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

<table>
<thead>
<tr>
<th>Stock Name</th>
<th>Stock Name</th>
</tr>
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<tbody>
<tr>
<td>NORTH ATLANTIC RIGHT WHALE (Eubalaena glacialis): Western Atlantic Stock</td>
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<tr>
<td>HUMPBACK WHALE (Megaptera novaeangliae): Gulf of Maine Stock</td>
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</tr>
<tr>
<td>FIN WHALE (Balaenoptera physalus): Western North Atlantic Stock</td>
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<tr>
<td>SEI WHALE (Balaenoptera borealis borealis): Nova Scotia Stock</td>
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</tr>
<tr>
<td>COMMON MINKE WHALE (Balaenoptera acuto-rostrata acuto-rostrata): Canadian East Coast Stock</td>
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</tr>
<tr>
<td>COMMON DOLPHIN (Delphinus delphis delphis): Western North Atlantic Stock</td>
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</tr>
<tr>
<td>COMMON BOTTLENOSE DOLPHIN (Tursiops truncatus truncatus): Western North Atlantic Northern Migratory Coastal Stock</td>
<td></td>
</tr>
<tr>
<td>COMMON BOTTLENOSE DOLPHIN (Tursiops truncatus truncatus): Western North Atlantic Southern Migratory Coastal Stock</td>
<td></td>
</tr>
<tr>
<td>COMMON BOTTLENOSE DOLPHIN (Tursiops truncatus truncatus): Northern North Carolina Estuarine System Stock</td>
<td></td>
</tr>
<tr>
<td>COMMON BOTTLENOSE DOLPHIN (Tursiops truncatus truncatus): Southern North Carolina Estuarine System Stock</td>
<td></td>
</tr>
<tr>
<td>HARBOR PORPOISE (Phocoena phocoena phocoena): Gulf of Maine/Bay of Fundy Stock</td>
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<tr>
<td>HARBOR SEAL (Phoca vitulina vitulina): Western North Atlantic Stock</td>
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<tr>
<td>GRAY SEAL (Halichoerus grypus atlantica): Western North Atlantic Stock</td>
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<td>SPERM WHALE (Physeter macrocephalus): Northern Gulf of Mexico Stock</td>
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<td>BRYDE'S WHALE (Balaenoptera edeni): Northern Gulf of Mexico Stock</td>
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<td>CUVIER'S BEAKED WHALE (Ziphius cavirostris): Northern Gulf of Mexico Stock</td>
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<td>BLAINVILLE'S BEAKED WHALE (Mesoplodon densirostris): Northern Gulf of Mexico Stock</td>
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<td>GERVAIS' BEAKED WHALE (Mesoplodon europaeus): Northern Gulf of Mexico Stock</td>
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<tr>
<td>COMMON BOTTLENOSE DOLPHIN (Tursiops truncatus truncatus): Northern Gulf of Mexico Oceanic Stock</td>
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<tr>
<td>PANTROPICAL SPOTTED DOLPHIN (Stenella attenuata attenuata): Northern Gulf of Mexico Stock</td>
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<td>STRIPED DOLPHIN (Stenella coeruleoalba): Northern Gulf of Mexico Stock</td>
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SPINNER DOLPHIN (*Stenella longirostris longirostris*): Northern Gulf of Mexico Stock 365

ROUGH-TOOTHED DOLPHIN (*Steno bredanensis*): Northern Gulf of Mexico Stock 376

CLYMENE DOLPHIN (*Stenella clymene*): Northern Gulf of Mexico Stock 388

FRASER’S DOLPHIN (*Lagenodelphis hosei*): Northern Gulf of Mexico Stock 399

KILLER WHALE (*Orcinus orca*): Northern Gulf of Mexico Stock 408

FALSE KILLER WHALE (*Pseudorca crassidens*): Northern Gulf of Mexico Stock 419

PYGMY KILLER WHALE (*Feresa attenuata*): Northern Gulf of Mexico Stock 429

DWARF SPERM WHALE (*Kogia sima*): Northern Gulf of Mexico Stock 439

PYGMY SPERM WHALE (*Kogia breviceps*): Northern Gulf of Mexico Stock 451

MELON-HEADED WHALE (*Peponocephala electra*): Northern Gulf of Mexico Stock 464

RISSO’S DOLPHIN (*Grampus griseus*): Northern Gulf of Mexico Stock 475

SHORT-FINNED PILOT WHALE (*Globicephala macrorhynchus*): Northern Gulf of Mexico Stock 487
TABLE 1. A SUMMARY (including footnotes) OF ATLANTIC MARINE MAMMAL STOCK ASSESSMENT REPORTS FOR STOCKS OF MARINE MAMMALS UNDER NMFS AUTHORITY THAT OCCUPY WATERS UNDER USA JURISDICTION.

Total Annual S.I. (serious injury) and Mortality and Annual Fisheries S.I. and Mortality are mean annual figures for the period 2013-2014 – 2017-2018. The “SAR revised” column indicates 2019 stock assessment reports that have been revised relative to the 2018 reports (Y=yes, N=no). If abundance, mortality, PBR or status have been revised, they are indicated with the letters “a”, “m”, “p” and “status” respectively. For those species not updated in this edition, the year of last revision is indicated. Unk = unknown and undet=undetermined (PBR for species with outdated abundance estimates is considered "undetermined").

<table>
<thead>
<tr>
<th>ID</th>
<th>Species</th>
<th>Stock Area</th>
<th>Update this Year</th>
<th>Nbest</th>
<th>Nbest CV</th>
<th>Nmin</th>
<th>Rmax</th>
<th>Fr</th>
<th>PBR</th>
<th>Total Annual S.I. and Mort. (M/S I)</th>
<th>Annual Fish. S.I. and Mort. (M/S I) (cv)</th>
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<th>SAR Revised of Last Update</th>
<th>Last Survey Year</th>
<th>Comment(s)</th>
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<td>Rmax</td>
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<td>0.24</td>
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<td>8.6±0.5</td>
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Total M/SI presented here is model-derived. As this has not been broken down by cause, the fishery M/SI reported here is observed interaction only.
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<td>0.1</td>
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<td>5,689</td>
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<td>0.4</td>
<td>46</td>
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<td>5,689</td>
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<td>46</td>
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<td>12</td>
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<td>Last Survey Year</td>
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<td>Cuvier's beaked whale</td>
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<td>4,282*</td>
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<td>8,085*</td>
<td>0.04</td>
<td>0.5</td>
<td>81</td>
<td>0.2</td>
<td>0</td>
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<td>8,085*</td>
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<td>0.5</td>
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<td>8,085*</td>
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| 54 | Hooded seal
Western North Atlantic | N | unk | unk | unk | 0.12 | 0.75 | unk | 1,680 | 0.6 (1.12) | N (2018) | n/a | NEC |
| 55 | Sperm whale
Gulf of Mexico | Y | 763 1.180 | 0.130.22 | 569983 | 0.04 | 0.1 | 1.4 2.0 | 0.2 0.2 (1.0) | Y | 2017, 2018 |
| 56 | Bryde’s whale
Gulf of Mexico | Y | 33 51 | 1.02 0.5 | 1634 | 0.04 | 0.1 | 0.03 0.1 | 0.8 0 | 0 Y | 2017, 2018 |
| 57 | Cuvier’s beaked whale
Gulf of Mexico | Y | 24 18 | 1.040.75 | 2610 | 0.04 | 0.5 | 0.4 0.1 | 0.5 2 | 0 N | 2017, 2018 |

Total M/SI is a minimum estimate and does not include Fisheries M/SI.

M/SI is a combined estimate for Blainville’s, Gervais’, and Cuvier’s beaked whales.
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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.
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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.
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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.
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<td></td>
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<tr>
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<td>129313</td>
<td>1.00</td>
<td>0.03</td>
<td>0.04</td>
<td>0.5</td>
<td>0.4 0.75</td>
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<td>0.5</td>
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<td>0.5</td>
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<td>N</td>
<td>N</td>
<td>N (2012)</td>
<td>SEC</td>
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<tr>
<td>105</td>
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<td>152613</td>
<td>1.02</td>
<td>1.15</td>
<td>0.04</td>
<td>0.5</td>
<td>0.4 2.8</td>
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<tr>
<td>106</td>
<td>Dwarf sperm whale</td>
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<td>186336</td>
<td>1.04</td>
<td>0.35</td>
<td>0.04</td>
<td>0.5</td>
<td>0.4 1.5</td>
<td>0.31</td>
<td>0</td>
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<td>N (2012)</td>
<td>SEC</td>
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<tr>
<td>107</td>
<td>Pygmy sperm whale</td>
<td>Gulf of Mexico</td>
<td>Y</td>
<td>186336</td>
<td>1.00</td>
<td>0.35</td>
<td>0.04</td>
<td>0.5</td>
<td>0.4 1.5</td>
<td>0.3 0.31 (1.0)</td>
<td>N</td>
<td>N</td>
<td>N (2012)</td>
<td>SEC</td>
<td></td>
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</tr>
<tr>
<td>108</td>
<td>Melon-headed whale</td>
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<td>2,335</td>
<td>1.74</td>
<td>0.68</td>
<td>0.04</td>
<td>0.5</td>
<td>13 0.3</td>
<td>0.3</td>
<td>0</td>
<td>N</td>
<td>N (2012)</td>
<td>SEC</td>
<td></td>
<td></td>
</tr>
<tr>
<td>109</td>
<td>Risso’s dolphin</td>
<td>Gulf of Mexico</td>
<td>Y</td>
<td>2,442</td>
<td>1.97</td>
<td>0.46</td>
<td>0.04</td>
<td>0.5</td>
<td>16 2.5</td>
<td>2.9 (0.85)</td>
<td>N</td>
<td>N (2015)</td>
<td>2017, 2018</td>
<td>SEC</td>
<td></td>
<td></td>
</tr>
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*Estimate for Kogia spp. only*
<table>
<thead>
<tr>
<th>ID</th>
<th>Species</th>
<th>Stock Area</th>
<th>Update this Year</th>
<th>Nbest</th>
<th>Nbest CV</th>
<th>Nmin</th>
<th>Rmax</th>
<th>Fr</th>
<th>PBR</th>
<th>Total Annual S.I. and Mort. M/S I</th>
<th>Annual Fish. S.I. and Mort. M/S I (cv)</th>
<th>Strategic Status</th>
<th>SAR Revised of Last Update</th>
<th>Last Survey Year</th>
<th>Comment</th>
<th>NMFS Ctr.</th>
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<tbody>
<tr>
<td>110</td>
<td>Pilot whale, short-finned</td>
<td>Gulf of Mexico</td>
<td>Y</td>
<td>2,415+1.32</td>
<td>0.66+0.43</td>
<td>1456034</td>
<td>0.04</td>
<td>0.5</td>
<td>15.75</td>
<td>0.53.9</td>
<td>0.50.4</td>
<td>(1.0)</td>
<td>N</td>
<td>2015</td>
<td>2017, 2018</td>
<td>Nmf 2015a</td>
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</table>

Nbest includes all Globicephala sp., though it is presumed that only short-finned pilot whales are present in the Gulf of Mexico.
<table>
<thead>
<tr>
<th>ID</th>
<th>Species</th>
<th>Stock Area</th>
<th>Update this Year</th>
<th>Nbest</th>
<th>Nbest CV</th>
<th>Nmin</th>
<th>Rmax</th>
<th>Fr</th>
<th>PBR</th>
<th>Total Annual Fish. S.I. and Mort. M/S</th>
<th>Annual Fish. S.I. and Mort. M/S (cv)</th>
<th>Strategic Status</th>
<th>SAR Revised of Last Update</th>
<th>Last Survey Year</th>
<th>Comment</th>
<th>NMFS Ctr.</th>
</tr>
</thead>
<tbody>
<tr>
<td>114</td>
<td>Pilot whale, short-finned</td>
<td>Puerto Rico and U.S. Virgin Islands</td>
<td>N</td>
<td>unk</td>
<td>unk</td>
<td>unk</td>
<td>0.04</td>
<td>0.5</td>
<td>unk</td>
<td>unk</td>
<td>unk</td>
<td>Y</td>
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<td>n/a</td>
<td>SEC</td>
<td>SEC</td>
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<tr>
<td>115</td>
<td>Spinner dolphin</td>
<td>Puerto Rico and U.S. Virgin Islands</td>
<td>N</td>
<td>unk</td>
<td>unk</td>
<td>unk</td>
<td>0.04</td>
<td>0.5</td>
<td>unk</td>
<td>unk</td>
<td>unk</td>
<td>Y</td>
<td>20114</td>
<td>n/a</td>
<td>SEC</td>
<td>SEC</td>
</tr>
<tr>
<td>116</td>
<td>Atlantic spotted dolphin</td>
<td>Puerto Rico and U.S. Virgin Islands</td>
<td>N</td>
<td>unk</td>
<td>unk</td>
<td>unk</td>
<td>0.04</td>
<td>0.5</td>
<td>unk</td>
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<td>unk</td>
<td>Y</td>
<td>20114</td>
<td>n/a</td>
<td>SEC</td>
<td>SEC</td>
</tr>
</tbody>
</table>

a. The R given for right whales is the default Rmax of 0.04. The total estimated human-caused mortality and serious injury to right whales is estimated at 6.85 per year. This is derived from two components: 1) non-observed fishery entanglement records at 5.55 per year, and 2) ship strike records at 1.3 per year.

b. The total estimated human-caused mortality and serious injury to the Gulf of Maine humpback whale stock is estimated at 12.15 per year. This average is derived from two components: 1) incidental fishery interaction records 7.75; 2) records of vessel collisions 4.4.

c. The total estimated human-caused mortality and serious injury to the Western North Atlantic fin whale stock is estimated at 2.35 per year. This average is derived from two components: 1) incidental fishery interaction records 1.55; 2) records of vessel collisions 0.8.

d. The total estimated human-caused mortality and serious injury to the Nova Scotia sei whale stock is estimated at 1.0 per year. This average is derived from two components: 1) incidental fishery interaction records 0.2; 2) records of vessel collisions 0.8.

e. The total estimated human-caused mortality and serious injury to the Canadian East Coast minke whale stock is estimated at 8.0 per year. This average is derived from four components: 1) 6.6 minke whales per year (unknown CV) from U.S. and Canadian fisheries using strandings and entanglement data; 2) 1.0 per year from vessel strikes; and 3) 0.2 from U.S. observed fisheries, and 4) 0.2 non-fishery entanglement takes.

f. Estimates may include sightings of the coastal form.

g. This estimate includes Gervais’s beaked whales and Blainville’s beaked whales for the Gulf of Mexico stocks, and all undifferentiated beaked whales in the Atlantic.

h. This estimate includes both the dwarf and pygmy sperm whales.

i. This estimate includes all Globicephala sp., though it is presumed that only short-finned pilot whales are present in the Gulf of Mexico.

j. Details for these 24 stocks are included in the collective report. Common bottlenose dolphin (Tursiops truncatus truncatus), Northern Gulf of Mexico Bay, Sound, and Estuary Stocks. However, each stock has been given its own row in this table.

k. The total annual human-caused mortality and serious injury for these stocks of common bottlenose dolphins is unknown because these stocks may interact with unobserved fisheries. Also, for Gulf of Mexico BSE stocks, mortality estimates for the shrimp trawl fishery are calculated at the state level and have not been included within mortality estimates for individual BSE stocks. Therefore, minimum counts of human-caused mortality and serious injury for these stocks are presented.

l. This minimum count does not include projected mortality estimates for 2012–2016 due to the DWH oil spill.

m. This minimum count includes projected mortality estimates for 2012–2016 due to the DWH oil spill.
NORTH ATLANTIC RIGHT WHALE (*Eubalaena glacialis*): Western Atlantic Stock

**STOCK DEFINITION AND GEOGRAPHIC RANGE**

The western North Atlantic right whale population ranges primarily from calving grounds in coastal waters of the southeastern U.S. to feeding grounds in New England waters and the Canadian Bay of Fundy, Scotian Shelf, and Gulf of St. Lawrence (Figure 1). Mellinger et al. (2011) reported acoustic detections of right whales near the nineteenth-century whaling grounds east of southern Greenland, but the number of whales and their origin is unknown. However, Knowlton et al. (1992) reported several long-distance movements as far north as Newfoundland, the Labrador Basin, and southeast of Greenland. In addition, resightings of photographically identified individuals have been made off Iceland, in the old Cape Farewell whaling ground east of Greenland (Hamilton et al. 2007), in northern Norway (Jacobsen et al. 2004), and in the Azores (Silva et al. 2012), and off Brittany in northwestern France (New England Aquarium unpub. Catalog record). The September 1999 Norwegian sighting represents one of only two published sightings in the 20th century of a right whale in Norwegian waters, and the first since 1926. Together, these long-range matches indicate an extended range for at least some individuals and perhaps the existence of important habitat areas not presently well described. A few published records from the Gulf of Mexico (Moore and Clark 1963; Schmidly et al. 1972; Ward-Geiger et al. 2011) likely represent occasional wanderings of individuals beyond the sole known primary calving and wintering ground in the waters of the southeastern U.S. (East Coast). The location of much of the population is unknown during the winter.

Davis et al. (2017) recently pooled together detections from a large number of passive acoustic devices and documented broad-scale use of much more of the U.S. eastern seaboard than previously believed during much of the year. Further, there has been an apparent shift in habitat use patterns (Davis et al. 2017). Surveys flown in an area from 31 to 160 km from the shoreline off northeastern Florida and southeastern Georgia since 1996 report the majority of right whale sightings occur within 90 km of the shoreline. One sighting occurred ~140 km offshore (NMFS unpub. data) and an offshore survey in March 2010 observed the birth of a right whale in waters 75 km off Jacksonville, Florida (Foley et al. 2011). Although habitat models predict that right whales are not likely to occur farther than 90 km from the shoreline (Gowan and Ortega-Ortiz 2015), the frequency with which right whales occur in offshore waters in the southeastern U.S. remains unclear.

Visual and acoustic surveys have demonstrated the existence of seven areas where western North Atlantic right
whales aggregate seasonally: the coastal waters of the southeastern U.S.; the Great South Channel; Jordan Basin; Georges Basin along the northeastern edge of Georges Bank; Cape Cod and Massachusetts Bay; the Bay of Fundy; and the Roseway Basin on the Scotian Shelf (Brown et al. 2001; Cole et al. 2013). Since 2013, increased detections and survey effort in the Gulf of St. Lawrence indicate right whale presence in late spring through early fall (Cole et al. 2016, Khan et al. 2016, 2018). Passive acoustic studies of right whales have demonstrated their year-round presence in the Gulf of Maine (Morano et al. 2012; Bort et al. 2015), New Jersey (Whitt et al. 2013), and Virginia (Salisbury et al. 2016). Additionally, right whales were acoustically detected off Georgia and North Carolina in 7 of 11 months monitored (Hodge et al. 2015). All of this work further demonstrates the highly mobile nature of right whales.

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

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

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

New England waters are important feeding habitats for right whales, where they feed primarily on copepods (largely of the genera Calanus and Pseudocalanus). Right whales must locate and exploit extremely dense patches of zooplankton to feed efficiently (Mayo and Marx 1990). These dense zooplankton patches are likely a primary characteristic of the spring, summer, and fall right whale habitats (Kenney et al. 1986, 1995). While feeding in the coastal waters off Massachusetts has been better studied than in other areas, right whale feeding has also been observed on the margins of Georges Bank, in the Great South Channel, in the Gulf of Maine, in the Bay of Fundy, and over the Scotian Shelf (Baumgartner et al. 2007). The characteristics of acceptable prey distribution in these areas are
beginning to emerge (e.g., Baumgartner et al. 2003; Baumgartner and Mate 2003). The National Marine Fisheries Service (NMFS) and Center for Coastal Studies aerial surveys during the springs of 1999–2011 found right whales along the NorthernEdge of Georges Bank, in the Great South Channel, in Georges Basin, and in various locations in the Gulf of Maine including Cashes Ledge, Platts Bank, and Wilkinson Basin. Analysis of the sightings data has shown that the utilization of these areas has a strong seasonal component (Pace and Merrick 2008). Although right whales were consistently found in these locations, studies also highlight the high interannual variability in right whale use of some habitats (Pendleton et al. 2009, Ganley et al. 2019). 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).

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

Genetic analyses based upon direct sequencing of mitochondrial DNA (mtDNA) have identified 7 mtDNA haplotypes in the western North Atlantic right whale, including heteroplasmcy 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 (Walidick et al. 2002). However, revised conclusions that nearly all the remains in the North American Basque whaling archaeological sites were bowhead whales (Balaena mysticetus) and not right whales (Rastogi et al. 2004; McLeod et al. 2008) contradict the previously held belief that Basque whaling during the 16th and 17th centuries was principally responsible for the loss of genetic diversity.

High-resolution (i.e., using 35 microsatellite loci) genetic profiling has completed for 75% of all North Atlantic right whales identified through 2006. This work has improved our understanding of genetic variability, the number of reproductively active individuals, reproductive fitness, parentage, and relatedness of individuals (Frasier et al. 2007, 2009). One emerging result finding of the genetic studies is the importance of obtaining biopsy samples from calves on the calving grounds. Between 1990 and 2010, only about 60% of all known calves were seen with their mothers in summering areas when their callosity patterns are stable enough to reliably make a photo-ID match later in life. The remaining 40% were not seen on a known summering ground. Because the calf’s genetic profile is the only reliable way to establish parentage, if the calf is not sampled when associated with its mother early on, then it is not possible to link it with a calving event or to its mother, and information such as age and familial relationships is lost. From 1980 to 2001, there were 64 calves born that were not sighted later with their mothers and thus unavailable to provide age-specific mortality information (Frasier et al. 2007).

An additional interpretation of paternity analyses is that the population size may be larger than was previously thought. Fathers for only 45% of known calves have been genetically determined, yet, genetic profiles were available for 65% of all photo-identified males (Frasier 2005). The conclusion was that the majority of these calves must have different fathers that cannot be accounted for by the unsampled males, therefore the population of males must be larger (Frasier 2005). However, a recent study compared comparing photo-identification and pedigree genetic data for
animals known or presumed to be alive during 1980–2016 and found that the presumed alive estimate is similar to the actual abundance of this population, which indicates that the majority of the animals have been photo-identified (Fitzgerald 2018).

**POPULATION SIZE**

The western North Atlantic right whale stock size is based on a published state-space model of the sighting histories of individual whales identified using photo-identification techniques (Pace et al. 2017). Sightings histories were constructed from the photo-ID recapture database as it existed in October 2018, which included photographic information up through January 2018. Using a hierarchical, state-space Bayesian open population model of these histories produced a median abundance value (N\text{best}) of 428–412 individuals (95% credible intervals 406–447, Table 1). As with any statistically-based estimation process, uncertainties exist in the estimation of abundance because it is based on a probabilistic model that makes certain assumptions about the structure of the data. Because the statistically-based uncertainty is asymmetric about N, the credible interval is used above to characterize that uncertainty (as opposed to a CV that may appear in other stock assessment reports).

**Table 1. Best and minimum abundance estimates for the western North Atlantic right whale (Eubalaena glacialis) with Maximum Productivity Rate (R\text{max}), Recovery Factor (F\text{r}) and PBR.**

<table>
<thead>
<tr>
<th>N\text{best}</th>
<th>95% Credible Interval</th>
<th>60% Credible Interval</th>
<th>N\text{min}</th>
<th>Fr</th>
<th>R\text{max}</th>
<th>PBR</th>
</tr>
</thead>
<tbody>
<tr>
<td>412</td>
<td>403–424</td>
<td>408–416</td>
<td>408</td>
<td>0.1</td>
<td>0.04</td>
<td>0.8</td>
</tr>
</tbody>
</table>

**Historical Abundance**

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

**Minimum Population Estimate**

The minimum population estimate is the lower limit of the two-tailed 60% credible interval about the median of the posterior abundance estimates using the methods of Pace et al. (2017). This is roughly equivalent to the 20th percentile of the log-normal distribution as specified by Wade and Angliss (1997). The median estimate of abundance for western North Atlantic right whales is 428–412. The minimum population estimate as of January 2017-2018 is 418 individuals (Table 1) and stands as N\text{min}. The 17 known mortalities from 2017 are not accounted for in this estimate.

**Current Population Trend**

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

Examination of the abundance estimates for the years 1990–2011 (Figure 2) suggests that abundance increased at about 2.8% per annum from posterior median point estimates of 270 individuals in 1990 to 481 in 2011, but that there was a 99.99% chance that abundance declined from 2011 to 2017–2018 when the final estimate was 428–412 individuals. The overall abundance decline between 2011 and 2018 was 14.35% (CI = 11.67% to 16.60%). As noted above, there seems to have been a considerable change in right whale habitat use patterns in areas where most of the population has been observed in previous years (e.g. Davies et al. 2017), exposing the population to additional anthropogenic threats (Hayes et al. 2018). This apparent change in habitat use has the effect that, despite relatively constant effort to find whales in traditional areas, the chance of seeing–photographically capturing an individual that is alive has decreased. However, the methods in Pace et al. (2017) account for changes in capture probability.

There were 17 right whale mortalities in 2017 (Daoust et al. 2017). This number exceeds the largest estimated mortality rate during the past 25 years. Further, despite high survey effort, only 5 and 0 calves were detected in 2017 and 2018, respectively.
Figure 2. (a) Abundance estimates for North Atlantic right whales. Estimates are the median values of a posterior distribution from modeled capture histories. Also shown are sex-specific abundance estimates. Cataloged whales may include some but not all calves produced each year. (b) Crude annual growth rates from the abundance values.
CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

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

During 1990–2017, at least 447 calves were born into the population. The number of calves born annually ranged from 1 to 39, and averaged 16 but was highly variable (SD=8.9). **No calves were born the winter of 2017–2018.** The fluctuating abundance observed from 1990 to 2017 makes interpreting a count of calves by year less clear than measuring population productivity, which we index by the number of calves detected/estimated abundance (Apparent Productivity Index or API). Productivity for this stock has been highly variable over time and has been characterized by periodic swings in per capita birth rates (Figure 3). Notwithstanding the high variability observed, **and as expected** for a small population, productivity in North Atlantic right whales lacks a definitive trend. Corkeron et al. (2018) found that during 1990–2016, calf count rate increased at 1.98% per year with outlying years of very high and low calf production. This is approximately a third of that found for three different southern right whale (*Eubalaena*...
australis) populations during the same time period (5.3–7.2%). Their projection models suggest that this rate could be 4% per year if female survival was the highest recorded over the time series from Pace et al. (2017). Reviewing the available literature, Corkeron et al. (2018) showed that female mortality is primarily anthropogenic, and concluded that anthropogenic mortality has limited the recovery of North Atlantic right whales. In a similar effort, Kenny (2018) projected a series of scenarios that varied entanglement mortality from observed to zero. Using a scenario with zero entanglement mortality, which included 15 ‘surviving’ females, and a five-year calving interval, the projected population size including 26 additional calf births would be 588 by 2016.

Figure 3. Productivity in the North Atlantic right whale population as characterized by calves detected/(estimated number of females).
North Atlantic right whales have thinner blubber than southern right whales off South Africa (Miller et al. 2011). Blubber thickness of male North Atlantic right whales (males were selected to avoid the effects of pregnancy and lactation) varied with *Calanus* abundance in the Gulf of Maine (Miller et al. 2011). Sightings of North Atlantic right whales correlated with satellite-derived sea-surface chlorophyll concentration (as a proxy for productivity), and calving rates correlated with chlorophyll concentration prior to gestation (Hlista et al. 2009). On a regional scale, observations of North Atlantic right whales correlate well with copepod concentrations (Pendleton et al. 2009).

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

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

The maximum net productivity rate is unknown for this stock. For purposes of this assessment, the maximum net productivity rate was assumed to be the default value of 0.04. This value is based on theoretical modeling showing that cetacean populations may not grow at rates much greater than 4% given the constraints of their reproductive life history (Barlow et al. 1995). 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 5-year calving interval, the projected population size including 26 additional calf births would have been 588 by 2016. Single Single-year production has exceeded 0.04 in this population several times, but those outputs are not likely sustainable given the 3-year minimum interval required between successful calving events and the small fraction of reproductively active females. This is likely related to synchronous calving that can occur in capital breeders under variable environmental conditions. Hence, uncertainty exists as to whether the default value is representative of maximum net productivity for this stock, but it is unlikely that it is much higher than the default.

POTENTIAL BIOLOGICAL REMOVAL

Potential biological removal (PBR) is the product of minimum population size, one-half the maximum net productivity rate and a recovery factor for endangered, depleted, threatened stocks, or stocks of unknown status relative to OSP (MMPA Sec. 3. 16 U.S.C. 1362; Wade and Angliss 1997). The recovery factor for right whales is 0.1 because this species is listed as endangered under the Endangered Species Act (ESA). The minimum population size is 418408. The maximum productivity rate is 0.04, the default value for cetaceans. PBR for the Western North Atlantic stock of the North Atlantic right whale is 0.8 (Table 1).

ANNUAL HUMAN-CAUSED SERIOUS INJURY AND MORTALITY

For the period 2013–2014 through 2017–2018, the minimum rate of average annual detected (i.e. observed) human-caused mortality and serious injury to right whales averaged 6.85 to 8.15 per year (Table 2). This is derived from two components: 1) incidental fishery entanglement records at 5.5 to 6.85 per year, and 2) vessel strike records at averaging 1.3 per year. Early analyses of the effectiveness of the ship strike rule were reported by Silber and Bettridge (2012). Recently, van der Hoop et al. (2015) concluded that large whale mortalities due to vessel strikes decreased inside active seasonal management areas (SMAs) and increased outside inactive SMAs. Analysis by Laist et al. (2014) incorporated an adjustment for drift around areas regulated under the ship strike rule and produced weak evidence that the rule was effective inside the SMAs. When simple logistic regression models fit using maximum likelihood-based estimation procedures are applied to previously reported vessel strikes between 2000 and 2017 (Henry et al. 2020), there is no apparent trend (Fig 4). However, the odds of an entanglement event are now increasing by 6.3% per year. Although PBR analyses in this SAR reflect data collected through 2016, there were 17 right whale mortalities in 2017 (Daoust et al. 2017). This number exceeds the largest estimated mortality rate during the past 25 years. Further, despite high survey effort, only 5 and 0 calves were detected in 2017 and 2018, respectively. Therefore, the decline in the right whale population will continue for at least an additional 2 years.
Beginning with the 2001 Stock Assessment Report, Canadian records have been incorporated into the mortality and serious injury rates to reflect the effective range of this stock. It is important to stress that serious injury determinations are made based upon the best available information; these determinations may change with the availability of new information (Henry et al. 2020 in review). For the purposes of this report, discussion is limited to those only records considered to be confirmed human-caused mortalities or serious injuries are reported in the observed mortality and serious injury (M/SI) rows of Table 2.

Annual rates calculated from detected mortalities should be considered a low 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. A key uncertainty is the fraction of the actual human-caused mortality represented by the detected serious injuries and mortalities. Research on small other cetaceans has shown the actual number of deaths can be several times higher than that observed (Wells and Allen et al. 2015; Williams et al. 2011). The hierarchical Bayesian, state-space model used to estimate North Atlantic right whale abundance (Pace et al. 2017) can also be used to estimate total mortality. The estimated annual rate of total mortality using this modeling approach is 18.6 animals for the period 2013–2017 (Pace et al. submitted). This estimated total mortality accounts for detected mortality and serious injury, as well as undetected (cryptic) mortality within the population. Figure 5 compares the observed mortality and serious injury totals for the years 2000–2017 to the estimates of total mortality from the state-space model. The detection rate of mortality and serious injury for the 5-year period 2013–2017 was 51% of the model’s annual mortality estimates (Pace et al. submitted). The estimated mortality for 2018 is not yet available because it is derived from a comparison with the population estimate for 2019, which, in turn, is contingent on the processing of all photographs collected through 2019 for incorporation into the state-space model of the sighting histories of individual whales. At this time, we are unable to apportion estimated mortality by cause (fishery interaction vs. vessel strike) or by nationality (occurring in U.S. vs. Canadian waters). However, an analysis of right whale mortalities between 2003 and 2018 found that of the examined non-calf carcasses for which cause of death could be determined, all mortality was human-caused (Sharpe et al. 2019). Based on these findings, 100% of the estimated mortality of 18.6 animals per year is assumed to be human-caused. This estimate of total annual human-caused mortality may be somewhat positively biased (i.e., a slight overestimate) given that some calf mortality is likely not human-caused.
Table 2: Average annual observed and estimated human-caused mortality and serious injury for the North Atlantic right whale (Eubalaena glacialis). Observed values are from confirmed interactions. Estimated total mortality is model-derived (Pace et al. 2017; submitted). Injuries prevented are a result of successful disentanglement efforts.

