	0	0	0	0	4	0.89	147	0.3	0	0
2014	0	0	10	0.95	19	1.06	0	0	11	0.63	390	0.39	na	ma
2015	0	0	7.4	1.0	48	0.52	0	0	0	0	474	0.17	2[a]	na
2016	0	0	0	0	18	0.95	0	0	0	0	245	0.29	1[a]	na
2017	0	0	0	0	3	0.62	0	0	0	0	298	0.18	0	0
2018	0	0	6	0.94	26	0.52	0	0	0	0	188	0.36	0	0
2019	0	0	7	0.93	17	0.35	0	0	5.4	0.88	316	0.15	0	0
2020	0	0	4.3	0.67	9	0.43	0	0	4.6	0.68	261	0.14	0	0
2021	0	0	0	0	9	0.40	0	0	0	0	241	0.13	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	22	1.09	0	0	16	0.24	930	0.16	0	0
2018	0	0	56	0.58	15	1.04	0	0	32	0.42	1113	0.32	1[a]	na
2019	0	0	22	0.53	7	0.93	0	0	30	0.37	2014	0.17	0	0
2020	0	0	34.7	.35	9.3	0.72	0	0	25.8	0.26	1357	0.14	0	0
2021	0	0	0	0	7	0.69	0	0	7.5	.60	1027	0.14	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	29	0.84	5.4	0.89	163	0.19
2020	2	1.01	1.8	0.89	72	0.22
2021	2	1.01	0	0	66	0.24

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
Rice'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.