<table>
<thead>
<tr>
<th>Years</th>
<th>Source</th>
<th>Average Annual Avg.</th>
</tr>
</thead>
<tbody>
<tr>
<td>2014−2018</td>
<td>Observed incidental fishery interactions</td>
<td>6.85</td>
</tr>
<tr>
<td>2014−2018</td>
<td>Observed vessel collisions</td>
<td>1.30</td>
</tr>
<tr>
<td>2014−2018</td>
<td>Observed total human-caused M/SI</td>
<td>8.15</td>
</tr>
<tr>
<td>2013-2017</td>
<td>Estimated total mortality</td>
<td>18.6</td>
</tr>
<tr>
<td>2014−2018</td>
<td>SI prevented</td>
<td>1.2</td>
</tr>
</tbody>
</table>

For North Atlantic right whales, estimates of the total mortality exceed or equal the number of detected serious injuries and mortalities (Figure 5) and currently 72% of mortalities since 2000 are estimated to have been observed. Because annual population estimates are now available (Pace et al. 2017), it is possible to estimate total annual mortality (and the number of undetected mortalities) by applying the basic population dynamic formula (Williams et al. 2002):

\[
N_{t+1} = N_t + B_t - D_t
\]

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

Solving for \(D_t\) yields:

\[
D_t = N_t + B_t - N_{t+1}
\]

which can then be used to estimate undetected mortality as: \(D_t - \text{observed deaths} = \text{undetected deaths}\).

The total mortality estimated described above is based on the assumption that all animals that exit from the population in the model are actual deaths and that all entries into the population are births. If immigration were occurring, new mature animals would be documented and captured in the estimate of \(B_t\). There is a lack of any evidence for permanent emigration from the population. Temporary emigration (e.g. the animal is not observed in the survey area for multiple years) only adds to individual capture heterogeneity, which is accommodated by the model given the longevity of the data sets. Importantly, these assumptions are not novel to the total mortality estimate, but a core part of the published Pace et al. (2017) population estimate. A method to assign cause to these undetected mortalities is currently under development; as such these additional mortalities are not counted towards PBR at this time. Another uncertainty is assigning many of the detected entanglements to country of origin. Gear recovered is often not adequately marked and whales have been known to carry gear for long periods of time and over great distances before being detected.
Figure 5. Time series of observed annual total mortality and serious injury (M/SI; black line) versus estimated total mortalities (blue points with associated error bars).

**Background**

Further, the small population size and low annual reproductive rate of right whales suggest that human sources of mortality have a greater effect relative to population growth rates than for other whales (Corkeron *et al.* 2018). The principal factor believed to be retarding growth and recovery of the population is entanglement with fishing gear.
Further, the small population size and low annual reproductive rate of right whales suggest that human sources of mortality have a greater effect relative to population growth rates than for other whales (Corkeron et al. 2018). The principal factor believed to be retarding growth and recovery of the population is entanglement with fishing gear (Kenny 2018). Between 1970 and 2018, a total of 124 right whale mortalities was recorded (Knowlton and Kraus 2001; Moore et al. 2005; Sharp et al. 2019). Of these, 18 (14.5%) were neonates that were believed to have died from perinatal complications or other natural causes. Of the remainder, 26 (21.0%) resulted from vessel strikes, 26 (21.0%) were related to entanglement in fishing gear, and 54 (43.5%) were of unknown cause. At a minimum, therefore, 42% of the observed total for the period and 43% of the 102 non-calf deaths was attributable to human impacts (calves accounted for six deaths from ship strikes and two from entanglements). One should be cautious in applying these percentages as more than minimum rates as they only represent carcasses, and exclude serious injury which is highly skewed towards entanglement. A recent analysis of human-caused serious injury and mortality during 2000–2017 (Figure 4) shows that entanglement injuries have been increasing steadily over the past twenty years while injuries from vessel strikes have shown no specific trend despite several reported cases in 2017 (Hayes et al. 2018).

Finally, it should be noted that, entanglement or minor and vessel collisions may not seriously injure or kill an animal directly, but may weaken or otherwise affect it’s reproductive success (van der Hoop et al. 2017, Corkeron et al. 2018), so that it is more likely to become vulnerable to further injury. Serious injury determinations for large whales commonly include animals carrying gear when these entanglements are constricting or appear to be determined to interfere with foraging (Henry et al. 2020 in review). 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 (Table 1). From 2013 through 2017, 28 of those examined records of mortality or serious injury (including records from both U.S. and Canadian waters, prorated to 27.75 using serious injury guidelines) involved entanglement or fishery interactions. For this time frame, the average reported mortality and serious injury to right whales due to fishery entanglement was 5.55 whales per year. Records are reviewed and those determined to be human-caused are detailed in Table 3. Information from an entanglement event often does not include...
the detail necessary to assign the entanglements to a particular fishery or location.

Although disentanglement is often unsuccessful or not possible for many cases, there are several documented cases of entanglements for which the intervention of by disentanglement teams averted a likely serious-injury determination. See Table 2 for annual average of serious injuries prevented by disentanglement. Seven serious injuries were prevented by intervention during 2013–2017 (Henry et al. 2020). Sometimes, even with disentanglement, an animal may die of injuries sustained from fishing gear. A female yearling right whale, #3107, was first sighted with gear wrapping its caudal peduncle on 6 July 2002 near Briar Island, Nova Scotia. Although the gear was removed on 1 September by the New England Aquarium disentanglement team, and the animal was alive during an aerial survey on 1 October, its carcass washed ashore at Nantucket on 12 October 2002 with deep entanglement injuries on the caudal peduncle. Additionally, but infrequently, a whale listed as seriously injured becomes gear-free without a disentanglement effort and is later in reasonable health. Such was the case for whale #1980, listed as a serious injury in 2008 but seen gear-free and apparently healthy in 2011.

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

Whales often free themselves of gear following an entanglement event, and such scarring may be a better indicator of fisheries interaction than entanglement records. A review of scars detected on identified individual right whales over a period of 30 years (1980–2009) documented 1,032 definite, unique entanglement events on the 626 individual whales identified (Knowlton et al. 2012). Most individual whales (83%) were entangled at least once, and over half of them (59%) were entangled more than once. About a quarter of the individuals identified in each year (26%) were entangled in that year. Juveniles and calves were entangled at higher rates than were adults. Scarring rates suggest that entanglements occur at about an order of magnitude more often than detected from observations of whales with gear on them. More recently, analyses of whales carrying entangling gear also suggest that entanglement wounds have become more severe since 1990, possibly due to increased use of stronger lines in fixed fishing gear (Knowlton et al. 2016).

Knowlton et al. (2012) concluded from their analysis of entanglement scarring rates over time from 1980–2009 that efforts made since 1997 of the prior decade to reduce right whale entanglement have not worked. Working from a completely different data source (observed mortalities of eight large whale species, 1970–2009), van der Hoop et al. (2012) arrived at a similar conclusion. Similarly, vessel strikes and entanglements were the two leading causes of death for known mortalities of right whales for which a cause of death could be determined. Across all 8 species of large whales, there was no detectable change in causes of anthropogenic mortality over time (van der Hoop et al. 2012). Pace et al. (2015), analyzed entanglement rates and serious injuries due to entanglement during 1999–2009, and found no support that mitigation measures implemented prior to 2009 had been effective at reducing takes due to commercial fishing. Since 2009, new entanglement mitigation measures (72 FR 193, 05 October 2007; 79 FR 124, 27 June 2014) have been implemented as part of the Atlantic Large Whale Take Reduction Plan, but their effectiveness has yet to be evaluated. One difficulty in assessing mitigation measures is the need for assessment efforts to be undertaken but rely on a statistically-significant time series to determine effectiveness.

Other Mortality

Vessel strikes are a major cause of mortality and injury to right whales (Kraus 1990; Knowlton and Kraus 2001, van der Hoop et al. 2012). Records from 2013–2014 through 2017–2018 have been summarized in Table 4. For this time frame, the average reported mortality and serious injury to right whales due to vessel strikes was 1.3 whales per year.

Early analyses of the effectiveness of the vessel-strike rule were reported by Silber and Bettridge (2012). Recently, van der Hoop et al. (2015) concluded that large whale mortalities due to vessel strikes decreased inside active seasonal management areas (SMAs) and increased outside inactive SMAs. Analysis by Laist et al. (2014) incorporated an adjustment for drift around areas regulated under the vessel-strike rule and produced weak evidence that the rule was effective inside the SMAs. When simple logistic regression models fit using maximum likelihood-based estimation procedures were applied to previously reported vessel strikes between 2000 and 2017 (Henry et al. in review), there was no apparent trend (Hayes et al. 2018).

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


<table>
<thead>
<tr>
<th>Date</th>
<th>Fate</th>
<th>ID</th>
<th>Location</th>
<th>Assigned Cause</th>
<th>Value against PBR</th>
<th>Country</th>
<th>Gear Type</th>
<th>Description</th>
</tr>
</thead>
<tbody>
<tr>
<td>07/12/2013</td>
<td>Prorated Injury</td>
<td>3123</td>
<td>off Virginia Beach, VA</td>
<td>EN</td>
<td>.75</td>
<td>XU</td>
<td>NR</td>
<td>Constricting gear cutting into mouthline; Partially disentangled; final configuration unknown. No resights post Jul/2013</td>
</tr>
<tr>
<td>01/15/2014</td>
<td>Serious Injury</td>
<td>4394</td>
<td>off Ossabaw Island, GA</td>
<td>EN</td>
<td>1</td>
<td>XU</td>
<td>NP</td>
<td>No gear present but new ent. injuries indicating prior constricting gear on both pectorals and at fluke insertion. Injury to left ventral fluke. Evidence of health decline. No resights post Feb/2014.</td>
</tr>
<tr>
<td>04/01/2014</td>
<td>Serious Injury</td>
<td>1142</td>
<td>off Atlantic City, NJ</td>
<td>EN</td>
<td>1</td>
<td>XU</td>
<td>NR</td>
<td>Constricting rostrum wrap with line trailing to at least mid-body. Resighted in 2018. Health decline evident.</td>
</tr>
<tr>
<td>04/09/2014</td>
<td>Prorated Injury</td>
<td>-</td>
<td>Cape Cod Bay, MA</td>
<td>VS</td>
<td>.52</td>
<td>US</td>
<td>-</td>
<td>Animal surfaced underneath a research vessel while it was underway (39 ft at 9 kts). Small amount of blood and some lacerations of unknown depth on lower left flank.</td>
</tr>
<tr>
<td>06/29/2014</td>
<td>Serious Injury</td>
<td>1131</td>
<td>off Cape Sable Island, NS</td>
<td>EN</td>
<td>1</td>
<td>XC</td>
<td>NR</td>
<td>At least 1, possibly 2, embedded rostrum wraps. Remaining configuration unclear but extensive. Animal in extremely poor condition: emaciated, heavy eyamid coverage, overall pale skin. No resights.</td>
</tr>
<tr>
<td>09/04/2014</td>
<td>Mortality</td>
<td>-</td>
<td>Far south of St. Pierre &amp; Miquelon, off the south coast of NL</td>
<td>EN</td>
<td>1</td>
<td>XC</td>
<td>NR</td>
<td>Carcass with constricting line around rostrum and body. No necropsy conducted, but evidence of extensive, constricting entanglement supports entanglement as COD.</td>
</tr>
<tr>
<td>Date</td>
<td>Fate</td>
<td>ID</td>
<td>Location</td>
<td>Assigned Cause</td>
<td>Value against PBR</td>
<td>Country</td>
<td>Gear Type</td>
<td>Description</td>
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<td>------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------</td>
</tr>
<tr>
<td>09/27/2014</td>
<td>Mortality</td>
<td>-</td>
<td>off Nantucket, MA</td>
<td>EN 1 US</td>
<td>NR</td>
<td></td>
<td></td>
<td>Fresh carcass with multiple lines wrapping around head, pectoral, and peduncle. Appeared to be anchored. No necropsy conducted, but extensive, constricting entanglement supports entanglement as COD.</td>
</tr>
<tr>
<td>12/18/2014</td>
<td>Serious Injury</td>
<td>3670</td>
<td>off Sapelo Sound, GA</td>
<td>EN 1 XU</td>
<td>NP</td>
<td></td>
<td></td>
<td>No gear present but new, healing entanglement injuries. Severe injuries to lip, peduncle and fluke edges. Poss. damage to right pectoral. Resights indicate health decline.</td>
</tr>
<tr>
<td>04/06/2015</td>
<td>Serious Injury</td>
<td>CT04CCB14</td>
<td>Cape Cod Bay, MA</td>
<td>EN 1 XU</td>
<td>NP</td>
<td></td>
<td></td>
<td>Encircling laceration at fluke insertion with potential to affect major artery. Source of injury likely constricting entanglement. No gear present. Evidence of health decline. No resights.</td>
</tr>
<tr>
<td>06/13/2015</td>
<td>Prorated Injury</td>
<td>-</td>
<td>off Westport, NS</td>
<td>EN .75 XC</td>
<td>NR</td>
<td></td>
<td></td>
<td>Line through mouth, trailing 300-400m ending in 2 balloon-type buoys. Full entanglement configuration unknown. No resights.</td>
</tr>
<tr>
<td>09/28/2015</td>
<td>Prorated Injury</td>
<td>-</td>
<td>off Cape Elizabeth, ME</td>
<td>EN .75 XU</td>
<td>NR</td>
<td></td>
<td></td>
<td>Unknown amount of line trailing from flukes. Attachment point(s) and configuration unknown. No resights.</td>
</tr>
<tr>
<td>11/29/2015</td>
<td>Serious Injury</td>
<td>3140</td>
<td>off Truro, MA</td>
<td>EN 1 XU</td>
<td>NR</td>
<td></td>
<td></td>
<td>New, significant ent. injuries indicating constricting wraps. No gear visible. In poor cond. with grey skin and heavy cyamid coverage. No resights.</td>
</tr>
<tr>
<td>Date</td>
<td>Fate</td>
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<td>Value against PBR</td>
<td>Country</td>
<td>Gear Type</td>
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</tr>
<tr>
<td>7/26/2016</td>
<td>Serious Injury</td>
<td>1427</td>
<td>Gulf of St Lawrence, QC</td>
<td>EN 1 XC NP</td>
<td>No gear present, but new entanglement injuries on peduncle and fluke insertions. No gear present. Resights show subsequent health decline: gray skin, rake marks, cyamids.</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>8/1/2016</td>
<td>Serious Injury</td>
<td>3323</td>
<td>Bay of Fundy, NS</td>
<td>EN 1 XC NP</td>
<td>No gear present, but new, severe entanglement injuries on peduncle, fluke insertions, and leading edges of flukes. No gear present. Significant health decline: emaciated, cyamids patches, peeling skin. No resights.</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Date</td>
<td>Fate</td>
<td>ID</td>
<td>Location</td>
<td>Assigned Cause</td>
<td>Value against PBR</td>
<td>Country</td>
<td>Gear Type</td>
<td>Description</td>
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<td>-------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------</td>
</tr>
<tr>
<td>8/13/2016</td>
<td>Serious Injury</td>
<td>4057</td>
<td>Bay of Fundy, NS</td>
<td>EN 1 CN PT</td>
<td>Free-swimming with extensive entanglement. Two heavy lines through mouth, multiple loose body wraps, multiple constricting wraps on both pectorals with lines across the chest, jumble of gear by left shoulder. Partially disentangled: left with line through mouth and loose wraps at right flipper that are expected to shed. Significant health decline: extensive cyamid coverage. Current entanglement appears to have exacerbated injuries from previous entanglement (see 16Feb2014 event). No resights.</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>8/16/2016</td>
<td>Prorated Injury</td>
<td>1152</td>
<td>off Baccaro, NS</td>
<td>EN 0.75 XC NR</td>
<td>Free-swimming with line and buoy trailing from unknown attachment point(s). No resights.</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>8/31/2016</td>
<td>Mortality</td>
<td>4320</td>
<td>Sable Island, NS</td>
<td>EN 1 CN PT</td>
<td>Decomposed carcass with multiple constricting wraps on pectoral with associated bone damage consistent with chronic entanglement.</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>9/23/2016</td>
<td>Mortality</td>
<td>3694</td>
<td>off Seguin Island, MA</td>
<td>EN 1 XC PT</td>
<td>Fresh, floating carcass with extensive, constricting entanglement. Thin blubber layer and other findings consistent with prolonged stress due to chronic entanglement. Gear previously reported as unknown.</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>12/04/2016</td>
<td>Prorated Injury</td>
<td>3405</td>
<td>off Sandy Hook, NJ</td>
<td>EN 0.75 XU NE</td>
<td>Lactating female. Free-swimming with netting crossing over blowholes and one line over back. Full configuration unknown. Calf not present, possibly already weaned. No resights. Gear type previously reported as NR.</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Date</td>
<td>Fate</td>
<td>ID</td>
<td>Location</td>
<td>Assigned Cause</td>
<td>Value against PBR</td>
<td>Country</td>
<td>Gear Type</td>
<td>Description</td>
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<td>-----------------------------------------------------------------------------</td>
</tr>
<tr>
<td>04/13/17</td>
<td>Mortality</td>
<td>4694</td>
<td>Cape Cod Bay, MA</td>
<td>VS</td>
<td>1</td>
<td>US</td>
<td>-</td>
<td>Carcass with deep hemorrhaging and muscle tearing consistent with blunt force trauma.</td>
</tr>
<tr>
<td>06/19/17</td>
<td>Mortality</td>
<td>1402</td>
<td>Gulf of St Lawrence, QC</td>
<td>VS</td>
<td>1</td>
<td>CN</td>
<td>-</td>
<td>Carcass with acute internal hemorrhaging consistent with blunt force trauma.</td>
</tr>
<tr>
<td>06/21/17</td>
<td>Mortality</td>
<td>3603</td>
<td>Gulf of St Lawrence, QC</td>
<td>EN</td>
<td>1</td>
<td>CN</td>
<td>PT</td>
<td>Fresh carcass found anchored in at least 2 sets of gear. Multiple lines through mouth and constricting wraps on left pectoral. Glucorticoid levels support acute entanglement as COD.</td>
</tr>
<tr>
<td>06/23/17</td>
<td>Mortality</td>
<td>1207</td>
<td>Gulf of St Lawrence, QC</td>
<td>VS</td>
<td>1</td>
<td>CN</td>
<td>-</td>
<td>Carcass with acute internal hemorrhaging consistent with blunt force trauma.</td>
</tr>
<tr>
<td>07/04/17</td>
<td>Serious Injury</td>
<td>3139</td>
<td>off Nantucket, MA</td>
<td>EN</td>
<td>1</td>
<td>XU</td>
<td>NP</td>
<td>No gear present, but evidence of recent extensive, constricting entanglement and health decline. No resights.</td>
</tr>
<tr>
<td>07/06/17</td>
<td>Mortality</td>
<td>-</td>
<td>Gulf of St Lawrence, QC</td>
<td>VS</td>
<td>1</td>
<td>CN</td>
<td>-</td>
<td>Carcass with fractured skull and associated hemorrhaging. Glucorticoid levels support acute blunt force trauma as COD.</td>
</tr>
<tr>
<td>07/19/17</td>
<td>Serious Injury</td>
<td>4094</td>
<td>Gulf of St Lawrence, QC</td>
<td>EN</td>
<td>1</td>
<td>CN</td>
<td>PT</td>
<td>Line exiting right mouth, crossing over back, ending at buoys aft of flukes. Non-constricting configuration, but evidence of significant health decline. No resights.</td>
</tr>
<tr>
<td>07/19/17</td>
<td>Mortality</td>
<td>2140</td>
<td>Gulf of St Lawrence, QC</td>
<td>VS</td>
<td>1</td>
<td>CN</td>
<td>-</td>
<td>Fresh carcass with acute internal hemorrhaging. Glucorticoid levels support acute blunt force trauma as COD.</td>
</tr>
<tr>
<td>08/06/17</td>
<td>Mortality</td>
<td>-</td>
<td>Martha's Vineyard, MA</td>
<td>EN</td>
<td>1</td>
<td>XU</td>
<td>NP</td>
<td>No gear present, but evidence of constricting wraps around both pectorals and flukes with associated tissue reaction. Histopathology results support entanglement as COD.</td>
</tr>
<tr>
<td>09/15/17</td>
<td>Mortality</td>
<td>4504</td>
<td>Gulf of St Lawrence, QC</td>
<td>EN</td>
<td>1</td>
<td>CN</td>
<td>PT</td>
<td>Anchored in gear with extensive constricting wraps with associated hemorrhaging.</td>
</tr>
<tr>
<td>Date</td>
<td>Fate</td>
<td>ID</td>
<td>Location</td>
<td>Assigned Cause</td>
<td>Value against PBR</td>
<td>Country</td>
<td>Gear Type</td>
<td>Description</td>
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<td>-----------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------</td>
</tr>
<tr>
<td>10/23/2017</td>
<td>Mortality</td>
<td>-</td>
<td>Nashawena Island, MA</td>
<td>EN</td>
<td>1</td>
<td>XU</td>
<td>NP</td>
<td>No gear present, but evidence of extensive ent involving pectorals, mouth, and body. Hemorrhaging associated with body and right pectoral injuries. Histo results support entanglement as COD.</td>
</tr>
<tr>
<td>1/22/2018</td>
<td>Mortality</td>
<td>3893</td>
<td>55 nm E of Virginia Beach, VA</td>
<td>EN</td>
<td>1</td>
<td>CN</td>
<td>PT</td>
<td>Extensive, severe constricting entanglement including partial amputation of right pectoral accompanied by severe proliferative bone growth. COD - chronic entanglement.</td>
</tr>
<tr>
<td>2/15/2018</td>
<td>Serious Injury</td>
<td>3296</td>
<td>33 nm E of Jekyll Island, GA</td>
<td>EN</td>
<td>1</td>
<td>XU</td>
<td>NP</td>
<td>No gear present, but extensive recent injuries consistent with constricting gear on right flipper, peduncle, and leading flake edges. Large portion of right lip missing. Extremely poor condition - emaciated with heavy cyamid load. No resights.</td>
</tr>
<tr>
<td>7/13/2018</td>
<td>Prorated Injury</td>
<td>3312</td>
<td>25.6 nm E of Miscou Island, NB</td>
<td>EN</td>
<td>0.75</td>
<td>CN</td>
<td>NR</td>
<td>Free swimming with line through mouth and trailing both sides. Full configuration unknown - unable to confirm extent of flipper involvement. No resights.</td>
</tr>
<tr>
<td>7/30/2018</td>
<td>Prorated Injury</td>
<td>3843</td>
<td>134 nm E of Grand Manan, NB</td>
<td>EN</td>
<td>0.75</td>
<td>XC</td>
<td>-</td>
<td>Free-swimming with buoy trailing 70ft behind whale. Attachment point(s) unknown. Severe, deep, raw injuries on peduncle &amp; 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.</td>
</tr>
<tr>
<td>8/25/2018</td>
<td>Mortality</td>
<td>-</td>
<td>Martha's Vineyard, MA</td>
<td>EN</td>
<td>1</td>
<td>XU</td>
<td>NP</td>
<td>No gear present. Evidence of constricting pectoral wraps with associated hemorrhaging. COD - acute entanglement</td>
</tr>
<tr>
<td>10/14/2018</td>
<td>Mortality</td>
<td>3515</td>
<td>134 nm E of Nantucket, MA</td>
<td>EN</td>
<td>1</td>
<td>XU</td>
<td>NP</td>
<td>No gear present, but evidence of constricting wraps across ventral surface and at pectorals. COD - acute, severe entanglement</td>
</tr>
<tr>
<td>12/20/2018</td>
<td>Prorated Injury</td>
<td>2310</td>
<td>Nantucket, MA</td>
<td>EN</td>
<td>0.75</td>
<td>XU</td>
<td>NR</td>
<td>Free-swimming with open bridle through mouth. Resight in Apr 2019 shows configuration changed, but unable to determine full configuration. Health appears stable. No additional resights.</td>
</tr>
<tr>
<td>Date</td>
<td>Fate</td>
<td>ID</td>
<td>Location</td>
<td>Assigned Cause</td>
<td>Value against PBR</td>
<td>Country</td>
<td>Gear Type</td>
<td>Description</td>
</tr>
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<td>--------------------------------------------------------------------------------</td>
</tr>
<tr>
<td>12/24/2018</td>
<td>Serious Injury</td>
<td>3208</td>
<td>South of Nantucket, MA</td>
<td>EN 1 XU NP</td>
<td>No gear present. Evidence of new, healed, constricting body wrap. Health decline evident - grey, lesions, thin.</td>
<td></td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

**Assigned Cause**

- Vessel strike: 0.13 (0.50/ 0.80/ 0.00/ 0.00)
- Entanglement: 5.55 (0.20/ 1.20/ 2.45/ 3.25/ 1.70/ 1.85)

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a. For more details on events please see Henry et al. 2020, in review.
b. The date sighted and location provided in the table are not necessarily when or where the serious injury or mortality occurred; rather, this information indicates when and where the whale was first reported beached, entangled, or injured.
c. Mortality events are counted as 1 against PBR. Serious injury events have been evaluated using NMFS guidelines (NOAA 2012).
d. CN=Canada, US=United States, XC=Unassigned 1st sight in CN, XU=Unassigned 1st sight in US.
e. H=hook, GN=gillnet, GU=gear unidentifiable, MF=monofilament, NP=none present, NR=none recovered/received, PT=pot/trap, WE=weir.

**HABITAT ISSUES**

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

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

**STATUS OF STOCK**

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

**REFERENCES CITED**


Humpback Whale (Megaptera novaeangliae): Gulf of Maine Stock

Stock Definition and Geographic Range

In the western North Atlantic, humpback whales feed during spring, summer, and fall over a geographic range encompassing the eastern coast of the United States, Scotian Shelf (including the Gulf of Maine), the Gulf of St. Lawrence, Newfoundland/Labrador, and western Greenland (Katona and Beard 1990). Other North Atlantic feeding grounds occur off Iceland and in the Norwegian Sea, including off northern Norway, Bear Island, Jan Mayen, and Franz Josef Land (Christensen et al. 1992; Palsbøll et al. 1997). These six regions represent relatively discrete subpopulations, fidelity to which is determined matrilineally (Clapham and Mayo 1987), which is supported by studies of the mitochondrial genome (Palsbøll et al. 1995; Palsbøll et al. 2001) and individual animal movements (Stevick et al. 2006, Kennedy et al. 2014). During the 2002 Comprehensive Assessment of North Atlantic humpback whales, the International Whaling Commission acknowledged the evidence for treating the Gulf of Maine as a separate management unit (IWC 2002).

During the summers of 1998 and 1999, the Northeast Fisheries Science Center conducted surveys for humpback whales on the Scotian Shelf to establish the occurrence and population identity of the animals found in this region, which lies between the well-studied populations of the Gulf of Maine and Newfoundland. Photographs from both surveys were compared to both the overall North Atlantic Humpback Whale Catalog and a large regional catalog from the Gulf of Maine (maintained by the College of the Atlantic and the Center for Coastal Studies, respectively); this work is summarized in Clapham et al. (2003). The match rate between the Scotian Shelf and the Gulf of Maine was 27% (14 of 52 Scotian Shelf individuals from both years). Comparable rates of exchange were obtained from the southern (28%, n=10 of 36 whales) and northern (27%, n=4 of 15 whales) ends of the Scotian Shelf (one whale was observed in both areas). In contrast, all of the 36 humpback whales identified by the same NMFS surveys elsewhere in the Gulf of Maine (including Georges Bank, southwestern Nova Scotia, and the Bay of Fundy) had been previously observed in the Gulf of Maine region. The sighting histories of the 14 Scotian Shelf whales matched to the Gulf of Maine suggested that many of them were...
During winter, whales from most North Atlantic feeding areas (including the Gulf of Maine) mate and calve in the West Indies, where spatial and genetic mixing among feeding groups occurs (Katona and Beard 1990; Clapham et al. 1993; Palsbøll et al. 1997; Stevick et al. 1998; Kennedy et al. 2013). Some whales using eastern North Atlantic feeding areas migrate to the Cape Verde Islands (Reiner et al. 1996; Wenzel et al. 2009, 2020; Stevick et al. 2016), and some individuals have been recorded in both the Cape Verde Islands and the southeast Caribbean (Stevick et al. 2016). Breeding populations in the West Indies and Cape Verdes are considered to be two Distinct Population Segments under the Endangered Species Act (81 FR 62259, September 8, 2016). In the West Indies, the majority of whales are found in the waters of the Dominican Republic, notably on Silver Bank and Navidad Bank, and in Samana Bay (Balcomb and Nichols 1982; Whitehead and Moore 1982; Mattila et al. 1989, 1994). Humpback whales also are found at much lower densities throughout the remainder of the Antillean arc (Winn et al. 1975; Levenson and Leapley 1978; Price 1985; Mattila and Clapham 1989). Passive acoustic data have documented humpback song from Silver Bank through Martinique, with singing beginning earliest on Silver Bank and later in Guadeloupe and Martinique (Heenehan et al. 2019). Although recognition of 2 breeding areas for North Atlantic humpbacks is the prevailing model, our knowledge of breeding season distribution is far from complete (see Smith and Pike 2009; Stevick et al. 2016).

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

Within the U.S. Atlantic EEZ, humpback whales have been sighted well away from the Gulf of Maine. From 2014–2019, winter surveys conducted off Virginia Beach documented over 160 humpback individuals using the Chesapeake Bay region, the majority of which were estimated to be juveniles or sub-adults based on body size (Aschettino et al. 2019). Photographic identification data indicated resights of some animals between years, and data from satellite tagging revealed utilization of this region by tagged individuals for multiple weeks (Aschettino et al. 2019). These data support previous studies that also documented Sightings of humpback whales in the vicinity of the Chesapeake and Delaware Bays occurred in 1992 (Swingle et al. 1993). Wiley et al. (1995) reported that 38 humpback whale strandings occurred during 1985–1992 in the U.S. mid-Atlantic and southeastern states. Humpback whale strandings increased, particularly along the Virginia and North Carolina coasts, and most stranded animals were sexually immature; in addition, the small size of many of these whales strongly suggested that they had only recently separated from their mothers. Wiley et al. (1995) concluded that these mid-Atlantic areas were becoming an increasingly important habitat for juvenile humpback whales. For the period 2013–2017, there are records of 95 humpback whale strandings between Maine and Florida in the Marine Mammal Health and Stranding Response database (accessed 23 October 2018). There have also been a number of wintertime humpback sightings in coastal waters of the southeastern U.S. (Zoodsma et al. 2016; Surrey-Marsden et al. 2018) Other sightings of note include 46 sightings of humpbacks in the New York-New Jersey Harbor Estuary documented between 2011 and 2016 (Brown et al. 2017). Multiple humpbacks were observed feeding off Long Island during July of 2016.
accessed 28 April 2017) and there were sightings during November–December 2016 near New York City (https://www.greateratlantic.fisheries.noaa.gov/mediacenter/2016/december/09_humans_and_humpbacks_of_new_york_2.html, accessed 28 April 2017). Additional sightings occurring during summer (about July–August) along the shelf break east of New Jersey and south of Georges Bank have been increasing since 2004 as seen New York during the NEFSC abundance surveys. Only two humpbacks were seen during surveys in 1995 and 1998 with similar levels of effort as the recent surveys after have been increasing since about 2004 (2016 survey described below; Palka 2020). Whether the increased numbers of sightings represent a distributional change, or are simply due to an increase in sighting effort and/or whale abundance, is unknown.

Stock identity of individuals observed off southeastern and mid-Atlantic states was investigated using fluke photographs of living and dead whales observed in the region (Barco et al. 2002). In this study, photographs of 40 whales (alive or dead) were of sufficient quality to be compared to catalogs from the Gulf of Maine (i.e., the closest feeding ground) and other areas in the North Atlantic. Of 21–22 live whales, 9–10 (43–45.5%) matched to the Gulf of Maine, 4–5 (19–22.7%) to Newfoundland, and 1 (4.85%) to the Gulf of St Lawrence. Of 19 dead humpbacks, 6 (31.6%) were known Gulf of Maine whales. Although the population composition of the mid-Atlantic is apparently dominated by Gulf of Maine whales, lack of photographic effort in Newfoundland makes it likely that the observed match rates under represent the true presence of Canadian whales in the region. Barco et al. (2002) suggested that the mid-Atlantic region primarily represents a supplemental winter feeding ground used by humpbacks. With populations recovering, additional surveys that include photo identification and genetic sampling should be conducted to determine which stocks are currently using the mid-Atlantic region.

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

**POPULATION SIZE**

The best abundance estimate available for the Gulf of Maine humpback whale stock is 1,396 (95% credible intervals 1,351–1,433; Table 1). This is based on a state-space model of the sighting histories of individual whales identified using photo-identification techniques (Pace et al. 2017). Sightings histories were constructed from the photo-ID recapture data through October 2016. The median abundance value was produced using a hierarchical, state-space Bayesian open population model of these histories.

**Gulf of Maine stock - Earlier estimates**

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

**Gulf of Maine Stock - Recent surveys and abundance estimates**

Humpback whales are uniquely identifiable based primarily on coloration patterns of the ventral side of the fluke and identification can be augmented by other features such as dorsal fin shape, scars and genetic data (Smith et al. 1999). A recent count of the minimum number alive (MNA) for 2015 was produced by counting the number of unique
individuals seen in 2015 in the Gulf of Maine stock area as well as seen both before and after 2015 (data provided by J. Robbins, Center for Coastal Studies, Provincetown, MA, USA). The humpback MNA for 2015 was 896 and includes not only cataloged whales but some calves born in 2015 but not yet identifiable. By comparison, an abundance of 335 (CV=0.42) humpback whales was estimated from a line transect survey conducted during June–August 2011 by ship and plane (Palka 2012). The aerial portion that contributed to the abundance estimate covered 5,313 km of tracklines over waters north of New Jersey and shallower than the 100-m depth contour through the U.S. and Canadian Gulf of Maine and up to and including the lower Bay of Fundy. The shipboard portion covered 3,107 km of tracklines in waters deeper than the 100-m depth contour out to beyond the U.S. EEZ. Both sighting platforms used a two-simultaneous-team data collection procedure, which allows estimation of abundance corrected for perception bias (Laake and Borchers 2004). Estimation of abundance was based on the independent-observer approach assuming point independence (Laake and Borchers 2004) and calculated using the mark-recapture distance sampling option in the computer program Distance (version 6.0, release 2, Thomas et al. 2009). This estimate did not include the portion of the Scotian Shelf that is known to be part of the range used by Gulf of Maine humpback whales. This estimate should not be compared to previous estimates that were derived using a different methodology. The now-outdated estimate of 823 humpbacks in the Gulf of Maine and Bay of Fundy in 2008 was based on a minimum number alive calculation. While that type of estimate is generally more accurate than one derived from line transect survey, the 2016 GAMMS guidelines (NMFS 2016) notes the decline of confidence in the reliability of abundance estimates older than eight years. For this report, two new-independent estimates presented in Table 1 are available from different methods—one based upon ship and aerial line transect surveys, a minimum number alive estimate based on unique individual counts, and a second an estimate obtained from applying mark and capture methods to photo identification records from the J. Robbins studies (Robbins and Pace 2018).

An abundance estimate of 2,368 (CV=0.48) humpback whales was generated from a shipboard and aerial survey conducted during 27 June–28 September 2016 (Figure 2 Table 1, Palka 2020) in a region covering 425,192 km². The aerial portion covered 11,782 km of tracklines that were over waters north of New Jersey from the coastline to the 100 m depth contour, throughout the U.S. waters. The shipboard portion covered 4,351 km of tracklines that were in waters offshore of central Virginia to Massachusetts (waters that were deeper than the 100 m depth contour out to beyond the outer limit of the U.S. EEZ). Both sighting platforms used a two-team data collection procedure, which allows estimation of abundance to correct for perception bias of the detected species (Laake and Borchers 2004) using standard mark-recapture distance sampling with covariates to assist in defining the detection function. The estimates were also corrected for availability bias which was estimated from dive patterns recorded from tagged humpbacks. The abundance resulting from the aerial survey in the U.S. portion of the Gulf of Maine was 1,372 (CV=0.70), where the availability bias correction factor was 1.541 (CV=0.185); thus, the uncorrected abundance was 890 (CV=0.68). The abundance resulting from the shipboard survey on the shelf break was 996 (CV=0.59), where the availability bias correction factor was 1.0.

An abundance estimates of 8,439 (CV=0.49) for the Newfoundland/Labrador portion and 1,854 (CV=0.40) for the Bay of Fundy/Scotian Shelf/Gulf of St. Lawrence portion of humpback whales in Canadian waters were was generated from the Canadian Northwest Atlantic 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).

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

Humpback whales are uniquely identifiable based primarily on coloration patterns of the ventral side of the fluke and identification can be augmented by other features such as dorsal fin shape, scars and genetic data (Smith et al. 1999). A count of the minimum number alive (MNA) for 2015 was produced by counting the number of unique individuals seen in 2015 in the Gulf of Maine stock area as well as seen both before and after 2015 (data provided by
As an alternative approach to estimating whale abundance, Robbins and Pace 2018 we analyzed the photo-identification database (Robbins and Pace 2018) and applied mark and recapture methods using a state-space model of the sighting histories of individual whales following the methodology described in Pace et al. 2017.

Sightings histories of Gulf of Maine humpback whales were constructed from the photo-ID recapture database as it existed in April 2019. The data were provided by an annual spatially arranged survey dedicated to gathering photo-id data on Gulf of Maine humpbacks. The estimation process was greatly enhanced by using photographic captures from sources other than the primary survey including additional research efforts by the principal survey team but outside of the dedicated survey effort, other cetacean research groups, and cooperating whale watch vessels. These later data were used to inform the known state matrix. All sightings from the primary survey were bounded by the hatched area noted on Figure 2 for capture-mark-recapture (CMR) sampling strata. A hierarchical, state-space Bayesian open population model of these histories produced a median abundance value of 1,396.393 individuals (95% credible intervals 1,363.1351−1,429.1433). As with any statistically-based estimation process, uncertainties exist in the estimation of abundance because it is based on a probabilistic model that makes certain assumptions about the data structure. Because the statistically-based uncertainty is asymmetric about N, the credible interval is used above to characterize that uncertainty (as opposed to a CV that may appear in other stock assessment reports).

The CMR estimate of 1,396.393 stands as the best available estimate for this stock assessment report for several reasons. First it is within the bounds of variation of the line-transect estimate from the U.S.-based survey in agreement with updated line transect survey estimate of 2,368, which has a large CV that ranges from 5,781 to 970. Second, the CMR estimate provides a tighter confidence interval and therefore is more precise. Third, the long term nature of the CMR database enabled the calculation of historical annual population estimates backwards in time through 2000, thus allowing trend analysis. Furthermore, humpback whales meet the key criteria for applying mark and recapture methodology as an animal with an established stock and home-range that is also uniquely identifiable. There is some spatial difference in sampling strata between the CMR and Line transect survey, which result in the CMR estimate better representing the population abundance. The line transect estimate is most accurate when fully sampling the defined seasonal range of the stock. The current estimate includes many sightings from the continental shelf areas east of New York and south of the Georges Bank New Jersey. While, this region is typically included for the Gulf of Maine stock particularly when assigning cases of anthropogenic human-caused mortality, further research is required to confirm that surveys south of the Gulf of Maine might be detecting animals from other stocks using U.S. waters during good forage conditions. At the same time the aerial portion of the line transect estimate did not go into the Canadian waters in the Bay of Fundy region, nor east of the Hague line. The CMR estimate can be generated from a sampling a subregion of a species range, if that region is used by the entire population, as was the case here where sampling of the GOM Gulf of Maine humpback stock was conducted throughout the Gulf of Maine. CMR estimates are potentially subject to error if there is permanent immigration/emigration into or out of the population. However there is little evidence for this, given a lack of photo ID reports for GOM Gulf of Maine animals observed permanently relocating to other stock regions which would be indicative of emigration. Similarly there is little evidence for immigration, given the lack of regular ‘new entrants’ into the population as adults. Temporary emigration (e.g. the animal is not observed in the survey area for multiple years) only adds to individual capture heterogeneity, which is accommodated by the model given the longevity of the data sets. Given the efficiency of the method, NMFS should consider investment to ensure the continuation of this data record.
Minimum Population Estimate

The minimum population estimate is the lower limit of the two-tailed 60% credible interval about the median of the posterior abundance estimates using the CMR methods of Pace et al. (2017). This is roughly equivalent to the 20th percentile of the log-normal distribution as specified by Wade and Angliss (1997). The minimum population estimate is $1,375^{1,380}$ using the CMR method.
Table 1. Summary of abundance estimates for Gulf of Maine humpback whales with month, year, and area covered during each abundance survey, and resulting abundance estimate (Nbest) and coefficient of variation (CV). The estimate considered best is in bold font.

<table>
<thead>
<tr>
<th>Month/Year</th>
<th>Type</th>
<th>Area or model</th>
<th>Nbest</th>
<th>CV</th>
</tr>
</thead>
<tbody>
<tr>
<td>Jun–Aug 2011</td>
<td>Virginia to lower Bay of Fundy</td>
<td>335</td>
<td>0.42</td>
<td></td>
</tr>
<tr>
<td>Jun–Oct 2015</td>
<td>Gulf of Maine and Bay of Fundy</td>
<td>896</td>
<td>0.00</td>
<td></td>
</tr>
<tr>
<td>Jun–Sep 2016</td>
<td>Central Virginia to lower Bay of Fundy</td>
<td>2,368</td>
<td>0.48</td>
<td></td>
</tr>
<tr>
<td>Aug–Sep 2016</td>
<td>Bay of Fundy/Scotian Shelf</td>
<td>1,894</td>
<td>0.4</td>
<td></td>
</tr>
<tr>
<td>Aug–Sep 2016</td>
<td>Central Virginia to lower Bay of Fundy + (Bay of Fundy/Scotia Shelf * 2/3 * 0.6)</td>
<td>3,110</td>
<td>0.38</td>
<td></td>
</tr>
<tr>
<td>Mid-summer 2016</td>
<td>State-space mark-recapture estimates (CMR; Figure 2)</td>
<td>1,396</td>
<td>n/a</td>
<td>0.015</td>
</tr>
</tbody>
</table>

Current Population Trend

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

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

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

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

POTENTIAL BIOLOGICAL REMOVAL

Potential Biological Removal (PBR) is the product of minimum population size, one-half the maximum productivity rate, and a recovery factor (MMPA Sec. 3. 16 U.S.C. 1362; Wade and Angliss 1997). The minimum population size for the Gulf of Maine stock is 1,380–375 whales. The maximum productivity rate is 0.065 (based on Barlow and Clapham 1997). In the 2015 and prior SARs, the recovery factor was 0.10 because this stock was listed as an endangered species under the Endangered Species Act (ESA). The 2016 revision to the ESA listing of humpback whales concluded that the West Indies Distinct Population Segment (of which the Gulf of Maine stock is a part) did not warrant listing (81 FR 62259, September 8, 2016). Consequently, in the 2016 SAR the recovery factor was revised to 0.5, the default value for stocks of unknown status relative to OSP (Wade and Angliss 1997). PBR for the Gulf of Maine humpback whale stock is 22.
Table 2. Best and minimum abundance estimates for the Gulf of Maine humpback whale (Megaptera novaeangliae) with Maximum Productivity Rate (Rmax), Recovery Factor (Fr.) and PBR.

<table>
<thead>
<tr>
<th>Nbest</th>
<th>95% Credible Interval</th>
<th>CV</th>
<th>Nmin</th>
<th>Fr</th>
<th>Rmax</th>
<th>PBR</th>
</tr>
</thead>
<tbody>
<tr>
<td>1,393</td>
<td>1,351–1,433</td>
<td>.015</td>
<td>1,375</td>
<td>0.5</td>
<td>0.065</td>
<td>22</td>
</tr>
</tbody>
</table>

ANNUAL HUMAN-CAUSED SERIOUS INJURY AND MORTALITY

For the period 2013 through 2017, the average minimum annual rate of detected (i.e. observed) human-caused mortality and serious injury to the Gulf of Maine humpback whale stock averaged 12.15 –15.25 animals per year. This value includes incidental fishery interaction records, 7.75–9.45; and records of vessel collisions, 4.45–8. (Table 2; Henry et al. 2020 in review). Although disentanglement is often unsuccessful or not possible for many cases, there are several documented cases of entanglements for which the intervention of disentanglement teams averted a likely prevented mortality or serious injury determination. Twenty-four serious injuries were prevented by intervention during 2013–2017 for an average of 5.2 cases per year during the period (Henry et al. 2020). Injury determinations are made based upon the best available information; these determinations may change with the availability of new information (Henry et al. in review). Only records considered to be confirmed human-causes mortalities or serious injuries are reported in the observed mortality and serious injury (M/SI) rows of Table 3.

Table 3: Average annual observed and estimated human-caused and natural mortality and serious injury for the Gulf of Maine humpback whale (Megaptera novaeangliae). Observed values are from confirmed interactions. Estimated total mortality value is model-derived (Pace et al. 2017; Robbins and Pace in prep.).

<table>
<thead>
<tr>
<th>Years</th>
<th>Source</th>
<th>Average Annual Avg.</th>
</tr>
</thead>
<tbody>
<tr>
<td>2014–2018</td>
<td>Observed incidental fishery interactions</td>
<td>9.45</td>
</tr>
<tr>
<td>2014–2018</td>
<td>Observed vessel collisions</td>
<td>5.8</td>
</tr>
<tr>
<td>2014–2018 TOTAL</td>
<td>Observed total human-caused M/SI</td>
<td>15.25</td>
</tr>
<tr>
<td>2014–2018</td>
<td>Observed natural mortality</td>
<td>1.8</td>
</tr>
<tr>
<td>2011–2015</td>
<td>Estimated total mortality</td>
<td>57.6</td>
</tr>
<tr>
<td>2014–2018</td>
<td>SI prevented</td>
<td>5.2</td>
</tr>
</tbody>
</table>

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

It is important to stress that serious injury determinations are made based upon the best available information; these determinations may change with the availability of new information (Henry et al. 2020). For the purposes of this report, takes against PBR are limited to those records considered confirmed human-caused mortalities or serious injuries.

Annual rates calculated from detected mortalities should be considered a low-biased, negatively-biased accounting of human-caused mortality; they represent a definitive lower bound. Detections of mortality and serious injury are haphazard, incomplete, and not the result of a designed sampling scheme. A key uncertainty is the fraction of the actual human-caused mortality and serious injury represented by the detected mortalities and serious injuries. Research
on small cetaceans has shown the actual number of deaths can be several times higher than that observed (Wells and Allen 2015; Williams et al. 2011). The hierarchical Bayesian, state-space model used to estimate North Atlantic right whale and humpback whale abundance (Pace et al. 2017; Hayes et al. 2020; Robbins and Pace in prep.) can also be used to estimate total mortality. A comparison of estimated and observed mortality values generated for the years 2000–2015 is presented in Figure 4. The detection rate for the 5-year period 2011–2015 was approximately 19% of all confirmed serious injury and mortality cases (natural and anthropogenic). The estimated annual rate of total mortality using this modeling approach is 57.6 animals for the period 2011–2015. The estimated mortality for 2016 forward is not yet available because it is contingent on the population estimate, which, in turn, is contingent on the processing of all photographs collected through 2020 for incorporation into the state-space model of the sighting histories of individual whales. This will be updated in the next stock assessment report. At this time, we are unable to apportion estimated mortality by cause (fishery interaction versus vessel strike) or by nationality (occurring in U.S. versus Canadian waters). However, because most of the observed mortality for humpback whales is human-caused, we conclude that the human-caused mortality in humpback whales currently is greater than PBR. Based on the proportion of observed human-caused M/SI of 89.4% (for the period 2014-2018), the average annual estimated human-caused M/SI would be 51.5 animals. However, only 38% of the total estimated mortality (57.6) would need to be human-caused for it to be greater than PBR (22). A protocol for apportioning mortality by cause is under development for use in the next stock assessment report.

Because annual population estimates are now available (Pace et al. 2017), it is possible to estimate total annual mortality (and the number of undetected).

\[
N_{t+1} = N_t + B_t - D_t
\]

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

Solving for \(D_t\) yields:

\[
D_t = N_t + B_t - N_{t+1}\]

which can then be used to estimate undetected mortality as: \(D_t - \text{observed deaths} = \text{undetected deaths}\)

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

There is mounting evidence that humpback whales have been over PBR for some time, and likely will be formally determined to be so in a future report. This is further supported by the NMFS declaration of Unusual Mortality Event No. 63.7 which includes cases from 2016 to the time of this writing in 2019 (https://www.fisheries.noaa.gov/national/marine-life-distress/2016-2019-humpback-whale-unusual-mortality-event-along-atlantic-coast). The literature and review of records described here suggest that there are significant human impacts beyond those recorded in the data assessed for serious injury and mortality. For example, a study of entanglement-related scarring on the caudal peduncle of 134 individual humpback whales in the Gulf of Maine suggested that between 48% and 65% had experienced entanglements (Robbins and Mattila 2001) and that 12-16% encounter gear annually (Robbins 2012).
To better assess human impacts (both vessel collision and commercial fishery mortality and serious injury) there needs to be greater emphasis on the timely recovery of carcasses and complete necropsies. Decomposed and/or unexamined animals (e.g., carcasses reported but not retrieved or no necropsy performed) represent 'lost data', some of which may relate to human impacts.

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

Background

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

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

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

Fishery-Related Serious Injuries and Mortalities

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

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

Although disentanglement is often unsuccessful or not possible for many cases, there are several documented cases of entanglements for which the intervention of disentanglement teams averted a likely serious injury determination. Twenty-four serious injuries were prevented by intervention during 2013–2017 (Henry et al. 2020).
<table>
<thead>
<tr>
<th>Date</th>
<th>Injured Determination</th>
<th>ID</th>
<th>Location</th>
<th>Assigned Cause</th>
<th>Value against PBR</th>
<th>Country</th>
<th>Gear Type</th>
<th>Description</th>
</tr>
</thead>
<tbody>
<tr>
<td>3-Apr-13</td>
<td>Mortality</td>
<td>-</td>
<td>off Ft Story, VA</td>
<td>VS</td>
<td>1</td>
<td>US</td>
<td>US</td>
<td>Fractured orbitals &amp; ribs w/ associated bruising</td>
</tr>
<tr>
<td>13-Sep-13</td>
<td>Mortality</td>
<td>-</td>
<td>York River, VA</td>
<td>VS</td>
<td>1</td>
<td>US</td>
<td>US</td>
<td>6 lacerations penetrate into muscle w/ associated hemorrhaging</td>
</tr>
<tr>
<td>16-Sep-13</td>
<td>Prorated Injury</td>
<td>-</td>
<td>off Chatham, MA</td>
<td>EN</td>
<td>0.75</td>
<td>XU</td>
<td>NR</td>
<td>Partial disentanglement; original &amp; final configurations unknown</td>
</tr>
<tr>
<td>28-Sep-13</td>
<td>Mortality</td>
<td>-</td>
<td>off Sallaire, NY</td>
<td>EN</td>
<td>1</td>
<td>XU</td>
<td>GN</td>
<td>Embedded line in mouth w/ associated hemorrhaging &amp; necrosis; evidence of constriction at pectorals, peduncle &amp; fluke w/ associated hemorrhaging; emaciated. Previously reported as GU.</td>
</tr>
<tr>
<td>4-Oct-13</td>
<td>Serious Injury</td>
<td>-</td>
<td>off Chatham, MA</td>
<td>EN</td>
<td>1</td>
<td>XU</td>
<td>NR</td>
<td>Full configuration unknown, but evidence of health decline; emaciation &amp; pale skin</td>
</tr>
<tr>
<td>02-Jun-14</td>
<td>Prorated Injury</td>
<td>-</td>
<td>15 mi E of Monomoy Island, MA</td>
<td>EN</td>
<td>0.75</td>
<td>XU</td>
<td>NR</td>
<td>Free-swimming with buoy and highflier trailing 100 ft aft of flukes. Attachment point(s) unknown. Unable to confirm if resighted on 21 Jun 2014.</td>
</tr>
<tr>
<td>Date</td>
<td>Injury Determination</td>
<td>ID</td>
<td>Location</td>
<td>Assigned Cause</td>
<td>Value against PBR</td>
<td>Country</td>
<td>Gear Type</td>
<td>Description</td>
</tr>
<tr>
<td>---------</td>
<td>----------------------</td>
<td>----</td>
<td>-------------------</td>
<td>----------------</td>
<td>-------------------</td>
<td>---------</td>
<td>-----------</td>
<td>-----------------------------------------------------------------------------</td>
</tr>
<tr>
<td>21-Jun-14</td>
<td>Prorated Injury</td>
<td>-</td>
<td>5 mi E of Gloucester, MA</td>
<td>EN</td>
<td>0.75</td>
<td>XU</td>
<td>NR</td>
<td>Free-swimming, trailing a buoy and possibly another buoy/high flyer aft. Attachment point(s) unknown. Unable to confirm if this is a resight of 02 Jun 2014.</td>
</tr>
<tr>
<td>18-Jul-14</td>
<td>Serious Injury</td>
<td>-</td>
<td>Provincetown Harbor, MA</td>
<td>EN</td>
<td>1</td>
<td>XU</td>
<td>NR</td>
<td>Free-swimming, trailing short amount of line from left side of mouth. No other gear noted, but evidence of previously more complicated, constricting entanglement. Current configuration deemed non-life threatening. Unsuccessful disentanglement attempt. In poor condition - emaciated with some cyanids. No resights.</td>
</tr>
<tr>
<td>03-Sep-14</td>
<td>Prorated Injury</td>
<td>-</td>
<td>off Long Island Beach, NJ</td>
<td>EN</td>
<td>.75</td>
<td>XU</td>
<td>NE</td>
<td>Full/final config. unknown. Seen with new vessel strike lacerations on 14 Aug 2014. No resights. Previously reported as gear unknown and being gear free (SI value=0) but gear status determined to be unconfirmed.</td>
</tr>
<tr>
<td>Date</td>
<td>Determination</td>
<td>ID</td>
<td>Location</td>
<td>Assigned Cause</td>
<td>Value against PBR</td>
<td>Country</td>
<td>Gear Type</td>
<td>Description</td>
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<tr>
<td>Date</td>
<td>Injury Determination</td>
<td>ID</td>
<td>Location</td>
<td>Assigned Cause</td>
<td>Value against PBR</td>
<td>Country</td>
<td>Gear Type</td>
<td>Description</td>
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<tr>
<td>01-Oct-14</td>
<td>Prorated Injury</td>
<td>-</td>
<td>15 mi E of Metompkin Inlet, VA</td>
<td>EN</td>
<td>.75</td>
<td>XU</td>
<td>NR</td>
<td>Free-swimming whale with line &amp;/or netting on left fluke blade. Gear appeared heavy. Full configuration unknown. No resights.</td>
</tr>
<tr>
<td>15-Dec-14</td>
<td>Prorated Injury</td>
<td>-</td>
<td>8.5 nm S of Grand Manan, NB</td>
<td>EN</td>
<td>.75</td>
<td>CN</td>
<td>PT</td>
<td>Fisherman found animal entangled in trawl. Grappled line, animal dove. Upon surfacing, appeared free of gear, but unable to confirm gear free. Original and final configuration unknown. Previously reported as XC.</td>
</tr>
<tr>
<td>25-Dec-14</td>
<td>Mortality</td>
<td>Triomphe</td>
<td>Little Cranberry Island, ME</td>
<td>EN</td>
<td>1</td>
<td>XU</td>
<td>NP</td>
<td>Fresh carcass with evidence of extensive constricting entanglement. No necropsy, but robust body condition and histopathology results of samples support EN as COD.</td>
</tr>
<tr>
<td>01-Feb-15</td>
<td>Serious Injury</td>
<td>-</td>
<td>off Beaufort, NC</td>
<td>EN</td>
<td>1</td>
<td>XU</td>
<td>NE</td>
<td>Constricting wrap at fluke insertion with line and monofilament netting trailing from flukes. Partial disentanglement by fisherman. Left with embedded gear and at least 40 ft of trailing line and netting. Unknown if there are additional attachment points. No resights. Gear previously reported as NR.</td>
</tr>
<tr>
<td>Date</td>
<td>Injury Determination</td>
<td>ID</td>
<td>Location</td>
<td>Assigned Cause</td>
<td>Value against PBR</td>
<td>Country</td>
<td>Gear Type</td>
<td>Description</td>
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<tr>
<td>03-Feb-15</td>
<td>Mortality</td>
<td>-</td>
<td>Corolla, NC</td>
<td>EN</td>
<td>1</td>
<td>US</td>
<td>NP</td>
<td>Fresh carcass with injuries consistent with constricting gear. No gear present. Full stomach indicating fed recently. COD likely peracute under water entrapment.</td>
</tr>
<tr>
<td>13-Apr-15</td>
<td>Mortality</td>
<td>-</td>
<td>off Fire Island, NY</td>
<td>VS</td>
<td>1</td>
<td>US</td>
<td>-</td>
<td>Extensive bruising and hemorrhaging at left gape and pectoral, throat, and right and left lateral thorax.</td>
</tr>
<tr>
<td>18-Apr-15</td>
<td>Mortality</td>
<td>-</td>
<td>Smith Point, NY</td>
<td>VS</td>
<td>1</td>
<td>US</td>
<td>-</td>
<td>Multifocal hemorrhage and edema in right lateral abdomen.</td>
</tr>
<tr>
<td>09-Jul-15</td>
<td>Prorated Injury</td>
<td>-</td>
<td>off Sandy Hook, NJ</td>
<td>EN</td>
<td>0.75</td>
<td>XU</td>
<td>NR</td>
<td>High flier trailing 30 ft aft of flukes. Attachment point(s) and configuration unknown. No resights.</td>
</tr>
<tr>
<td>Date</td>
<td>Injury Determination</td>
<td>ID</td>
<td>Location</td>
<td>Assigned Cause</td>
<td>Value against PBR</td>
<td>Country</td>
<td>Gear Type</td>
<td>Description</td>
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<tr>
<td>02-Aug-15</td>
<td>Serious Injury</td>
<td>-</td>
<td>off Race Point, Provincetown, MA</td>
<td>EN</td>
<td>1</td>
<td>XU</td>
<td>GN</td>
<td>Free-swimming with two sets of gear through its mouth: Primary gear=a closed bridle of gillnet joining mid-belly and trailing just past flukes and restricting movement; Secondary gear=an open bridle with one end leading to a buoy and the other to a pot. Disentangled from both sets of gear. Left with very short amount of gillnet through mouth that is expected to shed. Emaciated. No resights. Gillnet is primary cause of injury and of unknown origin. Pot/trap is US gear.</td>
</tr>
<tr>
<td>02-Aug-15</td>
<td>Prorated Injury</td>
<td>-</td>
<td>off Chatham, MA</td>
<td>EN</td>
<td>0.75</td>
<td>XU</td>
<td>NR</td>
<td>Calf with line around tail leading to buoys 4 ft aft of flukes. Full configuration unknown. No resights post 22Aug2015.</td>
</tr>
<tr>
<td>07-Sep-15</td>
<td>Prorated Injury</td>
<td>-</td>
<td>off Race Point, Provincetown, MA</td>
<td>EN</td>
<td>0.75</td>
<td>XU</td>
<td>MF</td>
<td>Monofilament line trailing from flukes. Attachment point(s) and configuration unknown. No resights.</td>
</tr>
<tr>
<td>Date&lt;sup&gt;b&lt;/sup&gt;</td>
<td>Injury Determination</td>
<td>ID</td>
<td>Location&lt;sup&gt;b&lt;/sup&gt;</td>
<td>Assigned Cause</td>
<td>Value against PBR&lt;sup&gt;c&lt;/sup&gt;</td>
<td>Country&lt;sup&gt;d&lt;/sup&gt;</td>
<td>Gear Type&lt;sup&gt;e&lt;/sup&gt;</td>
<td>Description</td>
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<tr>
<td>24-Sep-15</td>
<td>Prorated Injury</td>
<td>-</td>
<td>off Hampton, NH</td>
<td>EN</td>
<td>0.75</td>
<td>US</td>
<td>Anchor system</td>
<td>Became entangled in anchor line of fishing vessel during the night. Believed to be towing the entire system--45 lb anchor, 20 ft of chain, 350 ft of anchor line, 150 ft of float line, polyball and acorn buoy--in an unknown configuration. No resights.</td>
</tr>
<tr>
<td>25-Sep-15</td>
<td>Serious Injury</td>
<td>-</td>
<td>off Menemsha Harbor, MA</td>
<td>EN</td>
<td>1</td>
<td>XU</td>
<td>NR</td>
<td>Evidence of constricting body wrap, unable to confirm if gear embedded. Trailing 10 ft of line from flukes, full configuration unknown. Animal emaciated with heavy cymids. No resights.</td>
</tr>
<tr>
<td>04-Dec-15</td>
<td>Prorated Injury</td>
<td>-</td>
<td>off Brier Island, NS</td>
<td>EN</td>
<td>0.75</td>
<td>CN</td>
<td>PT</td>
<td>Likely anchored in gear. Partially disentangled by fishermen. Left free-swimming with a body wrap aft of blowholes and 2 balloon floats close to body. Final configuration unknown. No resights.</td>
</tr>
<tr>
<td>Date&lt;br&gt;15-Dec-15</td>
<td>Injury Determination</td>
<td>ID</td>
<td>Location&lt;br&gt;off North East Harbour, NS</td>
<td>Assigned Cause</td>
<td>Value against PBR</td>
<td>Country&lt;br&gt;CN</td>
<td>Gear Type&lt;br&gt;PT</td>
<td>Description</td>
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<tr>
<td>Prorated Injury</td>
<td>-</td>
<td></td>
<td></td>
<td>EN</td>
<td>0.75</td>
<td></td>
<td></td>
<td>Likely anchored in gear. Partially disentangled by fishermen. Left free-swimming with buoy and lines around front of whale and lines on the peduncle. Attachment point(s) and final configuration unknown. No resights.</td>
</tr>
<tr>
<td>07-Jan-16</td>
<td>Prorated Injury</td>
<td>--</td>
<td>off Greenwich, CT</td>
<td>EN</td>
<td>0.75</td>
<td>US</td>
<td>PT</td>
<td>Anchored in gear with line through mouth and around tail. Partially disentangled - all gear removed from mouth and some from tail. Post intervention whale was using pectorals to swim and tail was down, but unable to confirm if any gear remained and in what configuration. No resights.</td>
</tr>
<tr>
<td>09-Jan-16</td>
<td>Serious Injury</td>
<td>MAHWC-254</td>
<td>off Fort Story, VA</td>
<td>VS</td>
<td>1</td>
<td>US</td>
<td>-</td>
<td>Deep laceration across back - penetrating into muscle and impacting ability to dive. No resights.</td>
</tr>
<tr>
<td>03-Mar-16</td>
<td>Serious Injury</td>
<td>MAHWC-251</td>
<td>off Virginia Beach, VA</td>
<td>VS</td>
<td>1</td>
<td>US</td>
<td>-</td>
<td>Deep laceration on left fluke blade, near insertion. Fluke blade necrotic. No resights.</td>
</tr>
<tr>
<td>Date&lt;sup&gt;b&lt;/sup&gt;</td>
<td>Injury Determination</td>
<td>ID</td>
<td>Location&lt;sup&gt;b&lt;/sup&gt;</td>
<td>Assigned Cause</td>
<td>Value against PBR&lt;sup&gt;c&lt;/sup&gt;</td>
<td>Country&lt;sup&gt;d&lt;/sup&gt;</td>
<td>Gear Type&lt;sup&gt;e&lt;/sup&gt;</td>
<td>Description</td>
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<tr>
<td>24-Apr-16</td>
<td>Prorated Injury</td>
<td>-</td>
<td>off Race Point, Provincetown, MA</td>
<td>EN</td>
<td>0.75</td>
<td>XU</td>
<td>NR</td>
<td>Free-swimming with 2 buoys - submerged orange at 5 ft and white bullet at 10 ft - trailing behind flukes. Line appears to wrap flukes. Subsequent sighting only reported white buoy, but only one surfacing and no photos. Attachment point(s) and configuration unknown. No resights.</td>
</tr>
<tr>
<td>25-Apr-16</td>
<td>Mortality</td>
<td>-</td>
<td>Marshfield, MA</td>
<td>VS</td>
<td>1</td>
<td>US</td>
<td>-</td>
<td>Bruising deep to muscle and fascia by right pectoral and mandible at the base of the skull. Limited necropsy but depth and area of bruising consistent with blunt trauma from vessel strike.</td>
</tr>
<tr>
<td>25-Apr-16</td>
<td>Mortality</td>
<td>-</td>
<td>Napreague Bay, NY</td>
<td>VS</td>
<td>1</td>
<td>XU</td>
<td>-</td>
<td>Extensive bruising to ventral thoracic region along with fractured ribs.</td>
</tr>
<tr>
<td>Date</td>
<td>Injury Determination</td>
<td>ID</td>
<td>Location</td>
<td>Assigned Cause</td>
<td>Value against PBR</td>
<td>Country</td>
<td>Gear Type</td>
<td>Description</td>
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<tr>
<td>18-May-16</td>
<td>Serious Injury</td>
<td>Foggy</td>
<td>off Gloucester, MA</td>
<td>EN</td>
<td>1</td>
<td>XU</td>
<td>GU</td>
<td>Anchored with lines through mouth and 2 embedded body wraps with large float alongside right body. Entangling gear fouled in 2 other sets of gear. Animal in emaciated. Partial disentanglement - left with an open bridle of 2 lines through the mouth. Subsequent sightings show lines had relooped into a closed bridle and health continued to decline. No resights post July 2016.</td>
</tr>
<tr>
<td>21-May-16</td>
<td>Prorated Injury</td>
<td>-</td>
<td>off Mantoloking, NJ</td>
<td>EN</td>
<td>0.75</td>
<td>XU</td>
<td>GN</td>
<td>Full configuration unknown, but minimally wrapped in gear from head to dorsal. Unknown amount of gear removed by public. Unable to confirm if gear free. No resights.</td>
</tr>
<tr>
<td>24-Jun-16</td>
<td>Mortality</td>
<td>-</td>
<td>off Shinnecock Inlet, NY</td>
<td>VS</td>
<td>1</td>
<td>US</td>
<td>-</td>
<td>Extensive bruising to connective tissue and muscles of the left side, back, and right peduncle.</td>
</tr>
<tr>
<td>Date</td>
<td>Injury Determination</td>
<td>ID</td>
<td>Location</td>
<td>Assigned Cause</td>
<td>Value against PBR</td>
<td>Country</td>
<td>Gear Type</td>
<td>Description</td>
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<tr>
<td>26-Jun-16</td>
<td>Mortality</td>
<td>Snowplow</td>
<td>off Rockport, MA</td>
<td>VS</td>
<td>1</td>
<td>US</td>
<td>-</td>
<td>Limited necropsy, but significant evidence of blunt trauma to left head and pectoral consistent with vessel strike.</td>
</tr>
<tr>
<td>02-Sep-16</td>
<td>Prorated Injury</td>
<td>-</td>
<td>off Gloucester, MA</td>
<td>EN</td>
<td>0.75</td>
<td>XU</td>
<td>NR</td>
<td>Free-swimming and trailing red buoy. Attachment point(s) and configuration unknown. No resights.</td>
</tr>
<tr>
<td>10-Sep-16</td>
<td>Mortality</td>
<td>-</td>
<td>Martha's Vineyard, MA</td>
<td>EN</td>
<td>1</td>
<td>XU</td>
<td>NP</td>
<td>No gear present, but evidence of constricting entanglement with associated reactive tissue at fluke insertions. State of decomposition at time of exam precluded COD determination, but injuries and thin blubber layer are consistent with chronic entanglement.</td>
</tr>
<tr>
<td>Date</td>
<td>Injury Determination</td>
<td>ID</td>
<td>Location</td>
<td>Assigned Cause</td>
<td>Value against PBR</td>
<td>Country</td>
<td>Gear Type</td>
<td>Description</td>
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</tr>
<tr>
<td>16-Oct-16</td>
<td>Mortality</td>
<td>GOM-1626</td>
<td>off Ipswich, MA</td>
<td>EN</td>
<td>1</td>
<td>US</td>
<td>PT</td>
<td>No necropsy, but extensive entanglement. Line through mouth with constricting wraps on both flippers, body, and peduncle. Entanglement as COD most parsimonious. Confirmed as same individual released from weir on 27Sep2016.</td>
</tr>
<tr>
<td>13-Nov-16</td>
<td>Prorated Injury</td>
<td>NYC0052</td>
<td>off Belmar, NJ</td>
<td>EN</td>
<td>0.75</td>
<td>XU</td>
<td>MF</td>
<td>Free-swimming with monofilament over peduncle and trailing from flukes. Attachment point(s) and configuration unknown. No resights.</td>
</tr>
<tr>
<td>14-Nov-16</td>
<td>Prorated Injury</td>
<td>-</td>
<td>off Stone Harbor, NJ</td>
<td>EN</td>
<td>0.75</td>
<td>XUS</td>
<td>PT</td>
<td>Free-swimming with line wrapping left flipper and flukes and trailing. Full configuration unclear. No resights. Previously reported as XC, gear not recovered.</td>
</tr>
<tr>
<td>04-Dec-16</td>
<td>Prorated Injury</td>
<td>-</td>
<td>off Quogue, NY</td>
<td>EN</td>
<td>0.75</td>
<td>XU</td>
<td>NR</td>
<td>Free-swimming with high flier near flukes. Attachment point(s) and configuration unknown. No resights.</td>
</tr>
<tr>
<td>16-Dec-16</td>
<td>Mortality</td>
<td>HDRVA0 78</td>
<td>off Dam Neck, VA</td>
<td>EN</td>
<td>1</td>
<td>US</td>
<td>NP</td>
<td>No gear present, but evidence of extensive constricting entanglement. Fresh carcass with digestive system full of fish. COD dry drowning due to entanglement.</td>
</tr>
<tr>
<td>Date</td>
<td>Injury Determination</td>
<td>ID</td>
<td>Location</td>
<td>Assigned Cause</td>
<td>Value against PBR</td>
<td>Country</td>
<td>Gear Type</td>
<td>Description</td>
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</tr>
<tr>
<td>19-Dec-16</td>
<td>Prorated Injury</td>
<td></td>
<td>off Tiverton, NS</td>
<td>EN</td>
<td>0.75</td>
<td>SureXC</td>
<td>NR</td>
<td>Free-swimming with line around tail and buoy trailing. Full configuration unknown. No resights.</td>
</tr>
<tr>
<td>02-Feb-17</td>
<td>Mortality</td>
<td></td>
<td>Chesapeake Bay, VA</td>
<td>VS</td>
<td>1</td>
<td>US</td>
<td>-</td>
<td>Four lacerations that penetrated body cavity. Robust condition with full stomach. COD exsanguination and asphyxia from sharp trauma consistent with vessel strike.</td>
</tr>
<tr>
<td>05-Feb-17</td>
<td>Mortality</td>
<td></td>
<td>Chesapeake Bay, VA</td>
<td>VS</td>
<td>1</td>
<td>US</td>
<td>-</td>
<td>Extensive skeletal fracturing with associated hemorrhaging consistent with blunt trauma from vessel strike.</td>
</tr>
<tr>
<td>11-Feb-17</td>
<td>Mortality</td>
<td></td>
<td>Fort Story, VA</td>
<td>VS</td>
<td>1</td>
<td>US</td>
<td>-</td>
<td>Three lacerations that penetrated body cavity. Robust condition with full stomach. COD exsanguination from sharp trauma consistent with vessel strike.</td>
</tr>
<tr>
<td>14-Feb-17</td>
<td>Serious Injury</td>
<td></td>
<td>Virginia Beach, VA</td>
<td>VS</td>
<td>1</td>
<td>US</td>
<td>-</td>
<td>Two new, deep lacerations fore and aft of dorsal fin. No resights.</td>
</tr>
<tr>
<td>03-Apr-17</td>
<td>Mortality</td>
<td></td>
<td>Rockaway, NY</td>
<td>VS</td>
<td>1</td>
<td>US</td>
<td>-</td>
<td>Extensive hemorrhage and edema along back and side consistent with blunt trauma from vessel strike.</td>
</tr>
<tr>
<td>Date(^b)</td>
<td>Injury Determination</td>
<td>ID</td>
<td>Location(^b)</td>
<td>Assigned Cause</td>
<td>Value against PBR(^c)</td>
<td>Country(^d)</td>
<td>Gear Type(^e)</td>
<td>Description</td>
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</tr>
<tr>
<td>04-May-17</td>
<td>Mortality</td>
<td>-</td>
<td>Rehobeth Beach, DE</td>
<td>VS</td>
<td>1</td>
<td>US</td>
<td>-</td>
<td>Disarticulated left jaw and cervical vertebrae with associated hemorrhaging. Limited necropsy but injuries consistent with blunt trauma from vessel strike.</td>
</tr>
<tr>
<td>15-Jun-17</td>
<td>Mortality</td>
<td>-</td>
<td>Jamestown, RI</td>
<td>VS</td>
<td>1</td>
<td>US</td>
<td>-</td>
<td>Muscle contusions and associated cranial fractures consistent with blunt trauma from vessel strike.</td>
</tr>
<tr>
<td>18-Jun-17</td>
<td>Mortality</td>
<td>GOM-1625</td>
<td>Chatham, MA</td>
<td>EN</td>
<td>1</td>
<td>XU</td>
<td>NP</td>
<td>No gear present, but evidence of constricting entanglement with associated hemorrhaging at insertion of pectorals and fluke. Poor health condition.</td>
</tr>
<tr>
<td>15-Jul-17</td>
<td>Prorated Injury</td>
<td>2016 Calf of Thumper</td>
<td>off Race Point, Provincetown, MA</td>
<td>EN</td>
<td>.75</td>
<td>US</td>
<td>NR</td>
<td>Free-swimming with hook and monofilament trailing from right fluke blade. Attachment point(s) and full configuration unknown. No resights.</td>
</tr>
<tr>
<td>Date&lt;sup&gt;b&lt;/sup&gt;</td>
<td>Injury Determination</td>
<td>ID</td>
<td>Location&lt;sup&gt;b&lt;/sup&gt;</td>
<td>Assigned Cause</td>
<td>Value against PBR&lt;sup&gt;c&lt;/sup&gt;</td>
<td>Country&lt;sup&gt;d&lt;/sup&gt;</td>
<td>Gear Type&lt;sup&gt;e&lt;/sup&gt;</td>
<td>Description</td>
</tr>
<tr>
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<td>--------------------------------------------------------------------------</td>
</tr>
<tr>
<td>19-Aug-17</td>
<td>Prorated Injury</td>
<td>-</td>
<td>off Long Island, NY</td>
<td>EN</td>
<td>.75</td>
<td>XU</td>
<td>NR</td>
<td>Free-swimming with buoy trailing aft of flukes. Attachment point(s) and configuration unknown. No resights post 11Sep2017.</td>
</tr>
<tr>
<td>18-Sep-17</td>
<td>Prorated Injury</td>
<td>-</td>
<td>off Jonesport, ME</td>
<td>EN</td>
<td>.75</td>
<td>CN</td>
<td>PT</td>
<td>Anchored in gear. Fisher responded later, animal not relocated and gear missing section of pots and line. Final configuration unknown. No resights.</td>
</tr>
<tr>
<td>01-Oct-17</td>
<td>Mortality</td>
<td>-</td>
<td>off Narragansett, RI</td>
<td>VS</td>
<td>1</td>
<td>XU</td>
<td>-</td>
<td>Hemorrhaging along dorsal and left side consistent with blunt trauma from vessel strike.</td>
</tr>
<tr>
<td>14-Oct-17</td>
<td>Prorated Injury</td>
<td>-</td>
<td>off Race Point, Provincetown, MA</td>
<td>EN</td>
<td>.75</td>
<td>XU</td>
<td>NR</td>
<td>Free-swimming with buoy along right flank. Attachment point(s) and full configuration unknown. No resights.</td>
</tr>
<tr>
<td>21-Oct-17</td>
<td>Prorated Injury</td>
<td>GOM-1747</td>
<td>off Long Island, NY</td>
<td>EN</td>
<td>.75</td>
<td>XU</td>
<td>NR</td>
<td>Free-swimming with buoy in tow. Attachment point(s) and full configuration unknown. No resights.</td>
</tr>
<tr>
<td>Date</td>
<td>Inj/Determ</td>
<td>ID</td>
<td>Location</td>
<td>Assigned Cause</td>
<td>Value against PBR</td>
<td>Country</td>
<td>Gear Type</td>
<td>Description</td>
</tr>
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<td>-----------------------------------------------------------------------------</td>
</tr>
<tr>
<td>12-Nov-17</td>
<td>Prorated Injury</td>
<td>-</td>
<td>off Atlantic Beach, NY</td>
<td>EN</td>
<td>.75</td>
<td>US</td>
<td>MF</td>
<td>Free-swimming with monofilament trailing from right fluke. Attachment point(s) and full configuration unknown. No resights.</td>
</tr>
<tr>
<td>30-Nov-17</td>
<td>Prorated Injury</td>
<td>-</td>
<td>off Grand Manan, NB</td>
<td>EN</td>
<td>.75</td>
<td>CN</td>
<td>PT</td>
<td>Anchored at tail area, partially disentangled. Unable to confirm gear free or that all gear recovered. Final configuration unknown. No resights.</td>
</tr>
<tr>
<td>26-Dec-17</td>
<td>Mortality</td>
<td>-</td>
<td>East Atlantic Beach, NY</td>
<td>VS</td>
<td>1</td>
<td>US</td>
<td>-</td>
<td>Extensive bruising and edema on both sides of body consistent with blunt trauma from vessel strike.</td>
</tr>
<tr>
<td>1/28/201</td>
<td>Mortality</td>
<td>Peters Point, FL</td>
<td>VS</td>
<td>1</td>
<td>US</td>
<td></td>
<td></td>
<td>Hemorrhage with associated skeletal fractures consistent with blunt force trauma.</td>
</tr>
<tr>
<td>2/12/2018</td>
<td>Mortality</td>
<td>Breezy Point, NY</td>
<td>VS</td>
<td>1</td>
<td>US</td>
<td></td>
<td></td>
<td>Hemorrhage and edema along back and sides consistent with blunt force trauma.</td>
</tr>
<tr>
<td>5/5/2018</td>
<td>Mortality</td>
<td>Raritan Bay, NJ</td>
<td>VS</td>
<td>1</td>
<td>US</td>
<td></td>
<td></td>
<td>Extensive edema in musculature along sides and ventral surface consistent with blunt force trauma.</td>
</tr>
<tr>
<td>5/18/2018</td>
<td>Mortality</td>
<td>Long Beach, NY</td>
<td>VS</td>
<td>1</td>
<td>US</td>
<td></td>
<td></td>
<td>Extensive hemorrhage and edema along dorsal surface consistent with blunt force trauma.</td>
</tr>
<tr>
<td>Date</td>
<td>Injury Determination</td>
<td>ID</td>
<td>Location</td>
<td>Assigned Cause</td>
<td>Value against PBR</td>
<td>Country</td>
<td>Gear Type</td>
<td>Description</td>
</tr>
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<td>-------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------</td>
</tr>
<tr>
<td>6/1/2018</td>
<td>Mortality</td>
<td>-</td>
<td>Breezy Point, NY</td>
<td>VS</td>
<td>1</td>
<td>XU</td>
<td>-</td>
<td>Extensive hemorrhage and edema along dorsal, side, and ventral surfaces consistent with blunt force trauma.</td>
</tr>
<tr>
<td>7/14/2018</td>
<td>Serious Injury</td>
<td>2017 Calf OfRapier</td>
<td>5.8 nm W of Race Point, Provincetown, MA</td>
<td>EN</td>
<td>1</td>
<td>US</td>
<td>PT</td>
<td>Free-swimming with deeply embedded line at insertion of flukes with line trailing, buoy pinned near fluke. Partial disentanglement - remaining gear held in place by wounds, hope will shed. Compromised health - widespread cyanids, skin pitting. No resights.</td>
</tr>
<tr>
<td>7/14/2018</td>
<td>Serious Injury</td>
<td>-</td>
<td>0.5 nm S of Nantucket, MA</td>
<td>EN</td>
<td>1</td>
<td>XU</td>
<td>NR</td>
<td>Free-swimming with line crossing over back. Resight showed emaciated with line over back, a bundle of line by right shoulder, and line around peduncle. Suspect attachment points are mouth and/or flippers, but unable to confirm. No resights.</td>
</tr>
<tr>
<td>7/21/2018</td>
<td>Prorated Injury</td>
<td>Rhino</td>
<td>7.2 nm E of Hampton Beach, NH</td>
<td>EN</td>
<td>0.75</td>
<td>US</td>
<td>MF</td>
<td>Free-swimming with monofilament trailing from left fluke. Attachment point(s) and configuration unknown. No resights.</td>
</tr>
<tr>
<td>Date</td>
<td>Injury Determination</td>
<td>ID</td>
<td>Location</td>
<td>Assigned Cause</td>
<td>Value against PBRc</td>
<td>Countryd</td>
<td>Gear Typee</td>
<td>Description</td>
</tr>
<tr>
<td>---------</td>
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<td>-----------------------------------------------------------------------------</td>
</tr>
<tr>
<td>7/26/2018</td>
<td>Mortality</td>
<td>:</td>
<td>Napeague, NY</td>
<td>EN</td>
<td>1</td>
<td>XU</td>
<td>N</td>
<td>Multiple constricting wraps at fluke insertion, deeply embedded and partially severing flukes.</td>
</tr>
<tr>
<td>7/30/2018</td>
<td>Serious Injury</td>
<td>NYC0097</td>
<td>1.0 nm SE of Montauk, NY</td>
<td>EN</td>
<td>1</td>
<td>XU</td>
<td>NR</td>
<td>Free-swimming with line through mouth exiting right and trailing aft, exiting left and leading to multiple constricting wraps around left pectoral, then trailing 3-4ft. Partially embedded in flipper, large raw wound. No resights.</td>
</tr>
<tr>
<td>7/30/2018</td>
<td>Prorated Injury</td>
<td>Cardhu</td>
<td>8.2 nm NW of Race Point, Provincetown, MA</td>
<td>EN</td>
<td>0.75</td>
<td>US</td>
<td>AN</td>
<td>Free-swimming in anchoring system. Configuration of gear changed throughout event: Open bridle through mouth became closed bridle with multiple peduncle wraps. Partial disentanglement - removed mooring ball, anchor &amp; jumble of line aft of flukes. Final configuration unknown but included closed bridle trailing to a small jumble of line at flukes. No resights.</td>
</tr>
<tr>
<td>8/5/2018</td>
<td>Serious Injury</td>
<td>:</td>
<td>10 nm E of Long Island, NY</td>
<td>EN</td>
<td>1</td>
<td>XU</td>
<td>NR</td>
<td>Free-swimming, emaciated whale with constricting and embedded wraps at fluke insertion and on left fluke blade. Swimming by use of flippers. No resights.</td>
</tr>
<tr>
<td>Date</td>
<td>Injury Determination</td>
<td>ID</td>
<td>Location</td>
<td>Assigned Cause</td>
<td>Value against PBR</td>
<td>Country</td>
<td>Gear Type</td>
<td>Description</td>
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</tr>
<tr>
<td>8/11/2018</td>
<td>Serious Injury</td>
<td></td>
<td>Cape May, NJ</td>
<td>VS</td>
<td>1</td>
<td>US</td>
<td></td>
<td>At least 11 deep lacerations starting at dorsal fin and continuing down dorsal ridge towards fluke. Ulcerated lesion on right pectoral with some evidence of healing.</td>
</tr>
<tr>
<td>8/15/2018</td>
<td>Serious Injury</td>
<td></td>
<td>Stellwagen Bank</td>
<td>VS</td>
<td>1</td>
<td>US</td>
<td></td>
<td>130 ft whale watch vessel at 28 kts. No blood in water.</td>
</tr>
<tr>
<td>8/29/2018</td>
<td>Mortality</td>
<td>2016Calf Of Venom</td>
<td>1.5 nm E of Hampton Beach, NH</td>
<td>EN</td>
<td>1</td>
<td>US</td>
<td>NE</td>
<td>Free-swimming with netting, line, and floats around peduncle - configuration unclear. Disentangled by recreational boaters. Confirmed gear free on 03Sep, behaving abnormally. Carcass found on 07Sep, but no gear present. No necropsy conducted but primary COD from entanglement most parsimonious from abnormal behavior which is indicative of health decline. Ultimate COD unknown.</td>
</tr>
<tr>
<td>9/7/2018</td>
<td>Mortality</td>
<td>Peajack</td>
<td>off Brier Island, NS</td>
<td>EN</td>
<td>1</td>
<td>XC</td>
<td>PT</td>
<td>Anchored carcass with line through mouth leading to multiple constricting. No necropsy, but extensive constricting gear makes COD from entanglement most parsimonious.</td>
</tr>
<tr>
<td>Date</td>
<td>Injury Determination</td>
<td>ID</td>
<td>Location</td>
<td>Assigned Cause</td>
<td>Value against PBR</td>
<td>Country</td>
<td>Gear Type</td>
<td>Description</td>
</tr>
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<td>-----------------------------------------------------------------------------</td>
</tr>
<tr>
<td>9/8/2018</td>
<td>Prorated Injury</td>
<td>3.1 nm SE of Gloucester, MA</td>
<td>EN</td>
<td>0.75</td>
<td>XU</td>
<td>NR</td>
<td>Free-swimming with line and buoy trailing. Full configuration unknown. No resights.</td>
<td></td>
</tr>
<tr>
<td>9/21/2018</td>
<td>Prorated Injury</td>
<td>20 nm E of Rockport, MA</td>
<td>EN</td>
<td>0.75</td>
<td>XU</td>
<td>NR</td>
<td>Free-swimming with 2-3 buoys trailing 30-40ft aft of flukes. Attachment point(s) and full configuration unknown. No resights.</td>
<td></td>
</tr>
<tr>
<td>9/23/2018</td>
<td>Prorated Injury</td>
<td>10.5 nm SE of Gloucester, MA</td>
<td>EN</td>
<td>0.75</td>
<td>XU</td>
<td>NR</td>
<td>Free-swimming with line on fluke from unknown attachment point(s). No resights.</td>
<td></td>
</tr>
<tr>
<td>9/23/2018</td>
<td>Prorated Injury</td>
<td>14.1 nm S of Martha's Vineyard, MA</td>
<td>EN</td>
<td>0.75</td>
<td>XU</td>
<td>NR</td>
<td>Free-swimming with buoys trailing from unknown attachment point(s). No resights.</td>
<td></td>
</tr>
<tr>
<td>12/13/2018</td>
<td>Prorated Injury</td>
<td>0.7 nm E of Mayport, FL</td>
<td>EN</td>
<td>0.75</td>
<td>XU</td>
<td>NR</td>
<td>Free-swimming with unknown attachment point(s) and configuration. Partly disentangled by public. Final configuration unknown. No resights.</td>
<td></td>
</tr>
<tr>
<td>12/15/2018</td>
<td>Mortality</td>
<td>2016Calf Of Zeppeli II</td>
<td>Cape Point, Lewes, DE</td>
<td>VS</td>
<td>1</td>
<td>US</td>
<td>Skeletal fracturing with associated hemorrhaging. COD - blunt trauma from vessel strike.</td>
<td></td>
</tr>
<tr>
<td>Assigned Cause</td>
<td>Five-year mean (US/CN/XU/XC)</td>
<td></td>
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<tr>
<td>Vessel strike</td>
<td>4.45 (4.50/0.00/0.40/0.00)</td>
<td></td>
<td></td>
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<td></td>
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</tr>
<tr>
<td>Entanglement</td>
<td>7.75 (2.05/0.75/4.80/0.15)</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

a. For more details on events please see Henry et al. 2020 in review.
b. The date sighted and location provided in the table are not necessarily when or where the serious injury or mortality occurred; rather, this information indicates when and where the whale was first reported beached, entangled, or injured.
c. Mortality events are counted as 1 against PBR. Serious injury events have been evaluated using NMFS guidelines (NOAA 2012).
d. CN=Canada, US=United States, XC=Unassigned 1st sight in CN, XU=Unassigned 1st sight in US.
e. H=hook, GN=gillnet, GU=gear unidentifiable, MF=monofilament, NE=netting, NP=none present, NR=none recovered/received, PT=pot/trap, WE=weir.

**Other Mortality**

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

Between July and September 2003, an Unusual Mortality Event (UME) that included 16 humpback whales was invoked in offshore waters of coastal New England and the Gulf of Maine. Biotoxin analyses of samples taken from some of these whales found saxitoxin at very low/questionable levels and domoic acid at low levels, but neither were adequately documented and therefore no definitive conclusions could be drawn. Seven humpback whales were considered part of a large whale UME in New England in 2005. Twenty-one dead humpback whales found between 10 July and 31 December 2006 triggered a humpback whale UME declaration. Additionally, in January 2016 a humpback whale UME was declared for the U.S. Atlantic coast due to elevated numbers of mortalities. From January 2016 to December 2018, 85 humpback whales stranded between Maine and Florida. A portion of the whales have shown evidence of pre-mortem vessel strike; however, this finding is not consistent across all whales examined. (a total of 88 strandings in 2016–2018; https://www.fisheries.noaa.gov/national/marine-life-distress/2016-2018-humpback-whale-unusual-mortality-event-along-atlantic-coast).—This most recent UME is ongoing.

**HABITAT ISSUES**

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

**STATUS OF STOCK**

NMFS conducted a global status review of humpback whales (Bettridge et al. 2015) and recently revised the ESA listing of the species (81 FR 62259, September 8, 2016). The Distinct Population Segments (DPSs) that occur in waters under U.S. jurisdiction, as established in the Final Rule, do not necessarily equate to the existing MMPA stocks. NMFS is evaluating the stock structure of humpback whales under the MMPA, but no changes to current stock structure are proposed at this time. As noted within the humpback whale ESA-listing Final Rule, in the case of a species or stock that achieved its depleted status solely on the basis of its ESA status, such as the humpback whale, the species or stock would cease to qualify as depleted under the terms of the definition set forth in MMPA Section 3(1) if the species or stock is no longer listed as threatened or endangered. The final rule indicated that until the stock delineations are reviewed in light of the DPS designations, NMFS would consider stocks that do not fully or partially coincide with a listed DPS as not depleted for management purposes. Because it does not coincide with any ESA-listed DPS, the Gulf of Maine stock is not considered not depleted due to its ESA status, because it does not coincide with any ESA-listed DPS. The detected level of U.S. fishery human-caused mortality and serious injury derived from the available records, (average of 12.5 for 2013–2017) does not exceed the calculated PBR. However,
because the detected of 22 and, therefore, this is not a strategic stock if the recovery factor is set at 0.5. Because the observed mortality is estimated to be only 20-19% of all mortality (Figure 4), and total estimated human-caused average annual mortality and serious injury is 51.5 animals compared to PBR of 22, total annual mortality may be 60-70 animals in this stock. If anthropogenic causes are responsible for as little as 31% of potential total mortality, this stock could be over PBR. While detected mortalities yield an estimated minimum fraction of anthropogenic mortality as 0.85, additional research is being done before apportioning mortality to anthropogenic versus natural causes for undetected mortalities. Therefore, the accounting of human-caused mortality is biased low and the uncertainties associated with this assessment may have produced an incorrect determination of the stock is considered strategic status.

REFERENCES CITED


Robbins, J. and R.M. Pace, III. in prep. Abundance estimates and growth rates of Gulf of Maine humpback whales


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

**STOCK DEFINITION AND GEOGRAPHIC RANGE**

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

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

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

New England and Gulf of St. Lawrence waters represent a major feeding ground for fin whales. There is evidence of site fidelity by females, and perhaps some segregation by sexual, maturational, or reproductive class in the feeding area (Agler et al. 1993, Schleimer et al. 2019). Hain et al. (1992) showed that fin whales measured photogrammetrically off the northeastern U.S., after omitting all individuals smaller than 14.6 m (the smallest whale taken in Iceland), were significantly smaller (mean length=16.8 m; P <0.001) than fin whales taken in Icelandic whaling (mean=18.3 m). Seipt et al. (1990) reported that 49% of identified fin whales sighted on the Massachusetts Bay area feeding grounds were resighted within the same year, and 45% were resighted in multiple years. The authors suggested that fin whales on these grounds exhibited patterns of seasonal occurrence and annual return that in some respects were similar to those shown for humpback whales. This was reinforced by Clapham and Seipt (1991), who showed maternally-directed site fidelity for fin whales in the Gulf of Maine. Despite the suggested similarity in patterns of seasonal occurrence with humpback whales, the U.S. currently recognizes one stock of fin whales in the western North Atlantic.

Hain et al. (1992), based on an analysis of neonate stranding data, suggested that calving takes place during October to January in latitudes of the U.S. mid-Atlantic region; however, it is unknown where calving, mating, and wintering occur for most of the population. Results from the Navy’s SOSUS program (Clark 1995; Clark and Gagnon 2002) indicated a substantial deep-ocean distribution of fin whales. It is likely that fin whales occurring in the U.S. Atlantic EEZ undergo migrations into Canadian waters, open-ocean areas, and perhaps even subtropical or tropical regions (Edwards et al. 2015, 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 available for the western North Atlantic fin whales in the North Atlantic stock is 7,4186,802 (CV=0.2524). This estimate is the sum of the 2016 NOAA shipboard and aerial surveys and the 2016 NEFSC and Department of Fisheries and Oceans Canada (DFO) surveys (“Central Virginia to Newfoundland/Labrador (COMBINED)”) in Table 1, a Northwest Atlantic International Sightings Survey (NAISS). Because the survey areas did not overlap, the estimates from the two surveys were added together and the CVs pooled using a delta method to produce a species abundance estimate for the stock area. The 2016 estimate is larger than those from 2011 because the 2016 estimate is derived from a survey area extending from Newfoundland to Florida, which is about 1,300,000 km² larger than the 2011 survey area. In addition, the 2016 survey estimates in U.S. waters were corrected for availability bias (due to diving behavior), whereas the 2011 estimates were not corrected.

**Earlier abundance estimates**

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

**Recent surveys and abundance estimates**

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

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

An abundance estimate of 3,006 (CV=0.61) 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 38°N 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 38°N latitude between
the 100-m isobaths and the outer limit of the U.S. EEZ during 30 June–19 August. A total of 4,399 km of trackline
was covered on effort (NEFSC and SEFSC 2018). Both surveys utilized two visual teams and an independent observer
approach to estimate detection probability on the trackline (Laake and Borchers 2004). Mark-recapture distance
sampling was used to estimate abundance.

The Department of Fisheries and Oceans, Canada (DFO) generated fin whale estimates from a large-scale aerial
survey of Atlantic Canadian shelf and shelf break habitats extending from the northern tip of Labrador to the U.S.
border off southern Nova Scotia in August and September of 2016 (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 the Twin Otter data using two-team mark-recapture multi-covariate distance sampling methods. The
Gulf of St. Lawrence estimate was derived from the Skymaster data using single team multi-covariate distance
sampling with left truncation (to accommodate the obscured area under the plane) where size-bias was also
investigated, and the Otter-based perception bias correction was applied. An availability bias correction factor, which
was based on the cetaceans’ surface intervals, was applied to both abundance estimates.

<table>
<thead>
<tr>
<th>Month/Year</th>
<th>Area</th>
<th>Nbest</th>
<th>CV</th>
</tr>
</thead>
<tbody>
<tr>
<td>Jun−Aug 2011</td>
<td>Central Virginia to lower Bay of Fundy</td>
<td>1,595</td>
<td>0.33</td>
</tr>
<tr>
<td>Jun−Aug 2011</td>
<td>Central Florida to Central Virginia</td>
<td>23</td>
<td>0.76</td>
</tr>
<tr>
<td>Jun−Aug 2011</td>
<td>Central Florida to lower Bay of Fundy (COMBINED)</td>
<td>1,618</td>
<td>0.33</td>
</tr>
<tr>
<td>Jun−Sep 2016</td>
<td>Florida to lower Bay of Fundy</td>
<td>3,006</td>
<td>0.40</td>
</tr>
<tr>
<td>Aug−Sep 2016</td>
<td>Bay of Fundy/Scotian Shelf</td>
<td>2,235</td>
<td>0.413</td>
</tr>
<tr>
<td>Aug−Sep 2016</td>
<td>Newfoundland/Labrador</td>
<td>2,177</td>
<td>0.465</td>
</tr>
<tr>
<td>Jun−Sep 2016</td>
<td>Central Virginia to Newfoundland/Labrador (COMBINED)</td>
<td>7,418</td>
<td>0.25</td>
</tr>
</tbody>
</table>

Minimum Population Estimate

110
The minimum population estimate is the lower limit of the two-tailed 60% confidence interval of the log-normally distributed best abundance estimate. This is equivalent to the 20th percentile of the log-normal distribution as specified by Wade and Angliss (1997). The best estimate of abundance for fin whales is $7,418 \pm 802$ (CV=0.2524). The minimum population estimate for the western North Atlantic fin whale is $6,029 \pm 5,573$ (Table 2).

**Current Population Trend**

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

**CURRENT AND MAXIMUM NET PRODUCTIVITY RATES**

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

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

**POTENTIAL BIOLOGICAL REMOVAL**

Potential Biological Removal (PBR) is the product of minimum population size, one-half the maximum productivity rate, and a recovery factor (MMPA Sec. 3. 16 U.S.C. 1362; Wade and Angliss 1997). The minimum population size is $6,029 \pm 5,573$. 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 (Rmax), Recovery Factor (Fr.) and PBR.**

<table>
<thead>
<tr>
<th>Nbest</th>
<th>CV</th>
<th>Nmin</th>
<th>Fr</th>
<th>Rmax</th>
<th>PBR</th>
</tr>
</thead>
<tbody>
<tr>
<td>6,802</td>
<td>0.24</td>
<td>5,573</td>
<td>0.1</td>
<td>0.04</td>
<td>11</td>
</tr>
</tbody>
</table>

**ANNUAL HUMAN- CAUSED MORTALITY AND SERIOUS INJURY**

The total annual estimated average human-caused mortality and serious injury for the western North Atlantic fin whale for the period 2014–2018 is presented in Table 3. For the period 2013 through 2017, the minimum annual rate of human-caused mortality and serious injury to fin whales was 2.35 per year. This value includes incidental fishery interaction records, 1.55 (0 U.S./0.95 unknown but first reported in U.S. waters/0.6 Canadian waters); and records of vessel collisions, 0.8 (all U.S.) (Table 2a; Henry et al. 2020). Annual rates calculated from detected mortalities should not be considered an unbiased representation of human-caused mortality, but they represent a definitive lower bound. Detections are haphazard and not the result of a designed sampling scheme. As such they represent a minimum estimate of human-caused mortality which is almost certainly biased low. The size of this bias is uncertain.

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

<table>
<thead>
<tr>
<th>Years</th>
<th>Source</th>
<th>Annual Avg</th>
</tr>
</thead>
<tbody>
<tr>
<td>2014–2018</td>
<td>incidental fishery interactions</td>
<td>1.55</td>
</tr>
<tr>
<td>2014–2018</td>
<td>vessel collisions</td>
<td>0.80</td>
</tr>
<tr>
<td>TOTAL</td>
<td></td>
<td>2.35</td>
</tr>
</tbody>
</table>
Fishery-Related Serious Injury and Mortality

United States

U.S. fishery interaction records for large whales come through two main sources—dedicated fishery observer data and opportunistic reports collected in the Greater Atlantic Regional Fisheries Office/NMFS entanglement/stranding database. No confirmed fishery-related mortalities or serious injuries of fin whales have been reported in the NMFS Sea Sampling bycatch database (fishery observers) during this reporting period. A review of the records of stranded, floating, or injured fin whales for the reporting period 2013 through 2017 on file at NMFS found no records with substantial evidence of fishery interactions causing mortality in U.S. waters in the Greater Atlantic Regional Fisheries Office/NMFS entanglement/stranding database with substantial evidence of fishery interactions causing injury or mortality are presented in Table 4 (Table 2a; Henry et al. 2020). Serious injury determinations from fishery interaction records yielded a value of 4.75 over five years, for an annual average of 0.95 (Table 2a; Henry et al. 2020). These records are not statistically quantifiable in the same way as the observer fishery records, and they almost surely undercount entanglements for the stock.

Canada

The audited Greater Atlantic Regional Fisheries Office/NMFS entanglement/stranding database also contains records of fin whales first reported in Canadian waters or attributed to Canada, of which the confirmed mortalities and serious injuries from the last five years current reporting period are reported in Table 2b4. Three records with substantial evidence of fishery interactions causing mortality or serious injury were reported for the 2013–2017 period, resulting in a 5-year annual average of 0.6 animals.
Table 2a4. Confirmed human-caused mortality and serious injury records of fin whales (*Balaenoptera physalus*) first reported in U.S. waters or attributed to U.S. where the cause was assigned as either an entanglement (EN) or a vessel strike (VS): 2013–2018

<table>
<thead>
<tr>
<th>Date</th>
<th>Injury Determination</th>
<th>ID</th>
<th>Location</th>
<th>Assigned Cause</th>
<th>Value against PBR</th>
<th>Country</th>
<th>Gear Type</th>
<th>Description</th>
</tr>
</thead>
<tbody>
<tr>
<td>13-Jan-13</td>
<td>Mortality</td>
<td>-</td>
<td>East Hampton, NJ</td>
<td>VS</td>
<td>1</td>
<td>US</td>
<td>-</td>
<td>Fracturing of left cranium with associated hematoma</td>
</tr>
<tr>
<td>12-Apr-14</td>
<td>Mortality</td>
<td>-</td>
<td>Port Elizabeth, NJ</td>
<td>VS</td>
<td>1</td>
<td>US</td>
<td>-</td>
<td>Fresh carcass on bow of vessel. Large external abrasions w/ associated hemorrhage and skeletal fractures along right side.</td>
</tr>
<tr>
<td>5/13/14</td>
<td>Mortality</td>
<td>-</td>
<td>Rocky Harbour, NL</td>
<td>EN</td>
<td>1</td>
<td>CN PT</td>
<td>-</td>
<td>Fresh carcass hogs-tied in gear.</td>
</tr>
<tr>
<td>23-Jun-14</td>
<td>Prorated Injury</td>
<td>-</td>
<td>off Chatham, MA</td>
<td>EN</td>
<td>0.75</td>
<td>XU NR</td>
<td>-</td>
<td>Free-swimming, trailing 200ft of line. Attachment point(s) unknown. No resights.</td>
</tr>
<tr>
<td>20-Aug-14</td>
<td>Prorated Injury</td>
<td>-</td>
<td>off Provinctown, MA</td>
<td>EN</td>
<td>0.75</td>
<td>XU NR</td>
<td>-</td>
<td>Free-swimming, trailing buoy &amp; 200ft of line aft of flukes. Attachment point(s) unknown. No resights.</td>
</tr>
<tr>
<td>05-Oct-14</td>
<td>Mortality</td>
<td>-</td>
<td>off Manasquan, NJ</td>
<td>VS</td>
<td>1</td>
<td>US</td>
<td>-</td>
<td>Large area of hemorrhage along dorsal, ventral, and</td>
</tr>
<tr>
<td>Date&lt;sup&gt;b&lt;/sup&gt;</td>
<td>Injury Determination Fate</td>
<td>ID</td>
<td>Location&lt;sup&gt;b&lt;/sup&gt;</td>
<td>Assigned Cause</td>
<td>Value against PBR&lt;sup&gt;c&lt;/sup&gt;</td>
<td>Country&lt;sup&gt;d&lt;/sup&gt;</td>
<td>Gear Type&lt;sup&gt;e&lt;/sup&gt;</td>
<td>Description</td>
</tr>
<tr>
<td>-----------------</td>
<td>--------------------------</td>
<td>----</td>
<td>----------------------</td>
<td>----------------</td>
<td>----------------</td>
<td>----------------</td>
<td>----------------</td>
<td>----------------</td>
</tr>
<tr>
<td>06-Jun-15</td>
<td>Serious Injury</td>
<td>-</td>
<td>off Bar Harbor, ME</td>
<td>EN</td>
<td>1</td>
<td>XU</td>
<td>NR</td>
<td>right lateral surfaces consistent with blunt force trauma.</td>
</tr>
<tr>
<td>06-Jul-16</td>
<td>Prorated Injury</td>
<td>-</td>
<td>off Truro, MA</td>
<td>EN</td>
<td>0.75</td>
<td>XU</td>
<td>NR</td>
<td>Free-swimming with 2 buoys and 80 ft of line trailing from fluke. Line cutting deeply into right fluke blade. Emaciated. No resights.</td>
</tr>
<tr>
<td>08-Jul-16</td>
<td>Prorated Injury</td>
<td>-</td>
<td>off Virginia Beach, VA</td>
<td>EN</td>
<td>0.75</td>
<td>XU</td>
<td>H/MF</td>
<td>Free-swimming with lures in tow along left flipper area. Attachment point(s) and configuration unknown. No resights.</td>
</tr>
<tr>
<td>14-Dec-16</td>
<td>Prorated Injury</td>
<td>-</td>
<td>off Provincetown, MA</td>
<td>EN</td>
<td>0.75</td>
<td>XU</td>
<td>NR</td>
<td>Free-swimming with buoy trailing 6-8ft aft of flukes. Attachment point(s) and configuration unknown. No resights.</td>
</tr>
<tr>
<td>Date</td>
<td>Injuries Determination Fate</td>
<td>ID</td>
<td>Location</td>
<td>Assigned Cause</td>
<td>Value against PBR</td>
<td>Country</td>
<td>Gear Type</td>
<td>Description</td>
</tr>
<tr>
<td>------------</td>
<td>----------------------------</td>
<td>----</td>
<td>----------------</td>
<td>----------------</td>
<td>-------------------</td>
<td>---------</td>
<td>-----------</td>
<td>---------------------------------------------------------------------------------------------------------------------------------------------</td>
</tr>
<tr>
<td>05/30/17</td>
<td>Mortality</td>
<td></td>
<td>Port Newark, NJ</td>
<td>VS</td>
<td>1</td>
<td>US</td>
<td>-</td>
<td>Fresh carcass on bow of 656 ft vessel. Speed at strike unknown. Fisher found fresh carcass when hauling gear. Entangled at 78m depth, 51m from trap. Full configuration unknown, but unlikely to have drifted post-mortem in to gear.</td>
</tr>
<tr>
<td>8/25/17</td>
<td>Mortality</td>
<td></td>
<td>off Miscou Island, QC</td>
<td>EN</td>
<td>1</td>
<td>CN</td>
<td>PT</td>
<td>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.</td>
</tr>
<tr>
<td>6/22/2018</td>
<td>Mortality</td>
<td></td>
<td>16.5 nm E of Gaspe, QC</td>
<td>EN</td>
<td>1</td>
<td>CN</td>
<td>NP</td>
<td>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.</td>
</tr>
<tr>
<td>Date</td>
<td>Injury Determination</td>
<td>ID</td>
<td>Location</td>
<td>Assigned Cause</td>
<td>Value against PBR</td>
<td>Country</td>
<td>Gear Type</td>
<td>Description</td>
</tr>
<tr>
<td>------------</td>
<td>----------------------</td>
<td>-----------</td>
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<td>----------------</td>
<td>-------------------</td>
<td>---------</td>
<td>-----------</td>
<td>---------------------------------------------------------------------------------------------------------------------------------------------</td>
</tr>
</tbody>
</table>

**Assigned Cause** 5-Year mean (US/XU)

| Vessel strike | 0.8 (0.8/0.0) |
| Entanglement  | 0.95 (0/0.95/0.0) |

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

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

<table>
<thead>
<tr>
<th>Date</th>
<th>Injury Determination</th>
<th>ID</th>
<th>Location</th>
<th>Assigned Cause</th>
<th>Value against PBR</th>
<th>Country</th>
<th>Gear Type</th>
<th>Description</th>
</tr>
</thead>
<tbody>
<tr>
<td>6/6/13</td>
<td>Serious Injury</td>
<td>Capitaine Crochet</td>
<td>St. Lawrence Marine Park, Quebec</td>
<td>EN 1 CN PT</td>
<td>Pot resting on upper jaw w/ bridle lines embedding in mouth; health decline; emaciation</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>5/13/14</td>
<td>Mortality</td>
<td>-</td>
<td>Rocky Harbour, NL</td>
<td>EN 1 CN PT</td>
<td>Fresh carcass hauled in gear.</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>8/25/17</td>
<td>Mortality</td>
<td></td>
<td>off Miscou Island, QC</td>
<td>EN 1 CN PT</td>
<td>Fisher found fresh carcass when hauling gear. Entangled at 78m depth, 51m from trap. Full configuration unknown, but unlikely to have drifted post-mortem in to gear.</td>
<td></td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

### Assigned Cause

<table>
<thead>
<tr>
<th>Event</th>
<th>Value mean (CN/XC)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Vessel strike</td>
<td>0</td>
</tr>
<tr>
<td>Entanglement</td>
<td>0.6 (0.6/0.0)</td>
</tr>
</tbody>
</table>

---

**Other Mortality**

Death or injury as a result of vessel collision has significant anthropogenic impact on this stock (Schleimer et al. 2019). Known vessel strike cases are reported in Table 4. After reviewing NMFS records for 2013 through 2017, 4 were found that had sufficient information to confirm the cause of death as collisions with vessels (Table 2a; Henry et al. 2020). These records constitute an annual rate of serious injury or mortality of 0.8 fin whales from vessel collisions in U.S. waters.

### HABITAT ISSUES

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

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

**STATUS OF STOCK**

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

**REFERENCES CITED**


Garrison, L.P. 2016. Abundance of marine mammals in waters of the U.S. East Coast during summer 2011. PRBD Contribution #PRBD-2016-08, Southeast Fisheries Science Center, Protected Resources and Biodiversity Division, 75 Virginia Beach Dr., Miami, FL. 33140. PRBD Contribution #PRBD-2016-08, 21 pp.

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


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

**STOCK DEFINITION AND GEOGRAPHIC RANGE**

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

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

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

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

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

**POPULATION SIZE**

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

**Earlier abundance estimates**

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

**Recent surveys and abundance estimates**

An abundance estimate of 357 (CV=0.52) sei whales was generated from a shipboard and aerial survey conducted during June–August 2011 (Palka 2012). The aerial portion that contributed to the abundance estimate covered 5,212 km of tracklines that were over waters from north of New Jersey from the coastline to the 100-m depth contour, through the U.S. and Canadian Gulf of Maine and up to and including the lower Bay of Fundy. The shipboard portion covered 3,107 km of tracklines that were in waters offshore of Virginia to Massachusetts (waters that were deeper than the 100-m depth contour out to beyond the outer limit of the U.S. EEZ). Both sighting platforms used a double-platform data collection procedure, which allows estimation of abundance corrected for perception bias of the detected species (Laake and Borchers 2004). Estimation of the abundance was based on the independent observer approach assuming point independence (Laake and Borchers 2004) and calculated using the multiple covariate distance sampling (MCDS) option in the computer program Distance (version 6.0, release 2, Thomas et al. 2009). The abundance estimates of sei whales include a percentage of the estimate of animals identified as fin/sei whales (the two species being sometimes hard to distinguish). The percentage used is the ratio of positively identified sei whales to the total of positively identified fin whales and positively identified sei whales; the CV of the abundance estimate includes the variance of the estimated fraction. Although this is the best estimate available for this stock, it should be noted that the abundance survey from which it was derived excluded waters off the Scotian Shelf, an area encompassing a large portion of the stated range of the stock.
An estimate of 6,292 (CV=1.015) was the springtime (March–May) average abundance estimate generated from spatially- and temporally-explicit density models derived from visual two-team abundance survey data collected between 2010 and 2013 (Table 1; Palka et al. 2017). This estimate is for waters between Halifax, Nova Scotia and Florida, where the highest densities of animals were predicted to be on the Scotia shelf outside of U.S. waters. Over 25,000 km of shipboard and over 99,000 km of aerial visual line-transect survey data collected in all seasons in Atlantic waters from Florida to Nova Scotia during 2010–2014 were divided into 10x10 km² spatial grid cells and 8-day temporal time periods. Mark-recapture covariate Distance sampling was used to estimate abundance in each spatial-temporal cell which was corrected for perception bias. These density estimates and spatially- and temporally-explicit static and dynamic environmental data were used in Generalized Additive Models (GAMs) to develop spatially- and temporally-explicit animal density-habitat statistical models. These estimates were also corrected by platform- and species-specific availability bias correction factors that were based on dive time patterns.

An abundance estimate of 28 (CV=0.55) sei whales was generated from a summer shipboard and aerial survey conducted during 27 June–28 September 2016 (Table 1; Palka 2020) within a region covering 425,192 km². The estimate is only for waters along the continental shelf break from New Jersey to south of Nova Scotia. The aerial portion included 11,782 km of tracklines that were over waters north of New Jersey from the coastline to the 100-m depth contour, throughout U.S. waters. The shipboard portion included 4,351 km of tracklines that were in waters offshore of central Virginia to Massachusetts (waters that were deeper than the 100-m depth contour out to beyond the outer limit of the EEZ). Both sighting platforms used a two-team data collection procedure, which allows estimation of abundance to correct for perception bias of the detected species (Laake and Borchers 2004). The estimates were also corrected for availability bias.

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

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

<table>
<thead>
<tr>
<th>Month/Year</th>
<th>Area</th>
<th>N_best</th>
<th>CV</th>
</tr>
</thead>
<tbody>
<tr>
<td>Jun–Aug 2011</td>
<td>Central Virginia to lower Bay of Fundy</td>
<td>357</td>
<td>0.52</td>
</tr>
<tr>
<td>Apr–Jun 1999–2013</td>
<td>Maine to Florida in U.S. waters only</td>
<td>627</td>
<td>0.14</td>
</tr>
<tr>
<td>Jul–Sep 1995–2013</td>
<td>Gulf of St Lawrence entrance to Florida</td>
<td>717</td>
<td>0.30</td>
</tr>
<tr>
<td>Jun–Aug 2016</td>
<td>Continental shelf break waters from New Jersey to south of Nova Scotia</td>
<td>28</td>
<td>0.55</td>
</tr>
</tbody>
</table>

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

Current Population Trend

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

CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

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

POTENTIAL BIOLOGICAL REMOVAL

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

Table 2: Best and minimum abundance estimates for Nova Scotia sei whales (Balaenoptera borealis borealis) with Maximum Productivity Rate (Rmax), Recovery Factor (Fr.) and PBR.

<table>
<thead>
<tr>
<th>Nbest</th>
<th>CV</th>
<th>Nmin</th>
<th>Fr</th>
<th>Rmax</th>
<th>PBR</th>
</tr>
</thead>
<tbody>
<tr>
<td>6,292</td>
<td>1.02</td>
<td>3,098</td>
<td>0.1</td>
<td>0.04</td>
<td>6.2</td>
</tr>
</tbody>
</table>

ANNUAL HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

The most recent 5-year average human-caused mortality and serious injury rates are summarized in Table 3. For the period 2013 through 2017, the minimum annual rate of human-caused mortality and serious injury to sei whales was 1.0. This value includes incidental fishery interaction records, 0.2, and records of vessel collisions, 0.8 (Table 2; Henry et al. 2020). Annual rates calculated from detected mortalities should not be considered unbiased estimates of human-caused mortality, but they represent definitive lower bounds. Detections are haphazard, incomplete, and not the result of a designed sampling scheme. As such they represent a minimum estimate of human-caused mortality which is almost certainly biased low.

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

<table>
<thead>
<tr>
<th>Years</th>
<th>Source</th>
<th>Annual Avg.</th>
</tr>
</thead>
<tbody>
<tr>
<td>2014−2018</td>
<td>incidental fishery interactions</td>
<td>0.40</td>
</tr>
<tr>
<td>2014−2018</td>
<td>vessel collisions</td>
<td>0.80</td>
</tr>
<tr>
<td>TOTAL</td>
<td></td>
<td>1.20</td>
</tr>
</tbody>
</table>
Fishery-Related Serious Injury and Mortality

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

Table 2. Confirmed human-caused mortality and serious injury records of sei whales (*Balaenoptera borealis borealis*) where the cause was assigned as either an entanglement (EN) or a vessel strike (VS): 2013–2018

<table>
<thead>
<tr>
<th>Date</th>
<th>Injury Determination</th>
<th>ID</th>
<th>Location</th>
<th>Assigned Cause</th>
<th>Value against PBR</th>
<th>Country</th>
<th>Gear Type</th>
<th>Description</th>
</tr>
</thead>
<tbody>
<tr>
<td>5/7/2014</td>
<td>Mortality</td>
<td></td>
<td>Delaware River, PA</td>
<td>VS</td>
<td>1</td>
<td>US</td>
<td>-</td>
<td>Fresh carcass on bow of vessel.</td>
</tr>
<tr>
<td>07/25/2016</td>
<td>Mortality</td>
<td></td>
<td>Hudson River, Newark, NJ</td>
<td>VS</td>
<td>1</td>
<td>US</td>
<td>-</td>
<td>Fresh carcass on bow of ship (&gt;65 ft). Speed at strike unknown.</td>
</tr>
<tr>
<td>05/11/2017</td>
<td>Serious Injury</td>
<td></td>
<td>Cape Lookout Bight, NC</td>
<td>EN</td>
<td>1</td>
<td>XU</td>
<td>-</td>
<td>Free-swimming, emaciated, and carrying a large mass of heavily fouled gear consisting of line &amp; buoys crossing over back. Full configuration unknown, but evidence of</td>
</tr>
</tbody>
</table>
For the period 2013 through 2017 files at NMFS included four records with substantial evidence of vessel collision causing serious injury or mortality, which resulted in an annual rate of serious injury and mortality of 0.8 sei whales from vessel collisions.

**HABITAT ISSUES**

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

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

**STATUS OF STOCK**

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

REFERENCES CITED


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

**STOCK DEFINITION AND GEOGRAPHIC RANGE**

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

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

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

**POPULATION SIZE**

The best available current abundance estimate for common minke whales in the Canadian East Coast stock is the sum of the 2016 NEFSC and Department of Fisheries and Oceans Canada (DFO) surveys: 24,202 (CV=0.30). Because the survey areas did not overlap, the estimates from the two surveys were added together and the CVs pooled using a
delta method to produce a species abundance estimate for the stock area. This is assumed to be the majority of the Canadian East Coast stock. The 2016 estimate is larger than those from 2011 because the 2016 estimate is derived from a survey area extending from Newfoundland to Florida, which is about 1,300,000 km² larger than the 2011 survey area. In addition, some of the 2016 survey estimates in U.S. waters were corrected for availability bias (due to diving behavior), whereas the 2011 estimates were not corrected.

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

**Earlier estimates**

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

**Recent surveys and abundance estimates**

An abundance estimate of 2,591 (CV=0.81) common minke whales was generated from a shipboard and aerial survey conducted during June–August 2011 (Palka 2012). The aerial portion that contributed to the abundance estimate covered 5,313 km of tracklines that were over waters north of New Jersey from the coastline to the 100-m depth contour through the U.S. and Canadian Gulf of Maine, and up to and including the lower Bay of Fundy. The shipboard portion covered 2,107 km of tracklines that were in waters offshore of central Virginia to Massachusetts (waters that were deeper than the 100-m depth contour out to beyond the U.S. EEZ). Both sighting platforms used a double-platform data collection procedure, which allows estimation of abundance corrected for perception bias of the visually detected species (Laake and Borchers, 2004). Estimation of the abundance was based on the independent-observer approach assuming point independence (Laake and Borchers, 2004) and calculated using the multiple-covariate distance sampling option in the computer program Distance (version 6.0, release 2, Thomas et al. 2009).

An abundance estimate of 5,036 (CV=0.68) minke whales was generated from a shipboard and aerial survey conducted during 27 June–28 September 2016 (Palka 2020) in a region covering 425,192 km². The aerial portion included 11,782 km of tracklines that were over waters north of New Jersey from the coastline to the 100-m depth contour, throughout the U.S. waters. The shipboard portion consisted of 4,351 km of tracklines that were in waters offshore of central Virginia to Massachusetts (waters that were deeper than the 100-m depth contour out to beyond the U.S. EEZ). Both sighting platforms used a two-team data collection procedure, which allows estimation of abundance to correct for perception bias of the detected species (Laake and Borchers, 2004). The estimates were also corrected for availability bias.

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

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

<table>
<thead>
<tr>
<th>Month/Year</th>
<th>Area</th>
<th>N_best</th>
<th>CV</th>
</tr>
</thead>
<tbody>
<tr>
<td>Jul–Aug 2011</td>
<td>Central Virginia to lower Bay of Fundy</td>
<td>2,591</td>
<td>0.81</td>
</tr>
</tbody>
</table>

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Minimum Population Estimate

The minimum population estimate is the lower limit of the two-tailed 60% confidence interval of the log-normally distributed best abundance estimate. This is equivalent to the 20th percentile of the log-normal distribution as specified by Wade and Angliss (1997). The best estimate of abundance for the Canadian East Coast stock of common minke whales is 24,202–21,968 animals (CV=0.30). The minimum population estimate is 18,902–17,022 animals.

Current Population Trend

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

CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

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

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

POTENTIAL BIOLOGICAL REMOVAL

Potential Biological Removal (PBR) is the product of minimum population size, one-half the maximum productivity rate, and a recovery factor (MMPA Sec. 3. 16 U.S.C. 1362; Wade and Angliss 1997). The minimum population size is 18,902–17,022. The maximum productivity rate is 0.04, the default value for cetaceans. The recovery factor is 0.5, the default value for stocks of unknown status relative to OSP and with the CV of the average mortality estimate less than 0.3 (Wade and Angliss 1997). PBR for the Canadian East Coast common minke whale is 439–170 (Table 2).

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

<table>
<thead>
<tr>
<th>Month/Year</th>
<th>Area</th>
<th>Nbest</th>
<th>CV</th>
<th>Nmin</th>
<th>Fr</th>
<th>Rmax</th>
<th>PBR</th>
</tr>
</thead>
<tbody>
<tr>
<td>Jun–Sep 2016</td>
<td>Central Virginia to lower Bay of Fundy</td>
<td>5,036.2,80</td>
<td>0.68</td>
<td>2,80</td>
<td></td>
<td>0.81</td>
<td></td>
</tr>
<tr>
<td>Aug–Sep 2016</td>
<td>Gulf of St. Lawrence/Bay of Fundy/Scotian Shelf</td>
<td>6,158</td>
<td>0.40</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Aug–Sep 2016</td>
<td>Newfoundland/Labrador</td>
<td>13,008</td>
<td>0.46</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Jun–Sep 2016</td>
<td>Central Virginia to Labrador – COMBINED</td>
<td>24,202.21,968</td>
<td>0.30</td>
<td>17,022</td>
<td>0.5</td>
<td></td>
<td></td>
</tr>
</tbody>
</table>
ANNUAL HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

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

Data to estimate the mortality and serious injury of common minke whales come from the Northeast Fisheries Science Center Observer Program, the At-Sea Monitor Program, and from records of strandings and entanglements in U.S. and Canadian waters. For the purposes of this report, mortalities and serious injuries from reports of strandings and entanglements considered to be confirmed human-caused mortalities or serious injuries are shown in Table 2-4 while those recorded by the Observer or At-Sea Monitor Programs are shown in Table 3-5. Summary statistics are shown in Table 3.

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

<table>
<thead>
<tr>
<th>Years</th>
<th>Source</th>
<th>Annual Avg.</th>
</tr>
</thead>
<tbody>
<tr>
<td>2014−2018</td>
<td>incidental fishery interactions non-observed</td>
<td>8.95</td>
</tr>
<tr>
<td>2014−2018</td>
<td>U.S. fisheries using observer data</td>
<td>0.2</td>
</tr>
<tr>
<td>2014−2018</td>
<td>vessel collisions</td>
<td>1.20</td>
</tr>
<tr>
<td>2014−2018</td>
<td>other human interaction</td>
<td>0.2</td>
</tr>
<tr>
<td>TOTAL</td>
<td></td>
<td>10.55</td>
</tr>
</tbody>
</table>

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

Fishery-Information-Related Serious Injury and Mortality

Detailed fishery information is reported in Appendix III.

Earlier Interactions

See Appendix V for information on historical takes.

U.S. United States

U.S. fishery interaction records for large whales come through 2 main sources – dedicated fishery observer data and opportunistic reports collected in the Greater Atlantic Regional Fisheries Office/NMFS entanglement/stranding database. One confirmed fishery-related mortalities or serious injuries of minke whales has been reported in the NMFS Sea Sampling bycatch database (fishery observers) during this reporting period (Table 4). RA review of the records of stranded, floating, or injured minke whales for the reporting period 2014 through 2018 on file at NMFS found records in the audited Greater Atlantic Regional Fisheries Office/NMFS entanglement/stranding database with substantial evidence of fishery interactions causing injury or mortality (presented in Table 5; Henry et al. in review). These records are not statistically quantifiable in the same way as the observer fishery records, and they almost surely undercount entanglements for the stock.

Mid-Atlantic Gillnet
In December 2016 one minke whale mortality was observed in mid-Atlantic gillnet gear. A mortality estimate was not expanded to the entire fishery because the observed mortality was such a rare event. See Table 4 for bycatch estimates and observed mortality and serious injury for the current 5-year period, and Appendix V for historical bycatch information. Annual average estimated minke whale mortality and serious injury from the mid-Atlantic sink gillnet fishery during 2013 to 2017 was 0.2. This value was not expanded like other observed bycaught species (see Orphanides 2020) due to the low sample size.

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

<table>
<thead>
<tr>
<th>Fishery</th>
<th>Years</th>
<th>Data Typeᵃ</th>
<th>Observer Coverageᵇ</th>
<th>Observed Serious Injuryᶜ</th>
<th>Observed Mortality</th>
<th>Estimated Serious Injuryᶜ</th>
<th>Est. Mort.</th>
<th>Est. Combined Mortality</th>
<th>Est. CVs</th>
<th>Mean Combined Annual Mortality</th>
<th>CV of Mean</th>
</tr>
</thead>
<tbody>
<tr>
<td>Mid-Atl. Gillnet</td>
<td>2014</td>
<td>Obs. Data</td>
<td>0.05</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0.00</td>
</tr>
<tr>
<td></td>
<td>2015</td>
<td>Obs. Data</td>
<td>0.06</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0.00</td>
</tr>
<tr>
<td></td>
<td>2016</td>
<td>Weighout</td>
<td>0.08</td>
<td>0</td>
<td>1</td>
<td>0</td>
<td>1</td>
<td>1</td>
<td>0</td>
<td>2</td>
<td>0.00</td>
</tr>
<tr>
<td></td>
<td>2017</td>
<td>Weighout</td>
<td>0.09</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0.00</td>
</tr>
<tr>
<td></td>
<td>2018</td>
<td>Weighout</td>
<td>0.09</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0.00</td>
</tr>
<tr>
<td>TOTAL</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td>0.2</td>
<td>0.00</td>
</tr>
</tbody>
</table>

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

Other Fisheries

Confirmed mortalities and serious injuries of common minke whales in the last five years as recorded in the audited Greater Atlantic Regional Office/NMFS entanglement/stranding database are reported in Table 25. Data recorded during 2013 to 2017, as determined from stranding and entanglement records confirmed to be of U.S. origin or first sighted in U.S. waters, yielded a minimum detected average annual mortality and serious injury of 3.95 common minke whales per year in U.S. fisheries (Table 2a). One of the serious injury entanglement cases reported in Table 2a 5 was a non-fishery interaction (strapping) and so 0.2 was subtracted from the total entanglement 5-year average of 4.15. Most cases in which gear was recovered and identified involved gillnet or pot/trap gear.

CANADA

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

Other Fisheries

Mortalities and serious injuries that were likely a result of an interaction with an unknown Canadian fishery are detailed in Table 2b5. During 2013 to 2017, as determined from stranding and entanglement records confirmed to be of Canadian origin or first sighted in Canadian waters, the minimum detected average annual mortality and serious injury was 2.65 minke whales per year in Canadian fisheries (Table 2b; prorated value).
### Table 2a5. Confirmed human-caused mortality and serious injury records of common minke whales (Balaenoptera acutorostrata acutorostrata) first reported in U.S. waters or attributed to U.S.: 2013–2017

<table>
<thead>
<tr>
<th>Date</th>
<th>Injury determination</th>
<th>ID</th>
<th>Location</th>
<th>Assigned Cause</th>
<th>Value against PBR</th>
<th>Country</th>
<th>Gear Type</th>
<th>Description</th>
</tr>
</thead>
<tbody>
<tr>
<td>7/23/2013</td>
<td>Prorated Injury</td>
<td>-</td>
<td>off Newport, RI</td>
<td>EN</td>
<td>0.75</td>
<td>XU</td>
<td>NR</td>
<td>Full configuration unknown</td>
</tr>
<tr>
<td>8/17/2013</td>
<td>Serious Injury</td>
<td>-</td>
<td>off Newburyport, MA</td>
<td>EN</td>
<td>1</td>
<td>XU</td>
<td>NR</td>
<td>Constricting rostrum wrap cutting into upper lip</td>
</tr>
<tr>
<td>10/04/2013</td>
<td>Prorated Injury</td>
<td>-</td>
<td>off Seal Harbor, ME</td>
<td>EN</td>
<td>0.75</td>
<td>US</td>
<td>NR</td>
<td>Anchored, partially disentangled, final configuration unknown</td>
</tr>
<tr>
<td>7/2/2014</td>
<td>Mortality</td>
<td>-</td>
<td>Northumberland Strait, NB</td>
<td>EN</td>
<td>1</td>
<td>CN</td>
<td>NR</td>
<td>Carcass with constricting gear around lower jaw. Large open injury at attachment point on the left side.</td>
</tr>
<tr>
<td>7/10/2014</td>
<td>Prorated Injury</td>
<td>-</td>
<td>10 nm SE of Southport, ME of Bristol, ME</td>
<td>EN</td>
<td>0.75</td>
<td>XU</td>
<td>NR</td>
<td>Free-swimming, trailing 2 buoys. Attachment point(s) unknown</td>
</tr>
<tr>
<td>7/12/2014</td>
<td>Serious Injury</td>
<td>-</td>
<td>10 nm S of Southampton, NY South Shinnecock Inlet, NY</td>
<td>EN</td>
<td>1</td>
<td>XU</td>
<td>NR</td>
<td>Free-swimming with yellow plastic strapping cutting into top and sides of rostrum. No trailing gear.</td>
</tr>
<tr>
<td>7/17/2014</td>
<td>Mortality</td>
<td>-</td>
<td>South Addison, ME</td>
<td>EN</td>
<td>1</td>
<td>XU</td>
<td>NP</td>
<td>Fresh carcass with line impression across ventral surface &amp; evidence of constricting gear around peduncle and fluke insertion. Brusing evident at fluke injuries. No gear present.</td>
</tr>
<tr>
<td>7/29/2014</td>
<td>Mortality</td>
<td>-</td>
<td>5 nm E of Herring Cove, NS</td>
<td>VS</td>
<td>1</td>
<td>CN</td>
<td>:</td>
<td>Live animal w/ tongue completely ballooned out, forcing its jaws 90 degrees apart. Found dead at same location the next day. Carcass recovered with two traps &amp; constricting line around the peduncle. Necropsy found indication of blunt trauma to right jaw. Animal</td>
</tr>
<tr>
<td>Date</td>
<td>Injury determination</td>
<td>ID</td>
<td>Location</td>
<td>Assigned Cause</td>
<td>Value against PBR</td>
<td>Country</td>
<td>Gear Type</td>
<td>Description</td>
</tr>
<tr>
<td>------------</td>
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<td>----------------</td>
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<td>-----------</td>
<td>----------------------------------------------------------------------------------------------------------------------------------------------</td>
</tr>
<tr>
<td>12/24/2014</td>
<td>Mortality</td>
<td></td>
<td>Dam Neck, VA</td>
<td>VS</td>
<td>1</td>
<td>US</td>
<td>-</td>
<td>Anchored in gear was subsequently struck by a vessel (primary cause of death).</td>
</tr>
<tr>
<td>03/26/2015</td>
<td>Serious Injury</td>
<td></td>
<td>off Cape Canaveral, FL</td>
<td>EN</td>
<td>1</td>
<td>XU</td>
<td>NR</td>
<td>Fresh carcass with broken ribs &amp; fractured vertebrae w/ extensive hemorrhage &amp; edema.</td>
</tr>
<tr>
<td>04/16/2015</td>
<td>Mortality</td>
<td></td>
<td>Lockes Island, Shelburne, NS</td>
<td>EN</td>
<td>1</td>
<td>CN</td>
<td>NP</td>
<td>Evidence of constricting rostrum wrap, but unable to determine if gear still present. Emaciated.</td>
</tr>
<tr>
<td>05/09/2015</td>
<td>Mortality</td>
<td></td>
<td>Duck, NC</td>
<td>EN</td>
<td>1</td>
<td>XU</td>
<td>GU</td>
<td>Live stranded and euthanized. Embedded gear cutting into bone of mandible. Emaciated.</td>
</tr>
<tr>
<td>06/06/2015</td>
<td>Mortality</td>
<td></td>
<td>Coney Island, NY</td>
<td>VS</td>
<td>1</td>
<td>US</td>
<td>-</td>
<td>Fresh carcass with deep lacerations to throat area and head missing. Large area of bruising on dorsal surface.</td>
</tr>
<tr>
<td>06/14/2015</td>
<td>Prorated Injury</td>
<td></td>
<td>off Chatham, MA</td>
<td>EN</td>
<td>.75</td>
<td>XU</td>
<td>NR</td>
<td>Free-swimming with acorn buoy trailing 20-30 ft. Attachment point(s) and configuration unknown.</td>
</tr>
<tr>
<td>06/23/2015</td>
<td>Prorated Injury</td>
<td></td>
<td>off Ingonish, NS</td>
<td>EN</td>
<td>.75</td>
<td>CN</td>
<td>PT</td>
<td>Entangled in traps and buoys. Partially disentangled by fisherman. Original and final configuration unknown.</td>
</tr>
<tr>
<td>07/07/2015</td>
<td>Mortality</td>
<td></td>
<td>off Funk Island, NL</td>
<td>EN</td>
<td>1</td>
<td>CN</td>
<td>PT</td>
<td>Found at 340m depth in between two pots. Gear through mouth and wrapped around peduncle.</td>
</tr>
<tr>
<td>08/18/2015</td>
<td>Mortality</td>
<td></td>
<td>Roseville, PEI</td>
<td>EN</td>
<td>1</td>
<td>CN</td>
<td>NP</td>
<td>Evidence of constricting body, peduncle, and fluke wraps. No gear present. No necropsy but robust body condition supports.</td>
</tr>
<tr>
<td>Date</td>
<td>Injury determination</td>
<td>ID</td>
<td>Location</td>
<td>Assigned Cause</td>
<td>Value against PBR</td>
<td>Country</td>
<td>Gear Type</td>
<td>Description</td>
</tr>
<tr>
<td>--------------</td>
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<td>----------------</td>
<td>-------------------</td>
<td>---------</td>
<td>-----------</td>
<td>--------------------------------------------------------------------------------</td>
</tr>
<tr>
<td>09/01/2015</td>
<td>Mortality</td>
<td>-</td>
<td>Gloucester, MA</td>
<td>EN</td>
<td>1</td>
<td>US</td>
<td>NP</td>
<td>Evidence of entanglement as COD. Evidence of extensive, constricting gear with associated hemorrhaging. No gear present.</td>
</tr>
<tr>
<td>09/21/2015</td>
<td>Mortality</td>
<td>-</td>
<td>Cape Wolfe, Burton, PEI</td>
<td>EN</td>
<td>1</td>
<td>CN</td>
<td>NP</td>
<td>Evidence of constricting body wraps. No gear present. No necropsy but experts state peracute underwater entrapment most parsimonious.</td>
</tr>
<tr>
<td>12/06/2015</td>
<td>Mortality</td>
<td>-</td>
<td>off Port Joli, NS</td>
<td>EN</td>
<td>1</td>
<td>CN</td>
<td>PT</td>
<td>Live animal anchored in gear. Carcass recovered 4 days later.</td>
</tr>
<tr>
<td>5/3/2016</td>
<td>Mortality</td>
<td>-</td>
<td>Biddeford, ME</td>
<td>EN</td>
<td>1</td>
<td>US</td>
<td>PT</td>
<td>Line through mouth with evidence of constriction across ventral pleats and at peduncle. Hemorrhaging associated with these lesions.</td>
</tr>
<tr>
<td>7/21/2016</td>
<td>Serious Injury</td>
<td>-</td>
<td>Digby, NS</td>
<td>EN</td>
<td>1</td>
<td>XC</td>
<td>GU</td>
<td>Free-swimming with netting deeply embedded in rostrum. Disentangled, but significant health decline.</td>
</tr>
<tr>
<td>8/15/2016</td>
<td>Mortality</td>
<td>-</td>
<td>off Seguin Island, ME</td>
<td>EN</td>
<td>1</td>
<td>US</td>
<td>NR</td>
<td>Line exiting mouth leading to weighted/anchored gear.</td>
</tr>
<tr>
<td>8/30/2016</td>
<td>Mortality</td>
<td>-</td>
<td>3.1 nm SW of Matinicus Island, M</td>
<td>EN</td>
<td>1</td>
<td>US</td>
<td>PT</td>
<td>Fresh carcass anchored in gear with evidence of constricting wraps at peduncle and fluke insertions.</td>
</tr>
<tr>
<td>11/2/2016</td>
<td>Prorated Injury</td>
<td>-</td>
<td>Bonne Bay, Gros Morne National Park, NL</td>
<td>EN</td>
<td>0.75</td>
<td>XC</td>
<td>NR</td>
<td>Free-swimming and towing gear. Attachment point(s) and configuration unknown. No resights post 06Nov2016.</td>
</tr>
<tr>
<td>4/27/2017</td>
<td>Mortality</td>
<td>-</td>
<td>Staten Island, NY</td>
<td>VS</td>
<td>1</td>
<td>US</td>
<td>-</td>
<td>Evidence of bruising on dorsal and right scapular region. Histopathology results support blunt trauma from vessel strike most parsimonious as COD.</td>
</tr>
<tr>
<td>Date</td>
<td>Injury determination</td>
<td>ID</td>
<td>Location</td>
<td>Assigned Cause</td>
<td>Value against PBR</td>
<td>Country</td>
<td>Gear Type</td>
<td>Description</td>
</tr>
<tr>
<td>--------------</td>
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<td>----------------</td>
<td>-------------------</td>
<td>---------</td>
<td>-----------</td>
<td>----------------------------------------------------------------------------</td>
</tr>
<tr>
<td>7/6/2017</td>
<td>Mortality</td>
<td>EN</td>
<td>Manomet Point, MA</td>
<td>EN</td>
<td>1</td>
<td>US</td>
<td>PT</td>
<td>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.</td>
</tr>
<tr>
<td>7/22/2017</td>
<td>Mortality</td>
<td>EN</td>
<td>Piscataqua River, NH</td>
<td>EN</td>
<td>1</td>
<td>US</td>
<td>NP</td>
<td>Evidence of multiple constricting wraps on lower jaw and ventral pleats with associated hemorrhaging. No gear present.</td>
</tr>
<tr>
<td>8/9/2017</td>
<td>Mortality</td>
<td>EN</td>
<td>off Plymouth, MA</td>
<td>EN</td>
<td>1</td>
<td>US</td>
<td>NP</td>
<td>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.</td>
</tr>
<tr>
<td>8/11/2017</td>
<td>Prorated Injury</td>
<td>EN</td>
<td>off York, ME</td>
<td>EN</td>
<td>0.75</td>
<td>US</td>
<td>NR</td>
<td>Partially disentangled from anchoring gear. Final configuration unknown.</td>
</tr>
<tr>
<td>8/12/2017</td>
<td>Mortality</td>
<td>EN</td>
<td>off Tremont, ME</td>
<td>EN</td>
<td>1</td>
<td>US</td>
<td>GU</td>
<td>Fresh carcass of a pregnant female in gear. Constricting wrap injuries with associated hemorrhaging on dorsal and ventral surfaces and flukes.</td>
</tr>
<tr>
<td>8/14/2017</td>
<td>Mortality</td>
<td>EN</td>
<td>Pt. Judith, RI</td>
<td>EN</td>
<td>1</td>
<td>US</td>
<td>NP</td>
<td>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.</td>
</tr>
<tr>
<td>8/17/2017</td>
<td>Mortality</td>
<td>EN</td>
<td>Rye, NH</td>
<td>EN</td>
<td>1</td>
<td>US</td>
<td>NR</td>
<td>Evidence of constricting wraps on fluke blades and peduncle. Documented with line in baleen. Not present at time of necropsy. Limited necropsy, but extent</td>
</tr>
<tr>
<td>Date</td>
<td>Injury determination</td>
<td>ID</td>
<td>Location</td>
<td>Assigned Cause</td>
<td>Value against PBR</td>
<td>Country</td>
<td>Gear Type</td>
<td>Description</td>
</tr>
<tr>
<td>------------</td>
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<td>----------------</td>
<td>-------------------</td>
<td>---------</td>
<td>-----------</td>
<td>-------------</td>
</tr>
<tr>
<td>8/28/2017</td>
<td>Mortality</td>
<td>-</td>
<td>off Portland, ME</td>
<td>EN</td>
<td>1</td>
<td>US</td>
<td>PT</td>
<td>Fresh carcass anchored in gear. Endline wrapped around mouth and laceration from constricting gear on peduncle. Mud on flippers and mouth.</td>
</tr>
<tr>
<td>8/30/2017</td>
<td>Mortality</td>
<td>-</td>
<td>off North Cape, PEI</td>
<td>EN</td>
<td>1</td>
<td>CN</td>
<td>NR</td>
<td>Fresh carcass in gear. Full configuration unclear, but complex enough to not have drifted into post-mortem.</td>
</tr>
<tr>
<td>9/4/2017</td>
<td>Mortality</td>
<td>-</td>
<td>St. Carroll's, NL</td>
<td>EN</td>
<td>1</td>
<td>CN</td>
<td>NE</td>
<td>Alive in herring net. Found dead the next day. Fisher pulled carcass ashore and removed the net.</td>
</tr>
<tr>
<td>09/06/2017</td>
<td>Mortality</td>
<td>-</td>
<td>Newport, RI</td>
<td>VS</td>
<td>1</td>
<td>US</td>
<td>-</td>
<td>Hemorrhaging at left pectoral, left body, and aft of blowholes. Histopathology results support blunt trauma from vessel strike as COD.</td>
</tr>
<tr>
<td>9/17/2017</td>
<td>Mortality</td>
<td>-</td>
<td>Henry Island, NS</td>
<td>EN</td>
<td>1</td>
<td>CN</td>
<td>NR</td>
<td>Fresh carcass with gear in mouth and around flukes. Evidence of constricting wrap on dorsum. No necropsy, but configuration complex enough that unlikely to have drifted into gear post-mortem.</td>
</tr>
<tr>
<td>9/26/2017</td>
<td>Prorated Injury</td>
<td>-</td>
<td>off Richbuctou, NB</td>
<td>EN</td>
<td>0.75</td>
<td>CN</td>
<td>NR</td>
<td>Animal initially anchored in gear then not sighted. Unable to confirm if gear free, partially entangled, or drowned.</td>
</tr>
<tr>
<td>9/27/2017</td>
<td>Mortality</td>
<td>-</td>
<td>5.7nm NE of Richbuctou, NB</td>
<td>EN</td>
<td>1</td>
<td>CN</td>
<td>NP</td>
<td>No gear present. Fresh carcass with evidence of constricting wraps.</td>
</tr>
<tr>
<td>10/10/2017</td>
<td>Mortality</td>
<td>-</td>
<td>off Rockland, ME</td>
<td>EN</td>
<td>1</td>
<td>US</td>
<td>PT</td>
<td>Entangled in 2 different sets of gear. Constricting wrap around lower jaw. Found at depth</td>
</tr>
<tr>
<td>Date</td>
<td>Injury determination</td>
<td>ID</td>
<td>Location</td>
<td>Assigned Cause</td>
<td>Value against PBR</td>
<td>Country</td>
<td>Gear Type</td>
<td>Description</td>
</tr>
<tr>
<td>------------</td>
<td>----------------------</td>
<td>----</td>
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<td>----------------</td>
<td>-------------------</td>
<td>---------</td>
<td>-----------</td>
<td>-----------------------------------------------------------------------------</td>
</tr>
<tr>
<td>09-Feb-18</td>
<td>Mortality</td>
<td></td>
<td>Tiverton, Long Island, NS</td>
<td>EN</td>
<td>1</td>
<td>XC</td>
<td>NP</td>
<td>when fisher hauled gear. No gear present. Evidence of constricting body, flipper, and peduncle wraps. No necropsy conducted, but COD from entanglement most parsimonious.</td>
</tr>
<tr>
<td>25-May-18</td>
<td>Mortality</td>
<td></td>
<td>Digby, NS</td>
<td>VS</td>
<td>1</td>
<td>CN</td>
<td></td>
<td>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.</td>
</tr>
<tr>
<td>11-Jun-18</td>
<td>Mortality</td>
<td></td>
<td>Cape Dauphin, NS</td>
<td>EN</td>
<td>1</td>
<td>CN</td>
<td>PT</td>
<td>Fresh, pregnant carcass anchored in gear.</td>
</tr>
<tr>
<td>19-Jun-18</td>
<td>Mortality</td>
<td></td>
<td>East Point, PHI</td>
<td>EN</td>
<td>1</td>
<td>CN</td>
<td>NP</td>
<td>No gear present. Fresh, pregnant carcass with evidence of extensive constricting body and peduncle wraps with associated hemorrhaging.</td>
</tr>
<tr>
<td>22-Jun-18</td>
<td>Prorated Injury</td>
<td></td>
<td>4.5 nm N of Grand Manan, NB</td>
<td>EN</td>
<td>0.75</td>
<td>XC</td>
<td>NR</td>
<td>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. Evidence of extensive constricting body and mouth wraps with associated hemorrhaging. Deep lacerations at fluke insertion from constricting gear. COD - peracute underwater entrapment.</td>
</tr>
<tr>
<td>24-Jun-18</td>
<td>Mortality</td>
<td></td>
<td>Wellfleet, MA</td>
<td>EN</td>
<td>1</td>
<td>XU</td>
<td>GN</td>
<td>Evidence of extensive constricting body and mouth wraps with associated hemorrhaging. Deep lacerations at fluke insertion from constricting gear. COD - peracute underwater entrapment.</td>
</tr>
<tr>
<td>07-Jul-18</td>
<td>Mortality</td>
<td></td>
<td>1.6 nm E of Newcastle, NH</td>
<td>EN</td>
<td>1</td>
<td>US</td>
<td>PT</td>
<td>Anchored in gear with line through mouth and wrapping around body. Associated bruising at right corner of mouth. COD - peracute underwater entrapment.</td>
</tr>
<tr>
<td>Date\textsuperscript{b}</td>
<td>Injury determination</td>
<td>ID</td>
<td>Location\textsuperscript{b}</td>
<td>Assigned Cause\textsuperscript{f}</td>
<td>Value against PBR\textsuperscript{c}</td>
<td>Country\textsuperscript{d}</td>
<td>Gear Type\textsuperscript{e}</td>
<td>Description</td>
</tr>
<tr>
<td>------------------</td>
<td>----------------------</td>
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<td>-------------------------------</td>
<td>-------------------------</td>
<td>----------------</td>
<td>----------------</td>
<td>-----------------</td>
</tr>
<tr>
<td>22-Jul-18</td>
<td>Mortality</td>
<td>:</td>
<td>Cape Neddick, ME</td>
<td>EN</td>
<td>1</td>
<td>XU</td>
<td>NP</td>
<td>No necropsy, but evidence of constricting wrap at fluke insertion with associated hemorrhaging. Histopathology confirms pre-mortem human-induced trauma.</td>
</tr>
<tr>
<td>28-Jul-18</td>
<td>Mortality</td>
<td>:</td>
<td>Biddeford, ME</td>
<td>EN</td>
<td>1</td>
<td>XU</td>
<td>NP</td>
<td>No gear present, but evidence of constricting gear with associated bruising at mouth, around body and peduncle.</td>
</tr>
<tr>
<td>06-Aug-18</td>
<td>Prorated Injury</td>
<td>:</td>
<td>Fish Cove Point, NL</td>
<td>EN</td>
<td>0.75</td>
<td>CN</td>
<td>NF</td>
<td>Free-swimming towing net with float attached. Member of public cut off float. Original and final configuration unknown.</td>
</tr>
<tr>
<td>29-Aug-18</td>
<td>Prorated Injury</td>
<td>:</td>
<td>7.5 nm SE of Chatham, MA</td>
<td>EN</td>
<td>0.75</td>
<td>XU</td>
<td>NR</td>
<td>Free-swimming with buoy near flukes, full configuration unknown.</td>
</tr>
<tr>
<td>03-Sep-18</td>
<td>Mortality</td>
<td>:</td>
<td>Nancy Head, Campobello, NB</td>
<td>EN</td>
<td>1</td>
<td>CN</td>
<td>WE, SE</td>
<td>Live animal entrapped. Failed attempt by fisher to remove animal with seine. Animal became entangled in seine and drowned.</td>
</tr>
<tr>
<td>16-Sep-18</td>
<td>Mortality</td>
<td>:</td>
<td>0.7 nm SSE of Rye, NH</td>
<td>EN</td>
<td>1</td>
<td>US</td>
<td>PT</td>
<td>Fresh carcass anchored in gear. Constricting body, jaw, peduncle, and fluke wraps with associated hemorrhaging.</td>
</tr>
<tr>
<td>07-Nov-18</td>
<td>Mortality</td>
<td>:</td>
<td>Tangier Island, VA</td>
<td>EN</td>
<td>1</td>
<td>XU</td>
<td>NP</td>
<td>Constricting gear with associated hemorrhaging partly amputating tip of rostrum. Poor body condition. COD - chronic entanglement.</td>
</tr>
<tr>
<td>25-Dec-18</td>
<td>Mortality</td>
<td>:</td>
<td>Yarmouth Bar, NS</td>
<td>EN</td>
<td>1</td>
<td>XC</td>
<td>NP</td>
<td>No gear present. Evidence of constricting entanglement on head, ventral pleats, peduncle and flukes. No necropsy, but COD from entanglement most parsimonious.</td>
</tr>
<tr>
<td>Assigned Cause</td>
<td>5-Year mean (US/\text{CN}/XU/XC)</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
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<td>---------------------------------</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Vessel strike (US/XU)</td>
<td>0.81 (0.8/0.4/0.000)</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Entanglement (US/XU)</td>
<td>4.15 (2.53/2.85/4.45/2.05/0.9)</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

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

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

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

d. US=United States, XU=Unassigned 1st sight in U.S., CN=Canada, XC=Unassigned 1st sight in CNS.

e. H=hook, GN=gillnet, GU=gear unidentifiable, MF=monofilament, NP=none present, NR=none recovered/received, PT=pot/trap, WE=weir.

f. Assigned cause: EN=entanglement, VS=vessel strike, ET=entrapment (summed with entanglement).
<table>
<thead>
<tr>
<th>Date</th>
<th>Injury determination</th>
<th>ID</th>
<th>Location</th>
<th>Assigned Cause</th>
<th>Value against PBR</th>
<th>Country</th>
<th>Gear Type</th>
<th>Description</th>
</tr>
</thead>
<tbody>
<tr>
<td>8/31/2013</td>
<td>Mortality</td>
<td>-</td>
<td>Miminegash, PEI</td>
<td>EN</td>
<td>1</td>
<td>CN</td>
<td>NP</td>
<td>Fresh carcass with evidence of extensive, constricting gear.</td>
</tr>
<tr>
<td>7/2/2014</td>
<td>Mortality</td>
<td>-</td>
<td>Northumberland Strait, NB</td>
<td>EN</td>
<td>1</td>
<td>CN</td>
<td>NR</td>
<td>Carcass with constricting gear around lower jaw. Large open injury at attachment point on the left side.</td>
</tr>
<tr>
<td>7/29/2014</td>
<td>Mortality</td>
<td>-</td>
<td>5 nm E of Herring Cove, NS</td>
<td>VS</td>
<td>1</td>
<td>CN</td>
<td>-</td>
<td>Live animal with tongue completely ballooned out, forcing its jaws 90 degrees apart. Found dead at same location the next day. Carcass recovered with two traps &amp; constricting line around the peduncle. Necropsy found indication of blunt trauma to right jaw. Animal anchored in gear was subsequently struck by a vessel (primary cause of death).</td>
</tr>
<tr>
<td>04/16/2015</td>
<td>Mortality</td>
<td>-</td>
<td>Locke Island, Shelburne, NS</td>
<td>EN</td>
<td>1</td>
<td>CN</td>
<td>NP</td>
<td>Fresh carcass with evidence of constricting wraps. No gear present. Robust, pregnant, fish in stomach and intestines. No other</td>
</tr>
<tr>
<td>Date</td>
<td>Injury determination</td>
<td>ID</td>
<td>Location</td>
<td>Assigned Cause</td>
<td>Value against PBR</td>
<td>Country</td>
<td>Gear Type</td>
<td>Description</td>
</tr>
<tr>
<td>------------</td>
<td>----------------------</td>
<td>----</td>
<td>----------</td>
<td>----------------</td>
<td>------------------</td>
<td>---------</td>
<td>-----------</td>
<td>-----------------------------------------------------------------------------</td>
</tr>
<tr>
<td>06/23/2015</td>
<td>Injured</td>
<td></td>
<td>off Ingonish, NS</td>
<td>EN</td>
<td>.75</td>
<td>CN</td>
<td>PE</td>
<td>Entangled in traps and buoys. Partially disentangled by fisherman. Original and final configuration unknown.</td>
</tr>
<tr>
<td>07/07/2015</td>
<td>Mortality</td>
<td></td>
<td>off Funk Island, NL</td>
<td>EN</td>
<td>1</td>
<td>CN</td>
<td>PE</td>
<td>Found at 340m depth in between two pots. Gear through mouth and wrapped around peduncle.</td>
</tr>
<tr>
<td>08/18/2015</td>
<td>Mortality</td>
<td></td>
<td>Roseville, PEI</td>
<td>EN</td>
<td>1</td>
<td>CN</td>
<td>NP</td>
<td>Evidence of constricting body, peduncle, and fluke wraps. No gear present. No necropsy but robust body condition supports entanglement as COD.</td>
</tr>
<tr>
<td>09/21/2015</td>
<td>Mortality</td>
<td></td>
<td>Cape Wolfe, Burton, PEI</td>
<td>EN</td>
<td>1</td>
<td>CN</td>
<td>NP</td>
<td>Evidence of constricting body-wraps. No gear present. No necropsy but experts state post-mortem underwater entrapment most parsimonious.</td>
</tr>
<tr>
<td>12/06/2015</td>
<td>Mortality</td>
<td></td>
<td>off Port Joli, NS</td>
<td>EN</td>
<td>1</td>
<td>CN</td>
<td>PE</td>
<td>Live animal anchored in gear. Carcass recovered 4 days later.</td>
</tr>
<tr>
<td>05/03/2016</td>
<td>Mortality</td>
<td></td>
<td>Biddeford, ME</td>
<td>EN</td>
<td>1</td>
<td>US</td>
<td>PE</td>
<td>Carcass in gear. Live through mouth and evidence of constricting wraps on peduncle.</td>
</tr>
<tr>
<td>Date</td>
<td>Injury determination</td>
<td>ID</td>
<td>Location b</td>
<td>Assigned Cause f</td>
<td>Value against PBR a</td>
<td>Country d</td>
<td>Gear Type e</td>
<td>Description</td>
</tr>
<tr>
<td>-----------</td>
<td>----------------------</td>
<td>----</td>
<td>------------</td>
<td>------------------</td>
<td>---------------------</td>
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<td>-------------</td>
<td>-------------</td>
</tr>
<tr>
<td>7/21/2016</td>
<td>Serious Injury</td>
<td>-</td>
<td>Digby, NS</td>
<td>EN</td>
<td>1</td>
<td>XC</td>
<td>GU</td>
<td>Ventral pleats and peduncle with associated hemorrhaging.</td>
</tr>
<tr>
<td>8/15/2016</td>
<td>Mortality</td>
<td>-</td>
<td>off Seguin Island, ME</td>
<td>EN</td>
<td>1</td>
<td>US</td>
<td>NR</td>
<td>Line exiting mouth leading to weighted or anchored gear.</td>
</tr>
<tr>
<td>11/2/2016</td>
<td>Prorated Injury</td>
<td>-</td>
<td>Bonne Bay, Gros Morne National Park, NL</td>
<td>EN</td>
<td>0.75</td>
<td>XC</td>
<td>NR</td>
<td>Free swimming and towing gear. Attachment points(s) and configuration unknown. No resights post 06Nov2016.</td>
</tr>
<tr>
<td>8/30/2017</td>
<td>Mortality</td>
<td>-</td>
<td>off North Cape, PEI</td>
<td>EN</td>
<td>1</td>
<td>CN</td>
<td>NR</td>
<td>Fresh carcass in gear. Full configuration unclear, but complex enough to not have drifted into post-mortal.</td>
</tr>
<tr>
<td>9/4/2017</td>
<td>Mortality</td>
<td>-</td>
<td>St. Carroll’s, NL</td>
<td>EN</td>
<td>1</td>
<td>CN</td>
<td>NE</td>
<td>Alive in herring net. Found dead the next day. Fisher pulled carcass ashore.</td>
</tr>
<tr>
<td>Date</td>
<td>Injury determination</td>
<td>ID</td>
<td>Location</td>
<td>Assigned Cause</td>
<td>Value against PBR</td>
<td>Country</td>
<td>Gear Type</td>
<td>Description</td>
</tr>
<tr>
<td>------</td>
<td>----------------------</td>
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<td>----------------</td>
<td>-------------------</td>
<td>---------</td>
<td>-----------</td>
<td>-------------</td>
</tr>
<tr>
<td>9/17/2017</td>
<td>Mortality</td>
<td>-</td>
<td>Henry Island, NS</td>
<td>EN</td>
<td>1</td>
<td>CN</td>
<td>NR</td>
<td>Fresh carcass with gear in mouth and around flukes. Evidence of constricting wrap on dorsum. No necropsy, but configuration complex enough that unlikely to have drifted into gear post-mortem.</td>
</tr>
<tr>
<td>9/26/2017</td>
<td>Prorated Injury</td>
<td>-</td>
<td>off Richibucto, NB</td>
<td>EN</td>
<td>0.75</td>
<td>CN</td>
<td>NR</td>
<td>Animal initially anchored in gear, then not resighted. Unable to confirm if gear-free, partially entangled, or drowned.</td>
</tr>
</tbody>
</table>

**Assigned Cause**

| Vessel strike | 0.20 (0.20/0.00) |
| Entanglement | 2.65 (2.30/0.35) |

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**Notes:**

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

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

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

d. CN=Canada, XC=Unassigned 1st sight in CN

e. H=hook, GN=gillnet, GU=gear unidentifiable, MF=monofilament, NP=none present, NR=none recovered/received, PT=pot/trap, WE=weir.

f. Assigned cause: EN=entanglement, VS=vessel strike, ET=entrapment (summed with entanglement).
### Table 3. From observer program data, summary of the incidental mortality of the Canadian East Coast stock of minke whales (Balaenoptera acutorostrata) by commercial fishery including the years sampled, the type of data used, the annual observer coverage.

<table>
<thead>
<tr>
<th>Fishery</th>
<th>Years</th>
<th>Data Type</th>
<th>Observer Coverage</th>
<th>Observed Serious Injury</th>
<th>Observed Mortality</th>
<th>Estimated Serious Injury</th>
<th>Estimated Mortality</th>
<th>Combined Serious Injury</th>
<th>Estimated CVs</th>
<th>Mean Annual Combined Mortality</th>
</tr>
</thead>
<tbody>
<tr>
<td>Mid-Atlantic</td>
<td>2013</td>
<td>Obs. Data</td>
<td>0.03</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0.2 (na)</td>
</tr>
<tr>
<td>Mid-Atlantic</td>
<td>2014</td>
<td>Obs. Data</td>
<td>0.05</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0.2 (na)</td>
</tr>
<tr>
<td>Mid-Atlantic</td>
<td>2015</td>
<td>Data, Weighout</td>
<td>0.06</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0.2 (na)</td>
</tr>
<tr>
<td>Mid-Atlantic</td>
<td>2016</td>
<td>Data, Weighout</td>
<td>0.08</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>1</td>
<td>1</td>
<td>na</td>
<td>0.2 (na)</td>
</tr>
<tr>
<td>Mid-Atlantic</td>
<td>2017</td>
<td>Data, Weighout</td>
<td>0.09</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>na</td>
<td>0.2 (na)</td>
</tr>
</tbody>
</table>

**a.** Observer data (Obs. Data), used to measure bycatch rates, are collected within the Northeast Observer Program and At-sea Monitoring Program. NEFSC collects landings data (unallocated Dealer Data or Allocated Dealer Data) which are used as a measure of total landings. Mandatory Vessel Trip Reports (VTR) (Trip Logbook) are used to determine the spatial distribution of landings and fishing effort in the sink gillnet, bottom trawl and mid-water trawl fisheries. In addition, the Trip Logbooks are the primary source of the measure of total effort (tow duration) in the mid-water and bottom trawl fisheries.

**b.** Observer coverage for the U.S. mid-Atlantic coastal gillnet fisheries is based on tons of fish landed.

**c.** Serious injuries were evaluated since 2011 using new guidelines and include both at-sea monitor and traditional observer data (Josephson et al. 2019).

### Other Mortality

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

### United States

Common minke whales inhabit coastal waters during much of the year and are thus susceptible to collision with vessels. Vessel strike interactions in U.S. and Canadian waters are reported in Table 5. In 2014, a confirmed vessel strike resulted in a mortality off Dam Neck, Virginia. In 2015, a fresh carcass of a common minke whale was reported off Coney Island, New York with wounds consistent with vessel strike. In 2017 there are 2 records of minke whale mortalities as a result of vessel strikes. Thus, during 2013–2017, as determined from stranding and entanglement records, the minimum detected annual average was 0.8 common minke whales per year struck by vessels in U.S. waters or first seen in U.S. waters (Table 2a; Henry et al. 2020).

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

In January 2017, a minke whale Unusual Mortality Event (UME) was declared for the U.S. Atlantic coast due to elevated numbers of mortalities. From January 2017 to December 2018, 57 minke whales stranded between Maine and South Carolina. Preliminary findings in several of the whales have shown evidence of human interactions or infectious disease. This most recent UME is ongoing (https://www.fisheries.noaa.gov/national/marine-life-distress/2017-2020-minke-whale-unusual-mortality-event-along-atlantic-coast; accessed 31 July 2020). Anthropogenic mortalities and serious injuries that occurred in 2017 and 2018 as part of this UME are included in Table 5. An Unusual Mortality Event was established for minke whales in January 2017 due to elevated stranding along the Atlantic coast (https://www.fisheries.noaa.gov/national/marine-life-distress/2017-2018-minke-whale-unusual-mortality-event-along-atlantic-coast). Anthropogenic mortalities and serious injuries that occurred in 2017 are included in Tables 1a and 1b.

### Canada

The Nova Scotia Stranding Network documented whales and dolphins stranded on the coast of Nova Scotia between 1991 and 1996 (Hooker et al. 1997). Researchers with the Department of Fisheries and Oceans, Canada documented strandings on the beaches of Sable Island (Lucas and Hooker 2000). Starting in 1997, Common minke whales stranded on the coast of Nova Scotia were recorded by the Marine Animal Response Society (MARS) and the Nova Scotia Stranding Network (Tonya Wimmer/Andrew Reid, pers. comm.). The events that were determined to be human-caused serious injury or mortality are included in Table 2b.
The Whale Release and Strandings program reported the following common minke whale stranding mortalities in Newfoundland and Labrador (Ledwell and Huntington 2013, 2014, 2015, 2017, 2018) for the time period of this report: 0 in 2013, 1 in 2014, 2 in 2015, 0 in 2016 and 2 in 2017. Those that have been determined to be human-caused serious injury or mortality are included in Table 2b-5 (Ledwell and Huntington 2013, 2014, 2015, 2017, 2018).

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

HABITAT ISSUES

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

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

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

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

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

It is expected that the uncertainties described above will have little effect on the designation of the status of the entire stock. Even though the estimate of human-caused mortality and serious injury in this assessment (8 animals) is negatively biased due to using strandings and entanglement data as the primary source, it is well below the PBR calculated from the abundance estimate for the U.S. and Canadian portion of the Canadian East Coast common minke whale stock’s habitat (489).

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

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


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 (29ºW) (Doksaeter et al. 2008; Waring et al. 2008). Common dolphins have been noted to be associated with Gulf Stream features (CETAP 1982; Selzer and Payne 1988; Waring et al. 1992; Hamazaki 2002). The species is less common south of Cape Hatteras, although schools have been reported as far south as the Georgia/South Carolina border (32º N) (Jefferson et al. 2009). They exhibit seasonal movements, where they are found from Cape Hatteras northeast to Georges Bank (35° to 42°N) during mid-January to May (Hain et al. 1981; CETAP 1982; Payne et al. 1984), although some animals tagged and released after stranding in winters of 2010–2012 used habitat in the Gulf of Maine north to almost 44ºN (Sharp et al. 2016). Common dolphins move onto Georges Bank, Gulf of Maine, and the Scotian Shelf from mid-summer to autumn. Selzer and Payne (1988) reported very large aggregations (greater than 3,000 animals) on Georges Bank in autumn. Migration onto the Scotian Shelf and continental shelf off Newfoundland occurs during summer and autumn when water temperatures exceed 11ºC (Sergeant et al. 1970; Gowans and Whitehead 1995).

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

**POPULATION SIZE**

The current best abundance estimate for Western North Atlantic stock of common dolphins is $172,825^{172,947}$ (CV=0.21) which is the total of Canadian and U.S. surveys conducted in 2016 (Table 1). This estimate, derived from shipboard and aerial surveys, covers most of this stock’s known range. Because the survey areas did not overlap, the estimates from the three surveys were added together and the CVs pooled using a delta method to produce a species abundance estimate for the stock area. The 2016 estimate is larger than those from 2011 because the 2016 estimate is derived from a survey area extending from Newfoundland to Florida, which is about 1,300,000 km² larger than the 2011 survey area. In addition, some of the 2016 survey estimates in US waters were corrected for availability bias (due to diving behavior), whereas the 2011 estimates were not corrected (Table 1).

**Earlier estimates**

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

**Recent surveys and abundance estimates**

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

An abundance estimate of 2,993 (CV=0.87) common dolphins was generated from a shipboard survey conducted concurrently (June–August 2011) in waters between central Virginia and central Florida. This shipboard survey included shelf-break and inner continental slope waters deeper than the 50-m depth contour within the U.S. EEZ. The survey employed a double-platform visual team procedure searching with 25.150 “bigeye” binoculars. A total of 4,445 km of tracklines was surveyed. Estimation of the abundance was based on the independent-observer approach assuming point independence (Laake and Borchers 2004) and calculated using the MRDS option in the computer program Distance (version 6.0, release 2, Thomas et al. 2009) (Table 1).

Abundance estimates of 48,574$^{48,723}$ (CV=0.48) for the Newfoundland/Labrador portion and 43,124 (CV=0.28) for the Bay of Fundy/Scotian Shelf/Gulf of St. Lawrence portion of the stock area were generated from the Canadian Northwest Atlantic International Sightings Survey (NAISS) survey conducted in August–September 2016 (Table 1). This large-scale aerial survey covered Atlantic Canadian shelf and shelf break habitats from the northern tip of Labrador to the U.S border off southern Nova Scotia (Lawson and Gosselin 2018). Line-transect density and abundance analyses were completed using Distance 7.1 release 1 (Thomas et al. 2010).

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

<table>
<thead>
<tr>
<th>Month/Year</th>
<th>Area</th>
<th>Nbest</th>
<th>CV</th>
</tr>
</thead>
<tbody>
<tr>
<td>Jul–Aug 2011</td>
<td>Central Virginia to lower Bay of Fundy</td>
<td>67,191</td>
<td>0.29</td>
</tr>
<tr>
<td>Jun–Aug 2011</td>
<td>Central Florida to Central Virginia</td>
<td>2,903</td>
<td>0.87</td>
</tr>
<tr>
<td>Jun–Aug 2011</td>
<td>Central Florida to lower Bay of Fundy (COMBINED)</td>
<td>70,184</td>
<td>0.28</td>
</tr>
<tr>
<td>June–Sep 2016</td>
<td>Central Virginia to lower Bay of Fundy</td>
<td>80,227</td>
<td>0.31</td>
</tr>
<tr>
<td>June–Aug 2016</td>
<td>Florida to Central Virginia</td>
<td>900</td>
<td>0.57</td>
</tr>
<tr>
<td>June–Sep 2016</td>
<td>Newfoundland/Labrador</td>
<td>48,574</td>
<td>0.48</td>
</tr>
<tr>
<td>June–Sep 2016</td>
<td>Bay of Fundy/Scotian Shelf/Gulf of St. Lawrence</td>
<td>43,124</td>
<td>0.28</td>
</tr>
<tr>
<td>June–Sep 2016</td>
<td>Florida to Newfoundland/Labrador (COMBINED)</td>
<td>172,825</td>
<td>0.21</td>
</tr>
</tbody>
</table>

Minimum Population Estimate

The minimum population estimate is the lower limit of the two-tailed 60% confidence interval of the log-normally distributed best abundance estimate. This is equivalent to the 20th percentile of the log-normal distribution as specified by Wade and Angliss (1997). The best estimate of abundance for common dolphins is 172,825 animals (CV=0.21), derived from the 2016 aerial and shipboard surveys. The minimum population estimate for the western North Atlantic common dolphin is 145,091 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 long survey interval (see Appendix IV for a survey history of this stock). For example, the power to detect a precipitous decline in abundance (i.e., 50% decrease in 15 years) with estimates of low precision (e.g., CV > 0.30) remains below 80% (alpha = 0.30) unless surveys are conducted on an annual basis (Taylor et al. 2007). There is current work to standardize the strata-specific previous abundance estimates to consistently represent the same regions and include appropriate corrections for perception and availability bias. These standardized abundance estimates will be used in state-space trend models that incorporate environmental factors that could potentially influence the process and observational errors for each stratum.

CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

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

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

POTENTIAL BIOLOGICAL REMOVAL

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

Table 2. Best and minimum abundance estimates for the western North Atlantic common dolphin (Delphinus delphis delphis) with Maximum Productivity Rate (Rmax), Recovery Factor (Fr.), and PBR.

<table>
<thead>
<tr>
<th></th>
<th>Nbest</th>
<th>CV</th>
<th>Nmin</th>
<th>Fr</th>
<th>Rmax</th>
<th>PBR</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>172,974</td>
<td>0.21</td>
<td>145,216</td>
<td>0.5</td>
<td>0.04</td>
<td>1,452</td>
</tr>
</tbody>
</table>

ANNUAL HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

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

<table>
<thead>
<tr>
<th>Years</th>
<th>Source</th>
<th>Annual Avg.</th>
<th>CV</th>
</tr>
</thead>
<tbody>
<tr>
<td>2014-2018</td>
<td>U.S. fisheries using observer data</td>
<td>399</td>
<td>0.05</td>
</tr>
<tr>
<td>2014-2018</td>
<td>Research mortalities</td>
<td>0.2</td>
<td></td>
</tr>
<tr>
<td>TOTAL</td>
<td></td>
<td>399</td>
<td></td>
</tr>
</tbody>
</table>

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

Fishery information

Detailed fishery information is reported in Appendix III. Earlier Interactions

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

Northeast Sink Gillnet

Annual common dolphin mortalities were estimated using annual ratio-estimator methods (Hatch and Orphanides 2015–2016; Orphanides and Hatch 2017, Orphanides 2019, 2020, in press). See Table 2-4 for bycatch estimates and observed mortality and serious injury for the current 5-year period, and Appendix V for historical bycatch information.

Mid-Atlantic Gillnet

Common dolphins were taken in observed trips during most years. Annual common dolphin mortalities were estimated using annual ratio-estimator methods (Hatch and Orphanides 2015–2016; Orphanides and Hatch 2017, Orphanides 2019, 2020, in press). See Table 2-4 for bycatch estimates and observed mortality and serious injury for the current 5-year period, and Appendix V for historical bycatch information.

Northeast Bottom Trawl

This fishery is active in New England waters in all seasons. Annual common dolphin mortalities were estimated using annual stratified ratio-estimator methods (Lyssikatos et al. 2020, in press). See Table 2-4 for bycatch estimates and observed mortality and serious injury for the current 5-year period, and Appendix V for historical bycatch information.
Mid-Atlantic Bottom Trawl

Annual common dolphin mortalities were estimated using annual stratified ratio-estimator methods (Lyssikatos et al. 2020 in press). See Table 2-4 for bycatch estimates and observed mortality and serious injury for the current 5-year period, and Appendix V for historical bycatch information.

Pelagic Longline

Pelagic longline bycatch estimates of common dolphins for 2013-2014-2017-2018 were documented in Garrison and Stokes (2014, 2016, 2017, in press 2020, in review). There is a high likelihood that dolphins released alive with ingested gear or gear wrapped around appendages will not survive (Wells et al. 2008). See Table 2-4 for bycatch estimates and observed mortality and serious injury for the current 5-year period, and Appendix V for historical bycatch information.

Research Takes

In October 2016; the University of Rhode Island, Graduate School of Oceanography reported the incidental capture/drowning of a 206-cm female, common dolphin during a routine, weekly research trawl fishing trip in Narragansett Bay, Rhode Island. The incident was reported to Mystic Aquarium, Mystic, Connecticut; NOAA GARFO Office, Gloucester, Massachusetts; NOAA law enforcement; and NOAA Protected Species Branch, Woods Hole, Massachusetts. A complete necropsy was conducted at the Wood Hole Oceanographic Institution, Woods Hole, Massachusetts.
Table 2. Summary of the incidental serious injury and mortality of North Atlantic common dolphins (Delphinus delphis delphis) by commercial fishery including the years sampled, the type of data used, the annual observer coverage, the serious injuries and mortalities recorded by on-board observers, the estimated annual serious injury and mortality, the combined serious injury and mortality estimate, the estimated CV of the annual combined serious injury and mortality and the mean annual serious injury and mortality estimate (CV in parentheses).

<table>
<thead>
<tr>
<th>Fishery</th>
<th>Years</th>
<th>Data Type a</th>
<th>Observer Coverage b</th>
<th>Observed Serious Injury d</th>
<th>Observed Mortality</th>
<th>Estimated Serious Injury d</th>
<th>Estimated Mortality</th>
<th>Estimated Combined Mortality</th>
<th>Estimated CVs</th>
<th>Mean Combined Annual Mortality</th>
</tr>
</thead>
<tbody>
<tr>
<td>Northeast Sink Gillnet</td>
<td>2013</td>
<td>Obs. Data, Trip Logbook, Allocated Dealer Data</td>
<td>0.11</td>
<td>0</td>
<td>5</td>
<td>0</td>
<td>104</td>
<td>104</td>
<td>0.46</td>
<td>97.94 (0.19)</td>
</tr>
<tr>
<td></td>
<td>2014</td>
<td>Obs. Data</td>
<td>0.18</td>
<td>0</td>
<td>11</td>
<td>0</td>
<td>111</td>
<td>111</td>
<td>0.47</td>
<td></td>
</tr>
<tr>
<td></td>
<td>2015</td>
<td>Obs. Data, Allocated Dealer Data</td>
<td>0.14</td>
<td>0</td>
<td>3</td>
<td>0</td>
<td>55</td>
<td>55</td>
<td>0.54</td>
<td></td>
</tr>
<tr>
<td></td>
<td>2016</td>
<td>Obs. Data</td>
<td>0.10</td>
<td>0</td>
<td>8</td>
<td>0</td>
<td>80</td>
<td>80</td>
<td>0.38</td>
<td></td>
</tr>
<tr>
<td></td>
<td>2017</td>
<td>Obs. Data</td>
<td>0.12</td>
<td>0</td>
<td>20</td>
<td>0</td>
<td>133</td>
<td>133</td>
<td>0.28</td>
<td></td>
</tr>
<tr>
<td></td>
<td>2018</td>
<td>Obs. Data</td>
<td>0.11</td>
<td>0</td>
<td>10</td>
<td>0</td>
<td>93</td>
<td>93</td>
<td>0.45</td>
<td></td>
</tr>
<tr>
<td>Mid-Atlantic Gillnet</td>
<td>2013</td>
<td>Obs. Data, Weighout</td>
<td>0.03</td>
<td>0</td>
<td>2</td>
<td>0</td>
<td>62</td>
<td>62</td>
<td>0.67</td>
<td>18.17 (0.28)</td>
</tr>
<tr>
<td></td>
<td>2014</td>
<td>Obs. Data</td>
<td>0.05</td>
<td>0</td>
<td>1</td>
<td>0</td>
<td>17</td>
<td>17</td>
<td>0.86</td>
<td></td>
</tr>
<tr>
<td></td>
<td>2015</td>
<td>Obs. Data</td>
<td>0.06</td>
<td>0</td>
<td>3</td>
<td>0</td>
<td>30</td>
<td>30</td>
<td>0.55</td>
<td></td>
</tr>
<tr>
<td></td>
<td>2016</td>
<td>Obs. Data</td>
<td>0.08</td>
<td>0</td>
<td>1</td>
<td>0</td>
<td>7</td>
<td>7</td>
<td>0.97</td>
<td></td>
</tr>
<tr>
<td></td>
<td>2017</td>
<td>Obs. Data</td>
<td>0.09</td>
<td>1</td>
<td>1</td>
<td>11</td>
<td>11</td>
<td>22</td>
<td>0.71</td>
<td></td>
</tr>
<tr>
<td></td>
<td>2018</td>
<td>Obs. Data</td>
<td>0.09</td>
<td>0</td>
<td>1</td>
<td>0</td>
<td>8</td>
<td>8</td>
<td>0.91</td>
<td></td>
</tr>
<tr>
<td>Northeast Bottom Trawl c</td>
<td>2013</td>
<td>Obs. Data, Logbook</td>
<td>0.15</td>
<td>0</td>
<td>4</td>
<td>0</td>
<td>17</td>
<td>17</td>
<td>0.53</td>
<td>14.17 (0.28)</td>
</tr>
<tr>
<td></td>
<td>2014</td>
<td>Obs. Data</td>
<td>0.17</td>
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<td>-</td>
<td>-</td>
<td>-</td>
<td>418.399 (0.05)</td>
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</table>

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

b. Observer coverage is defined as the ratio of observed to total metric tons of fish landed for the gillnet fisheries and the ratio of observed to total trips for bottom trawl and Mid-Atlantic mid-water trawl (including pair trawl) fisheries. Beginning in May 2010 total observer coverage reported for bottom trawl and gillnet gear includes samples collected from the at-sea monitoring program in addition to traditional observer coverage through the Northeast Fisheries Observer Program (NEFOP).

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

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

Other Mortality

From 2013 to 2017, 608 common dolphins were reported stranded between Maine and Florida (Table 3; are...
reported in Table 5 (NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 23–20 October–November 2018). The total includes mass-stranded common dolphins in Massachusetts during 2013 (a total of 9 in 3 events), 2014 (a total of 14 in 4 events), 2015 (a total of 37 in 13 events), and 2016 (a total of 35 animals in 9 events), and 2017 (over 90 animals in 20 events), and 2018 (a total of 28 animals in 9 events). Mass strandings in Virginia in 2013 (a total of 6 in 2 events). Animals released or last sighted alive include 12 in 2013, 12 in 2014, 9 in 2015, 23 in 2016 and 70 in 2017 and 18 in 2018. In 2013, 10 cases were classified as human interaction, 4 of which were fishery interactions. In 2014, 5 cases were classified as human interaction, 1 of which was a fishery interaction. In 2015, 2 cases were classified as human interactions, both in Rhode Island. Seven cases in 2016 were coded as human interaction, 1 of which was a fishery interaction. Six cases in 2017 were coded as human interaction, 2 of which were classified as fishery interactions and 1 of which was classified as a boat collision. In 2018, 5 cases were coded as human interactions, 3 of which involved fishing gear. In an analysis of mortality causes of stranded marine mammals on Cape Cod and southeastern Massachusetts between 2000 and 2006, Bogomolni (2010) reported that 61% of stranded common dolphins were involved in mass-stranding events, and 37% of all the common dolphin stranding mortalities were disease-related.

The Marine Animal Response Society of Nova Scotia reported no common dolphins stranded in 2013, 3 in 2014, 2 in 2015, 5 in 2016, and 5 in 2017 and 5 in 2018 (Tonya Wimmer/Andrew Reid, pers. comm.).

Table 3. Common dolphin (Delphinus delphis delphis) reported strandings along the U.S. Atlantic coast, 2013–2018.

<table>
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<td>1</td>
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<td>3</td>
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<td>82</td>
<td>190</td>
<td>84</td>
<td>540499</td>
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</table>

a. Massachusetts mass strandings (2013: 4, 3, 2, 2014: 2, 2, 5, 2015: 2, 2, 2, 2, 2, 3, 3, 3, 4, 4, 2016: 8, 5, 4, 4, 2017: 2x5, 3x3, 4x4, 5x5, 7x2, 14x1). Two mass strandings in Virginia in April 2013 – a group of 4 and a group of 2.

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

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

**HABITAT ISSUES**

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

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

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

**STATUS OF STOCK**

Common dolphins are not listed as threatened or endangered under the Endangered Species Act, and the Western North Atlantic stock is not considered strategic under the Marine Mammal Protection Act. The 2013–2017 average annual human-related mortality does not exceed PBR. The total U.S. fishery-related mortality and serious injury for this stock is not less than 10% of the calculated PBR and, therefore, cannot be considered to be insignificant and approaching zero mortality and serious injury rate. The status of common dolphins, relative to OSP, in the U.S. Atlantic EEZ is unknown. Population trends for this species have not been investigated.

**REFERENCES CITED**


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


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

**STOCK DEFINITION AND GEOGRAPHIC RANGE**

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

This stock exhibits spatiotemporal overlap with multiple common bottlenose stocks in the wNA. The stock is best defined by its distribution during warm water months (best described by July and August) when it overlaps with the fewest stocks. During warm water months, this stock occupies coastal waters from the shoreline to approximately the 20-m isobath between Assateague, Virginia, and Long Island, New York (Figure 1) (Garrison *et al.* 2017b). The stock migrates in late summer and fall and, during cold water months (best described by January and February), occupies coastal waters from approximately Cape Lookout, North Carolina, to the North Carolina/Virginia border (Garrison *et al.* 2017b). Four common bottlenose dolphins tagged during 2003 and 2004 off the coast of New Jersey in late summer moved south to North Carolina and inhabited waters near and just south of Cape Hatteras during cold water months. These animals then returned to coastal waters of New Jersey in the following warm water months (Garrison *et al.* 2017b). Similarly, a dolphin tagged in late
The distribution of the Northern Migratory Coastal Stock overlaps in certain seasons with several other common
bottlenose dolphin stocks. Overlap with the Southern Migratory Coastal Stock in coastal waters of northern North
Carolina and Virginia is possible during spring and fall migratory periods, but the degree of overlap is unknown and
it may vary depending on annual water temperature (Garrison et al. 2016). When the stock has migrated in cold water
months to coastal waters from just north of Cape Hatteras, North Carolina, to just south of Cape Lookout, North
Carolina, it overlaps spatially with the Northern North Carolina Estuarine System (NNCES) Stock (Garrison et al.
2017b). Depending on the timing of the northward migration in the spring, it may overlap with the NNCES stock in
coastal waters (< 1 km from shore) as far north as Virginia Beach, Virginia, and the mouth of the Chesapeake Bay. It
may also overlap with the Southern North Carolina Estuarine System Stock (Garrison et al. 2017b) in nearshore coastal
waters south of Cape Hatteras in winter, although the degree of overlap with the latter stock is not well defined. This
stock may also overlap to some degree with the wNA Offshore Stock of common bottlenose dolphins. A combined
genetic and logistic regression analysis that incorporated depth, latitude, and distance from shore was used to model
the probability that a particular common bottlenose dolphin group seen in coastal waters was of the coastal morphotype
(Garrison et al. 2017a). North of Cape Hatteras during summer months, there is strong separation between the coastal
and offshore morphotype (Kenney 1990; Garrison et al. 2017a), and the coastal morphotype is nearly completely
absent in waters >20 m depth. South of Cape Hatteras, the regression analysis indicated that the coastal morphotype
occurs at lower densities over the continental shelf, in waters >20 m deep where it overlaps to some degree with the
offshore morphotype. For the purposes of defining stock boundaries and identifying bycaught dolphins, the offshore
boundary of the Northern Migratory Coastal Stock is defined as the 20-m isobath in summer north of Cape Hatteras and
the 200-m isobath in winter between Cape Hatteras and Cape Lookout.

**POPULATION SIZE**

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

**Background**

Estimating the abundance of the Northern Migratory Coastal Stock is complicated by the spatiotemporal overlap
the stock has with other coastal and estuarine common bottlenose dolphins as described above. Summer surveys are
best for estimating the abundance for the Northern Migratory Coastal Stock because it overlaps least with other coastal,
estuarine, and offshore stocks of common bottlenose dolphins during warm water months. Abundance for the Northern
Migratory Coastal Stock is estimated using summer sightings made in the 0–20 m depth stratum during summer aerial
surveys north of Assateague, Virginia (37.9°N) to Sandy Hook, New Jersey (40.3°N). The definition of the southern
summer boundary and inter-annual variation in stock distribution are significant unquantified sources of uncertainty.
Earlier abundance estimates (>8 years old)

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

Recent surveys and abundance estimates

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

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

The abundance estimates for the 2010, 2011, and 2016 summer aerial surveys were 14,314 (CV=0.74), 15,630 (CV=0.29) and 6,639 (CV=0.41), respectively (Table 1; Garrison et al. 2017a). The 2016 estimate was used as the best estimate of the current population size for the stock due to possible effects from the 2013–2015 unusual mortality event. Uncertainties in the abundance estimate arise primarily from annual, and unquantified, variation in stock distribution.

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

<table>
<thead>
<tr>
<th>Month</th>
<th>Year</th>
<th>Area Covered</th>
<th>Abundance Estimate (Nbest)</th>
<th>CV</th>
</tr>
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</tr>
</tbody>
</table>

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Minimum Population Estimate

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

Current Population Trend

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

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

CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

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

POTENTIAL BIOLOGICAL REMOVAL

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

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

<table>
<thead>
<tr>
<th>Nbest</th>
<th>CV</th>
<th>Nmin</th>
<th>Fr</th>
<th>Rmax</th>
<th>PBR</th>
</tr>
</thead>
<tbody>
<tr>
<td>6,639</td>
<td>0.41</td>
<td>4,759</td>
<td>0.5</td>
<td>0.04</td>
<td>48</td>
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</tbody>
</table>

ANNUAL HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

The total annual human-caused mortality and serious injury for the Northern Migratory Coastal Stock during 2011-2015 is unknown. The mean annual fishery-related mortality and serious injury for observed fisheries and strandings identified as fishery-related ranged between 12.2 and 21.5 (CV=0.32) and 13.2 (CV=0.22). No additional mortality and serious injury was documented from other human-caused sources (e.g., fishery research) and therefore, the minimum total mean annual human-caused mortality and serious injury for this stock during 2011-2015 ranged between 6.1 and 12.2 and 13.2 and 21.5 (Tables 2a, 2b and 2c). This range reflects several sources of uncertainty and is a minimum because 1) not all fisheries that could interact with this stock are observed, 2) stranding data are used as an indicator of fishery-related interactions and not all dead animals are detected and recovered by the stranding network (Peltier et al. 2012; Wells et al. 2015), 3) cause of death is not (or cannot be) routinely determined for stranded carcasses, 4) the estimate includes an actual count of verified human-caused deaths and serious injuries and should be considered a minimum (NMFS 2016), and 5) the spatiotemporal overlap between this stock and other common bottlenose dolphin stocks in North Carolina and Virginia introduces uncertainty in assignment of mortalities to stock. In the sections below, dolphin mortalities and serious injuries were ascribed to a stock or stocks by comparing the season and geographic location of the take/stranding to the stock boundaries and geographic range delimited for each stock (Lyssikatos and Garrison 2018).

Fishery Information

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

Note: Animals reported in the sections to follow were ascribed to a stock or stocks of origin following methods described in Maze-Foley et al. (2019). These include strandings, observed takes (through an observer program), fisherman self-reported takes (through the Marine Mammal Authorization Program), research takes, and
opportunistic at-sea observations. In one instance in this report, the stock identity of a stranded animal was supported via sighting history in the Mid-Atlantic Bottlenose Dolphin Catalog (Urian in review).

Mid-Atlantic Gillnet

The mid-Atlantic gillnet fishery operates along the coast from North Carolina through New York (2016 List of Fisheries) and overlaps with the Northern Migratory Coastal Stock throughout its range. North Carolina is the largest component of the mid-Atlantic gillnet fishery in terms of fishing effort and observed marine mammal takes (Palka and Rossman 2001; Lyssikatos and Garrison 2018). This fishery is currently observed by the Northeast Fisheries Observer Program and previously was observed by both the Northeast and Southeast Fisheries Observer Programs (through 2016). The Bottlenose Dolphin Take Reduction Plan (BDTRP) was implemented in May 2006 and resulted in changes to gillnet gear configurations and fishing practices (50 CFR 24776, April 26, 2006, available at https://www.federalregister.gov/documents/2006/04/26/06-3909/taking-of-marine-mammals-incidental-to-commercial-fishing-operations-bottlenose-dolphin-take http://www.nmfs.noaa.gov/pr/pdfs/fr/71-24776.pdf). In addition, two amendments to the BDTRP were implemented in 2008 and 2012 regarding gear restrictions for medium-mesh gillnets in North Carolina waters (https://www.fisheries.noaa.gov/national/marine-mammal-protection/bottlenose-dolphin-take-reduction-plan). Mortality estimates for the period 2002–2006 (immediately prior to implementation of the BDTRP) and 2007–2011 are available in the 2015 stock assessment report for the Northern Migratory Coastal Stock (Waring et al. 2016). The current report covers the most recent available five-year estimate (NMFS 2016) for 2011–2015.

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

During the most recent five-year time period, 2011–2015, the combined average Northeast (NEFOP) and Southeast (SEFOP) Fisheries Observer Program observer coverage (measured in trips) for this fishery was 2,625,16% in state waters (0–3 miles from shore) and 5,369,95% in federal waters (3–200 miles from shore) (Lyssikatos in review and Garrison 2018). During these trips, observers documented three entangled dolphins that may have been from the Northern Migratory Coastal Stock, all off of North Carolina, two off of New Jersey, and one off of Virginia, that may have been from the Northern Migratory Coastal Stock. In April 2018, the NEFOP observed one mortality in medium-mesh gillnet gear off the coast of Virginia that was ascribed solely to the Northern Migratory Coastal Stock. Also in October 2017 the NEFOP observed one mortality in medium-mesh gillnet gear off the coast of New Jersey ascribed solely to the Northern Migratory Coastal Stock. In February 2017, the NEFOP documented a dolphin entangled in small-mesh gillnet gear off the coast of North Carolina that was released alive, and it could not be determined if the animal was seriously injured. The animal was ascribed to the Northern Migratory Coastal and NNCES stocks. In August 2015, the NEFOP observed one mortality in medium-mesh gillnet gear off the coast of New Jersey that was ascribed solely to the Northern Migratory Coastal Stock. In January 2015, one mortality was observed by the NEFOP off Hatteras, North Carolina, entangled in a medium-mesh gillnet gear within 0.23 km of shore. This dolphin was ascribed to the Northern Migratory Coastal and NNCES stocks (this animal was also self-reported by the fisherman per the Marine Mammal Authorization Program). In February 2013, the NEFOP documented a dolphin entangled in small-mesh gillnet gear that was released alive without serious injury; it was
Historical stranding data have documented multiple cases of dead, stranded dolphins recovered with gillnet gear attached (Byrd et al. 2014; Waring et al. 2016; Lyssikatos and Garrison 2018). However, six mortalities and one live animal were documented entangled in gillnet gear, none were documented over the current five-year period, 2014–2018, that may have been from the Northern Migratory Coastal Stock (two of these mortalities were also documented by the Marine Mammal Authorization Program). The live animal was disentangled and released alive but it could not be determined whether the animal was seriously injured (Maze-Foley and Garrison 2020). From 2014 to 2019, six dead, stranded dolphins were recovered with markings indicative of interaction with gillnet gear (Read and Murray 2000), but no gear was attached to the carcasses and it is unknown whether the interactions with the gear contributed to the death of these animals. Four of the six cases were ascribed to the Northern Migratory Coastal Stock alone, and two were ascribed to the Northern and Southern Migratory Coastal stocks, and three were ascribed to the Northern Migratory Coastal and NNCES stocks.

Northeast Sink Gillnet

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

Chesapeake Bay Inshore Gillnet

During 2011–2015, there were two documented strandings of a common bottlenose dolphin entangled in inshore gillnet gear in Chesapeake Bay. In 2013, in Maryland, a dead dolphin was recovered entangled in large-mesh gillnet gear. In 2015, in Virginia, a stranded animal was found entangled in gillnet gear (this animal was also self-reported by the fisherman per the Marine Mammal Authorization Program). Both of these animals were ascribed to the Northern and Southern Migratory Coastal stocks, and both are included in the annual human-caused mortality and serious injury total for this stock (Table 2b) as well as in the stranding database and stranding totals presented in Table 4. There is no observer coverage of this fishery within Maryland waters of Chesapeake Bay; within Virginia waters of Chesapeake Bay, there is a low level of observer coverage (<1%). No estimate of bycatch mortality is available for this fishery, and the documented interaction in this commercial gear represents a minimum known count of interactions in the last five years. Six other dead, stranded common bottlenose dolphins were recovered within Chesapeake Bay with markings indicative of interaction with gillnet gear (Read and Murray 2000), but no gear was attached to the carcasses and it is unknown whether the interactions with the gear contributed to the death of these animals. Five of these animals were ascribed solely to the Northern Migratory Coastal Stock (one case supported by Urian in review), and one was ascribed to the Northern and Southern Migratory Coastal and NNCES stocks.

Virginia Pound Net

During 2011–2015, there were no documented mortalities or serious injuries involving pound net gear in Virginia. One common bottlenose dolphin stranding (mortality) that was ascribed to the Northern and Southern Migratory Coastal Stocks was found entangled in pound net gear in Virginia (in 2011; Northeast Regional Marine Mammal Stranding Network; NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 6 June 2016), and was included in the annual human-caused mortality and serious injury total for this stock (Table 2b). However, an additional four dead, stranded dolphins were recovered with twisted twine markings indicative of interactions with pound net gear, but it is unknown whether the interactions with the gear contributed to the death of these animals and these cases are not included in the annual human-caused
mortality and serious injury total for this stock. Of these four, one was ascribed solely to the Northern Migratory Coastal Stock, and the remaining three were ascribed to the Northern and Southern Migratory Coastal and NNCE stocks. All of the strandings discussed here occurred inside estuarine waters near the mouth of the Chesapeake Bay in March 2016, April, May, September, or October (Northeast Regional Marine Mammal Stranding Network; NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 13 August 2019). Because there is no systematic observer program for the Virginia pound net fishery, no estimate of bycatch mortality is available, and the documented interaction in this commercial gear represents a minimum known count of interactions in the last five years. An amendment to the BDTRP was implemented in 2015 requiring gear restrictions for VA pound nets in estuarine and coastal state waters of Virginia to reduce bycatch.

Mid-Atlantic Menhaden Purse Seine

During the years 2011–2014–2015, there were no documented mortalities or serious injuries in mid-Atlantic menhaden purse seine gear of common bottlenose dolphins that could be ascribed to the Northern Migratory Coastal Stock. The mid-Atlantic menhaden purse seine fishery historically reported an annual incidental take of one to five common bottlenose dolphins (NMFS 1991, pp. 5–73). There has been very limited federal observer coverage since 2008. No observer coverage was allocated to this fishery during 2014–2015. Because there is no systematic observer program for this fishery, no estimate of bycatch mortality is available.

Atlantic Blue Crab Trap/Pot

During the years 2011–2014–2015, stranding data identified four cases of common bottlenose dolphins entangled in trap/pot gear that could be ascribed to the Northern Migratory Coastal Stock. Two cases were serious injuries, and for the remaining two cases, it could not be determined whether the animals were seriously injured in commercial blue crab trap/pot gear of common bottlenose dolphins. One serious injury occurred in 2017 in unidentified trap/pot gear and was ascribed solely to the Northern Migratory Coastal Stock. The second serious injury occurred in 2017 in commercial blue crab trap/pot gear and was ascribed to the Northern and Southern Migratory Coastal and NNCE stocks. The serious injuries are included in the annual human-caused mortality and serious injury total for this stock (Table 3b). Also in 2017, there was one entanglement in unidentified trap/pot gear ascribed to the Northern and Southern Migratory Coastal stocks. In 2018, there was an entanglement in unidentified trap/pot gear that was ascribed to the Northern and Southern Migratory Coastal and NNCE stocks. For both of these cases, it could not be determined whether the animals were seriously injured. All of the cases were included in the stranding database and in the stranding totals presented in Table 4 (Northeast Regional Marine Mammal Stranding Network; National Marine Mammal Health and Stranding Response Database unpublished data, accessed 13 August 2019). Details regarding the serious injury determinations can be found in Maze-Foley and Garrison (2020). Because there is no systematic observer program, it is not possible to estimate the total number of interactions or mortalities associated with crab traps/pots. However, stranding data indicate that interactions with trap/pot gear occur at some unknown level in North Carolina (Byrd et al. 2014) and other regions of the southeast U.S. (Noke and Odell 2002; Burdett and McFee 2004).

Mid-Atlantic Haul/Beach Seine

During the years 2011–2014–2015, one serious injury of a common bottlenose dolphin occurred associated with the mid-Atlantic haul/beach seine fishery that could be ascribed to the Northern Migratory Coastal Stock. During 2014, a common bottlenose dolphin was found within a haul seine net in Virginia and released alive seriously injured (Maze-Foley and Garrison 2017–2020). The animal was ascribed to the Northern and Southern Migratory Coastal and NNCE stocks. This case was included in the stranding database and in the stranding totals presented in Table 4 (Northeast Regional Marine Mammal Stranding Network; NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 13 August 2019). As well as in the annual human-caused mortality and serious injury total for this stock (Table 2b). The mid-Atlantic haul/beach seine fishery had limited observer coverage by the NEFOP in 2010–2011. No observer coverage was allocated to this fishery during 2012–2015. No estimate of bycatch mortality is available for this fishery, and the documented interaction in this commercial gear represents a minimum known count of interactions in the last five years.

Hook and Line (Rod and Reel)

During the years 2011–2014–2015, stranding data identified two mortalities and one serious injury of common bottlenose dolphins.
bottlenose dolphins that could be ascribed to the Northern Migratory Coastal Stock for which hook and line gear entanglement or ingestion were documented. For one mortality, available evidence suggested the hook and line gear interaction contributed to the cause of death (2012-2018, Maryland/Virginia). This animal was ascribed solely to the Northern and Southern Migratory Coastal Stocks. For a second mortality that was also ascribed solely to the Northern Migratory Coastal Stock, the carcass was in a state of advanced decomposition and it could not be determined whether the hook and line gear interaction contributed to cause of death (2018, Delaware). For a third mortality, available evidence suggested the hook and line gear interaction did not contribute to the cause of death (2016, Virginia). This animal was ascribed to the Northern and Southern Migratory Coastal and NNCES stocks (Urian in review assigned solely to the NNCES Stock). The fourth mortality, The second dolphin was in a state of advanced decomposition and it could not be determined whether the hook and line gear interaction contributed to cause of death (2014, Virginia; Maze-Foley et al. 2019). This mortality was ascribed solely to the Northern Migratory Coastal Stock alone. In addition, there was one live animal documented with an entanglement (2017, Virginia), and this animal was considered seriously injured (Maze-Foley and Garrison 2020). It was ascribed to the Northern and Southern Migratory Coastal stocks. The two mortalities All of these cases were included in the stranding database and in the stranding totals presented in Table 4 (Northeast Regional Marine Mammal Stranding Network; NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 13 August 2019–6 June 2016). The 2012-2018 mortality for which evidence suggested the gear contributed to the cause of death and the 2017 serious injury are included in the annual human-caused mortality and serious injury total for this stock (Table 23b).

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

**Other Mortality**

Historically, there have been occasional mortalities of common bottlenose dolphins during activities including during directed live capture-release studies, turtle relocation trawls, and fisheries surveys (Waring et al. 2016); however, none were documented during 2011-2014–2018-2015 that could be ascribed to the Northern Migratory Coastal Stock. All mortalities and serious injuries from known human-caused sources for the Northern Migratory Coastal Stock are summarized in Tables 23a, 23b and 23c.

### Table 3a. Summary of the incidental mortality and serious injury of common bottlenose dolphins of the Northern Migratory Coastal Stock for the commercial mid-Atlantic gillnet fishery, which has an ongoing, systematic federal observer program. The years sampled, the type of data used, the annual percentage observer coverage, the observed serious injuries and mortalities recorded by on-board observers, and the mean annual estimate of mortality and serious injury and its CV are provided. Minimum and maximum values are reported due to uncertainty in the assignment of mortalities to this particular stock because there is spatial overlap with other common bottlenose dolphin stocks in certain areas and seasons.
## Table 3b. Summary of the incidental mortality and serious injury of common bottlenose dolphins of the Northern Migratory Coastal Stock during 2014–2018 from commercial fisheries that do not have ongoing, systematic federal observer programs. Counts of mortality and serious injury based on stranding data are given. Minimum and maximum values are reported in individual cells when there is uncertainty in the assignment of mortalities to this particular stock due to spatial overlap with other common bottlenose dolphin stocks in certain areas and seasons.

### Table 2b. Summary of the incidental mortality and serious injury of common bottlenose dolphins of the Northern Migratory Coastal Stock during 2011–2015 from commercial fisheries that do not have ongoing, systematic federal observer programs. Counts of mortality and serious injury based on stranding data are given. Minimum and maximum values are reported in individual cells when there is uncertainty in the assignment of mortalities to this particular stock due to spatial overlap with other common bottlenose dolphin stocks in certain areas and seasons.

<table>
<thead>
<tr>
<th>Fishery</th>
<th>Years</th>
<th>Data Type</th>
<th>Observed Serious Injury</th>
<th>Observed Mortality</th>
<th>Mean Annual Estimated Mortality and Serious Injury (CV) Based on Observer Data</th>
</tr>
</thead>
<tbody>
<tr>
<td>Mid-Atlantic Gillnet</td>
<td>2011–2015</td>
<td>Obs. Data Logbook</td>
<td>0, 0, 0, 0, 0</td>
<td>0-0,0,0,2,0</td>
<td>Min=6.4, 11.8 (0.1832) Max=12.2, 19.5 (0.1422)</td>
</tr>
<tr>
<td></td>
<td>2014–2018</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td>Mean Annual</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td>Mortality</td>
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</tr>
<tr>
<td></td>
<td>due to the</td>
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</tr>
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<tr>
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<td></td>
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<td></td>
</tr>
<tr>
<td></td>
<td>gillnet</td>
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<table>
<thead>
<tr>
<th>Fishery</th>
<th>Years</th>
<th>Data Type</th>
<th>5-year Count Based on Stranding Data and the Marine Mammal Authorization Program</th>
</tr>
</thead>
<tbody>
<tr>
<td>Northeast Sink Gillnet</td>
<td>2014–2018</td>
<td>Federal Observer, and Marine Mammal</td>
<td>Min=0 Max=4</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Authorization Program</td>
<td></td>
</tr>
<tr>
<td>Chesapeake Bay Inshore</td>
<td>2011–2015, 2014–2018</td>
<td>Limited Observer and Stranding Data</td>
<td>Min=0 Max=21</td>
</tr>
<tr>
<td>Gillnet§</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Virginia Pound Net§</td>
<td>2011–2015, 2014–2018</td>
<td>Stranding Data</td>
<td>Min=0 Max=10</td>
</tr>
<tr>
<td>Mid-Atlantic Purse Seine</td>
<td>2011–2015, 2014–2018</td>
<td>Limited Observer and Stranding Data</td>
<td>0</td>
</tr>
</tbody>
</table>
### Atlantic Blue Crab Trap/Pot

<table>
<thead>
<tr>
<th>Year</th>
<th>Stranding Data</th>
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</thead>
<tbody>
<tr>
<td>2011-2015</td>
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<td>2014-2018</td>
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### Mid-Atlantic Haul/Beach Seine

<table>
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<th>Max</th>
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</tbody>
</table>

### Hook and Line

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<tr>
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<th>Stranding Data</th>
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<th>Max</th>
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</thead>
<tbody>
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<td>2011-2015</td>
<td>2014-2018</td>
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<table>
<thead>
<tr>
<th>Min</th>
<th>Max</th>
</tr>
</thead>
<tbody>
<tr>
<td>0.4</td>
<td>12.0</td>
</tr>
</tbody>
</table>

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

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

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

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

*c* Hook and line interactions are counted here if the available evidence suggested the hook and line gear contributed to the cause of death per Maze-Foley et al. (2019). See "Hook and Line" text for more details.

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

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

### Mean Annual Mortality due to the observed commercial mid-Atlantic gillnet fishery (2011–2015 2014–2018) (Table 23a)

<table>
<thead>
<tr>
<th>Min</th>
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<tbody>
<tr>
<td>6.1</td>
<td>11.8</td>
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### Mean Annual Mortality due to unobserved commercial fisheries (2011–2015 2014–2018) (Table 23b)

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<th>Max</th>
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</thead>
<tbody>
<tr>
<td>0.4</td>
<td>4.0</td>
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</table>
### Mean Annual Mortality due to research and other takes (2011–2015 2014–2018)

<p>| | |</p>
<table>
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<tr>
<th></th>
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<tbody>
<tr>
<td>Min</td>
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</tr>
<tr>
<td>Max</td>
<td>0</td>
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</tbody>
</table>

### Minimum Total Mean Annual Human-Caused Mortality and Serious Injury (2011–2015 2014–2018)

<p>| | |</p>
<table>
<thead>
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<th></th>
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</thead>
<tbody>
<tr>
<td>Min</td>
<td>6.1</td>
</tr>
<tr>
<td>Max</td>
<td>12.2</td>
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</tbody>
</table>

Strandings

Between 2014–2014 and 2018–2015, 4,414,692 common bottlenose dolphins that were ascribed to the Northern Migratory Coastal Stock stranded along the Atlantic coast between North Carolina and New York (Table 43; Northeast Regional (NER) Marine Mammal Stranding Network; Southeast Regional (SER) Marine Mammal Stranding Network; NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 21 May 2019 18 May 2016 (SER) and 13 August 2019 6 June 2016 (NER); Maze-Foley et al. 2019). There was evidence of human interaction for 80 of these strandings, of which 51 (64%) were fisheries interactions and 4 (5%) showed evidence of a boat strike. No evidence of human interaction was detected for 134 strandings, and for the remaining 478 strandings, it could not be determined if there was evidence of human interaction. (HI) for 275 of these strandings, and for 247 it was determined there was no evidence of HI. The remaining 99 showed evidence of HI, of which 57 (64%) were fisheries interactions and 10 (11%) showed evidence of a boat strike (Table 3). It should be recognized that evidence of human interaction (HI) does not indicate cause of death, but rather only that there was evidence of interaction with a fishery (e.g., line marks, net marks) or evidence of a boat strike, gunshot wound, mutilation, etc., at some point. Stranding data probably underestimate the extent of human and fishery-related mortality and serious injury because not all of the dolphins that die or are seriously injured in human interactions wash ashore, or, if they do, they are not all recovered (Peltier et al. 2012; Wells et al. 2015). Additionally, not all carcasses will show evidence of human interaction, entanglement or other fishery-related interaction due to decomposition, scavenger damage, etc. (Byrd et al. 2014). Finally, the level of technical expertise to recognize signs of human interaction varies among stranding network personnel.

The assignment of animals to a single stock is impossible in some seasons and regions due to spatial and temporal overlap among several common bottlenose dolphin stocks (Maze-Foley et al. 2019). Of the 1,111,692 strandings ascribed to the Northern Migratory Coastal Stock, 634,297 were ascribed solely to this stock. Therefore, the counts in Table 43 likely include some animals from the Southern Migratory Coastal, and NNCES, and Offshore stocks and, therefore, overestimate the number of strandings for the Northern Migratory Coastal Stock; those strandings that could not be ascribed to the Northern Migratory Coastal Stock alone are also included in the counts for these other stocks as appropriate. In addition, stranded carcasses are not routinely identified to either the offshore or coastal morphotype of common bottlenose dolphin, therefore, it is possible that some of the reported strandings were of the offshore form though that number is likely to be low, especially for states south of New York (Byrd et al. 2014).

This stock has also been impacted by two large unusual mortality events (UMEs), one in 1987–1988 and one in 2013–2015, both of which have been attributed to morbillivirus epidemics (Lipscomb et al. 1994; Morris et al. 2015). Both UMEs included deaths of dolphins north of Assateague, Virginia, in summer, corresponding solely to the Northern Migratory Coastal Stock area. When the impacts of the 1987–1988 UME were being assessed, only a single coastal stock of common bottlenose dolphin was thought to exist along the U.S. eastern seaboard from New York to Florida (Scott et al. 1988), so impacts to the Northern Migratory Coastal Stock alone are not known. However, it was estimated that between 10 and 50% of the coast-wide stock died as a result of this UME (Scott et al. 1988; Eguichi 2002). For the 2013–2015 UME, a total of 1,872,1614 stranded common bottlenose dolphins were recovered in the
UME area which stretched from New York to Brevard County, Florida. Of these, 381,348 stranded dolphins were recovered from the states of New York, New Jersey, Delaware, and Maryland (https://www.fisheries.noaa.gov/national/marine-life-distress/2013-2015-bottlenose-dolphin-unusual-mortality-event-mid-atlantic http://www.nmfs.noaa.gov/pr/health/mmume/midatldolphins2013.html, accessed 13 November 2019October 3, 2016). While some of these deaths may be attributable to the Offshore Stock, the majority likely came from the Northern Migratory Coastal Stock given their geographic location. This number is likely an underestimate of the total number of deaths for this stock, however, because it does not include animals that stranded in Virginia and North Carolina in cold water months that might have come from this stock, and not all dolphins that died during the UME would have been recovered. An analysis of trends in abundance for common bottlenose dolphins coast-wide (New Jersey to Florida) indicated a statistically significant decline in population size between 2011 and 2016 (Garrison et al. 2017a). A weighted generalized linear model was used to evaluate trends in coast-wide population size based on aerial surveys conducted between 2002 and 2016 (see Population Size above for survey descriptions). The model included a linear term for survey year and an interaction term to test for a difference in slope between 2002–2011 and 2011–2016. Estimates were weighted by the inverse of their standard error to reduce the influence of less certain estimates. There was no significant trend in population size between 2002 and 2011; however, there was a statistically significant (p=0.0308) change in slope between 2011 and 2016, indicating a decline in population size. The coast-wide inverse-variance weighted average estimate for coastal common bottlenose dolphins during 2011 was 41,456 (CV=0.30) while the estimate during 2016 was 19,470 (CV=0.23; Garrison et al. 2017a). It is possible that this apparent decline in common bottlenose dolphin abundance in coastal waters along the eastern seaboard is a result of the 2013–2015 UME. An assessment of the impacts of the 2013–2015 UME on common bottlenose dolphin stocks in the wNA is ongoing.

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

<table>
<thead>
<tr>
<th>State</th>
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<th>2016</th>
<th>2017</th>
<th>2018</th>
<th>Total</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
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<td>HI No</td>
<td>CBD</td>
<td>HI Yes</td>
<td>HI No</td>
<td>CBD</td>
</tr>
<tr>
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<td>3</td>
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</tr>
<tr>
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<td>44</td>
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<td>6</td>
<td>55</td>
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<td>0</td>
<td>7</td>
<td>1</td>
<td>2</td>
<td>5</td>
</tr>
<tr>
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<td>24</td>
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<td>1</td>
<td>8</td>
<td>0</td>
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<td>2</td>
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<tr>
<td>Total</td>
<td>137</td>
<td>151</td>
<td>118</td>
<td>133</td>
<td>153</td>
<td>692</td>
</tr>
</tbody>
</table>

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

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

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

<table>
<thead>
<tr>
<th>State</th>
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<th>2015</th>
</tr>
</thead>
<tbody>
<tr>
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<td>HI</td>
<td>HI</td>
</tr>
<tr>
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</tr>
<tr>
<td></td>
<td></td>
<td>Yes</td>
<td>No</td>
<td>Yes</td>
<td>No</td>
</tr>
<tr>
<td>North Carolina</td>
<td>6\textsuperscript{a} &amp; 16 &amp; 22 &amp; 11\textsuperscript{c} &amp; 14 &amp; 16 &amp; 2\textsuperscript{d} &amp; 24 &amp; 36 &amp; 3\textsuperscript{e} &amp; 15 &amp; 23 &amp; 3\textsuperscript{f} &amp; 10 &amp; 23</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Virginia\textsuperscript{g}</td>
<td>2\textsuperscript{h} &amp; 3 &amp; 36 &amp; 11\textsuperscript{i} &amp; 7 &amp; 37 &amp; 11\textsuperscript{j} &amp; 17 &amp; 137 &amp; 5\textsuperscript{k} &amp; 5 &amp; 44 &amp; 9\textsuperscript{l} &amp; 5 &amp; 55</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Maryland</td>
<td>2\textsuperscript{m} &amp; 4 &amp; 4 &amp; 0 &amp; 2 &amp; 2 &amp; 4\textsuperscript{n} &amp; 19 &amp; 43 &amp; 0 &amp; 1 &amp; 6 &amp; 0 &amp; 2 &amp; 8</td>
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<td></td>
<td></td>
<td></td>
</tr>
<tr>
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<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
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<td></td>
<td></td>
<td></td>
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<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Annual Total</td>
<td>129</td>
<td>144</td>
<td>551</td>
<td>137</td>
<td>150</td>
</tr>
</tbody>
</table>
179

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

*b Includes 5 FIs.

c *Includes 10 FIs, 2 of which had markings indicative of interactions with gillnet gear (mortalities).*

d *Includes 2 FIs, 1 of which had markings indicative of interactions with gillnet gear (mortality).*

e *Includes 3 FIs, 1 of which had markings indicative of interactions with gillnet gear (mortality).*

f Includes 2 FIs.

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

h *Includes 6 FIs. One FI was an entanglement interaction in a Virginia pound net (mortality) and 2 FIs were mortalities with twisted twine markings indicative of interactions with Virginia pound net gear.*

i *Includes 1 mortality with evidence of a boat strike and 7 FIs. One FI was an entanglement interaction with hook and line gear (mortality) and 1 FI was a mortality with twisted twine markings indicative of interactions with Virginia pound net gear.*

j *Includes 8 FIs, 1 of which was a mortality with twisted twine markings indicative of interactions with Virginia pound net gear.*

k *Includes 3 FIs, 1 of which involved ingestion of hook and line gear (mortality). Another animal was released alive seriously injured following capture in a haul seine. Also includes 1 mortality with evidence of a boat strike.*

l *Includes 3 FIs. One FI was an entanglement interaction with commercial gillnet gear (mortality, Chesapeake Bay inshore gillnet fishery), and 2 FIs had markings indicative of interactions with gillnet gear (mortalities). Also includes 1 mortality with evidence of a boat strike.*

m *Includes 1 FI.

n *Includes 3 FIs, 1 of which was an entanglement interaction with commercial gillnet gear (mortality, Chesapeake Bay inshore gillnet fishery).*

o *Includes 2 mortalities with evidence of a boat strike.*

p *Includes 1 FI.*

q *Includes 1 FI.*

r *Includes 1 FI.

s *Includes 1 mortality and 1 live animal with evidence of a boat strike.*
* Includes 1 FI and 2 mortalities with evidence of a boat strike.

* Includes 1 FI.

* A mortality with evidence of a boat strike.
HABITAT ISSUES

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

STATUS OF STOCK

Common bottlenose dolphins in the western North Atlantic are not listed as threatened or endangered under the Endangered Species Act, but the Northern Migratory Coastal Stock is a strategic stock due to its designation as depleted under the MMPA. From 1995 to 2001, NMFS recognized only the western North Atlantic Coastal Stock of common bottlenose dolphins in the western North Atlantic, and this stock was listed as depleted as a result of a UME in 1988–1989 (64 FR 17789, April 6, 1993). The stock structure was revised in 2008, 2009, and 2010, to recognize resident estuarine stocks and migratory and resident coastal stocks. The Northern Migratory Coastal Stock retains the depleted designation as a result of its origin from the western North Atlantic Coastal Stock. This stock is presumed to be below OSP due to its designation as depleted. PBR for the Northern Migratory Coastal Stock is 48 and so the zero mortality rate goal, 10% of PBR, is 4.8. The documented mean annual human-caused mortality for this stock for 2014–2018 ranged between a minimum of 6.1 and a maximum of 12.2. However, these estimates are biased low for the following reasons: 1) the total U.S. human-caused mortality and serious injury for the Northern Migratory Coastal Stock cannot be directly estimated because of the spatial overlap among the stocks of common bottlenose dolphins that occupy waters of North Carolina and Virginia resulting in uncertainty in the stock assignment of some takes, 2) there are several commercial fisheries operating within this stock’s boundaries that have little to no observer coverage, and 3) this mortality estimate incorporates a count of verified human-caused deaths and serious injuries and should be considered a minimum (NMFS 2016). The total fishery-related mortality and serious injury for this stock is not less than 10% of the calculated PBR and therefore, cannot be considered to be insignificant and approaching a zero mortality and serious injury rate. The impacts of two large UMEs on the status of this stock are unknown. Analysis of trends in abundance suggests a probable decline in stock size between 2010–2011 and 2016, concurrent with a large UME in the area; however, there is limited power to evaluate trends given uncertainty in stock distribution, lack of precision in abundance estimates, and a limited number of surveys.

REFERENCES CITED


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


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

**STOCK DEFINITION AND GEOGRAPHIC RANGE**

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

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

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

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

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

**POPULATION SIZE**

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

**Background**

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

**Earlier abundance estimates (>8 years old)**

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

**Recent surveys and abundance estimates**

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

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

The abundance estimates for 2010, 2011, and the 2016 summer aerial surveys were 9,217 (CV=0.51), 4,987 (CV=0.63), and 3,751 (CV=0.60), respectively (Garrison et al. 2017a). The 2016 estimate was used as the best estimate of the current population size for the stock due to possible effects from the 2013–2015 unusual mortality event. Uncertainties in the abundance estimate arise primarily from annual, and unquantified, variation in stock distribution. Another unquantified source of uncertainty in the abundance estimate is the potential overlap of this stock.
(during summer) with the NNCES Stock in near-shore ocean waters within 1 km from shore.

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

<table>
<thead>
<tr>
<th>Month/Year</th>
<th>Area</th>
<th>Nbest</th>
<th>CV</th>
</tr>
</thead>
<tbody>
<tr>
<td>July–August 2002</td>
<td>Cape Lookout, North Carolina (34.6°N) to Assateague, Virginia (37.9°N)</td>
<td>19,316</td>
<td>0.31</td>
</tr>
<tr>
<td>July–August 2004</td>
<td>Cape Lookout, North Carolina (34.6°N) to Assateague, Virginia (37.9°N)</td>
<td>29,535</td>
<td>0.33</td>
</tr>
<tr>
<td>July–August 2010</td>
<td>Cape Lookout, North Carolina (34.6°N) to Assateague, Virginia (37.9°N)</td>
<td>9,217</td>
<td>0.51</td>
</tr>
<tr>
<td>July–August 2011</td>
<td>Cape Lookout, North Carolina (34.6°N) to Assateague, Virginia (37.9°N)</td>
<td>4,987</td>
<td>0.64</td>
</tr>
<tr>
<td>July–August 2016</td>
<td>Cape Lookout, North Carolina (34.6°N) to Assateague, Virginia (37.9°N)</td>
<td>3,751</td>
<td>0.60</td>
</tr>
</tbody>
</table>

Minimum Population Estimate

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

Current Population Trend

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

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

CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

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

POTENTIAL BIOLOGICAL REMOVAL

Potential Biological Removal (PBR) is the product of the minimum population size, one-half the maximum productivity rate, and a “recovery” factor (MMPA Sec. 3. 16 U.S.C. 1362; Wade and Angliss 1997; Wade 1998). The minimum population size of the Southern Migratory Coastal Stock of common bottlenose dolphins is 2,353. The maximum productivity rate is 0.04, the default value for cetaceans. The recovery factor is 0.5 because this stock is depleted. The recovery factor is 0.48 because the CV of the average mortality estimate is greater than 0.3 (Wade and Angliss 1997). PBR for this stock of common bottlenose dolphins is 2423 (Table 2).

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

<table>
<thead>
<tr>
<th>Nbest</th>
<th>CV</th>
<th>Nmin</th>
<th>Fr</th>
<th>Rmax</th>
<th>PBR</th>
</tr>
</thead>
<tbody>
<tr>
<td>3,751</td>
<td>0.60</td>
<td>2,353</td>
<td>0.5</td>
<td>0.04</td>
<td>24</td>
</tr>
</tbody>
</table>

ANNUAL HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

The total annual human-caused mortality and serious injury for the Southern Migratory Coastal Stock during 2014–2015 is unknown. The minimum mean annual fishery-related mortality and serious injury for observed fisheries and strandings identified as fishery-related ranged between 0 and 18.3 (CV=0.31). No additional mortality or serious injury was documented from other human-caused sources (e.g., fishery research) and therefore, the minimum total mean annual human-caused mortality and serious injury for this stock during 2014–2015 ranged between 0 and 18.3 (Tables 23a, 23b and 23c). This range reflects several sources of uncertainty and is a minimum because: 1) not all fisheries that could interact with this stock are observed, 2) stranding data are used as an indicator of fishery-related interactions and not all dead animals are detected and recovered by the stranding network (Peltier et al. 2012; Wells et al. 2015), 3) cause of death is not (or cannot be) routinely determined for stranded carcasses, 4) the estimate includes an actual count of verified human-caused deaths and serious injuries and should be considered a minimum (NMFS 2016), and 5) the spatiotemporal overlap between this stock and other common bottlenose dolphin stocks throughout its range introduces uncertainty in assignment of mortalities to stock. In the sections below, dolphin mortalities and serious injuries were ascribed to a stock or stocks by comparing the season
and geographic location of the take/stranding to the stock boundaries and geographic range delimited for each stock (Lyssikatos and Garrison 2018).

**Fishery Information**

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

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

**Mid-Atlantic Gillnet**

The mid-Atlantic gillnet fishery operates along the coast from North Carolina through New York (2016 List of Fisheries) and overlaps with the Southern Migratory Coastal Stock in the northern part of its range. North Carolina is the largest component of the mid-Atlantic gillnet fishery in terms of fishing effort and observed marine mammal takes (Palka and Rossman 2001; Lyssikatos and Garrison 2018 [in review]). This fishery is currently observed by the Northeast Fisheries Observer Program, and previously was observed by both the Northeast and Southeast Fisheries Observer Programs (through 2016). The Bottlenose Dolphin Take Reduction Team was convened in October 2001, in part, to reduce bycatch in gillnet gear. The Bottlenose Dolphin Take Reduction Plan (BDTRP) was implemented in May 2006 and resulted in changes to gillnet gear configurations and fishing practices (50 CFR 24776, April 26, 2006, available at [https://www.federalregister.gov/documents/2006/04/26/06-3909/taking-of-marine-mammals-incidental-to-commercial-fishing-operations-bottlenose-dolphin-take](http://www.nmfs.noaa.gov/pr/pdfs/fr/fr71-24776.pdf)). In addition, two subsequent amendments to the BDTRP were implemented in 2008 and 2012 regarding gear restrictions for medium-mesh gillnets in North Carolina waters (https://www.fisheries.noaa.gov/national/marine-mammal-protection/bottlenose-dolphin-take-reduction-plan). Mortality estimates for the period (2002–2006) immediately prior to implementation of the BDTRP and 2007–2011 are available in the 2015 stock assessment report for the Northern Migratory Coastal Stock (Waring et al. 2016 [5]). The current report covers the most recent available five-year estimate (NMFS 2016) for 2011–2015. Mortality estimation for this stock is difficult because 1) observed takes are statistically rare events, 2) the Southern Migratory Coastal, Northern Migratory Coastal, NNCEs, and SNCEs stocks of common bottlenose dolphin overlap in coastal waters of North Carolina and Virginia at different times of the year, and therefore it is not always possible to definitively assign every observed mortality, or extrapolated bycatch estimate, to a specific stock, and 3) the low levels of federal observer coverage in state waters are likely insufficient to consistently detect bycatch events (Lyssikatos and Garrison 2018). To help address the first problem, two different analytical approaches were used to estimate common bottlenose dolphin bycatch rates during the period 2011–2014 and 2015: 1) a simple annual ratio estimator of catch per unit effort (CPUE = observed catch/observed effort) per year based directly upon the observed data and 2) a pooled CPUE approach (where all observed data from the most recent five years were combined into one sample to estimate CPUE; Lyssikatos and Garrison 2018). In each case, the annual reported fishery effort (defined as a fishing trip) was multiplied by the estimated bycatch rate to develop annual estimates of fishery-related mortality. Next, the two model estimates (and the associated uncertainty) were averaged, in order to account for the uncertainty in the two approaches, to produce an estimate of the mean mortality of common bottlenose dolphins for this fishery (Lyssikatos and Garrison 2018). To help address the second problem, minimum and maximum mortality estimates were calculated per stock to indicate the range of uncertainty in assigning observed takes to stock (Lyssikatos and Garrison 2018). Uncertainties and potential biases are described in Lyssikatos and Garrison (2018).

During the most recent 5-year time period, 2011–2014 and 2018, the combined average Northeast (NEFOP) and Southeast (SEFOP) Fisheries Observer Program observer coverage (measured in trips) for this fishery was 2.67% in state waters (0–3 miles from shore) and 5.36% in federal waters (3–200 miles from shore) (Lyssikatos in review and Garrison 2018). During these trips, observers documented one dolphin (mortality) entangled in...
small-mesh gillnet gear dolphin (mortality) off the coast of North Carolina that may have been from the Southern Migratory Coastal Stock. One observed take (NEFOP) occurred in July 2017, and the second observed take (SEFOP) occurred off northern North Carolina in September 2014, and was both ascribed to the NNSES and Southern Migratory Coastal stocks (Lyssikatos in review and Garrison 2018). The resultant 5-year mean minimum and maximum mortality estimates (2012–2017) for the Southern Migratory Coastal Stock were therefore 0 and 12.516.3 (CV=0.2331) animals per year, respectively (Table 23a; Lyssikatos in review and Garrison 2018).

Historical and recent stranding data have documented multiple cases of dead, stranded dolphins recovered with gillnet gear attached (Byrd et al. 2014; Waring et al. 20165; Lyssikatos and Garrison 2018). In October 2011, the stranding network recovered a dead dolphin from a fisherman who had incidentally caught it in a small mesh gillnet targeting spot in southern North Carolina during an unobserved trip. In July 2018, the stranding network recovered a dead dolphin entangled in gillnet gear in Virginia. This animal was ascribed to the Northern and Southern Migratory Coastal and SNCES stocks. Because there is already an observer program-based bycatch estimate for the Southern Migratory Coastal Stock for the mid-Atlantic gillnet fishery, and the bycatch estimate was not zero, the additional recovered animal was not added to the bycatch estimate. However, the overall minimum annual mortality for this stock is likely not zero. During the current 5-year period there were also four common bottlenose dolphin strandings, allfive in North Carolina and two in Virginia, with markings indicative of interaction with gillnet gear (Read and Murray 2000), but no gear was attached to the carcasses and it is unknown whether the interactions with the gear contributed to the death of these animals. All four cases were ascribed to multiple stocks including the Southern Migratory Coastal Stock.

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

There have been no documented mortalities or serious injuries of common bottlenose dolphins associated with the Southeastern U.S. Atlantic Shark Gillnet or Southeast Atlantic Gillnet fisheries during 2011–2016 that could be ascribed to the Southern Migratory Coastal Stock (Gulak et al. 2012; Mathers et al. 2013; 2014; 2015; 2016; 2017; 2018; 2020). These fisheries target sharks and finfish in waters between North Carolina and southern Florida. The majority of fishing effort occurs in federal waters because Florida, Georgia, and South Carolina, with limited exception, prohibit the use of gillnets in state waters. The Southeast Gillnet Observer Program observes these fisheries year-round (e.g., Mathers et al. 2016).

Chesapeake Bay Inshore Gillnet

During 2011–2018, stranding data documented one interactions (mortalityies) between a common bottlenose dolphins and inshore gillnet gear in Chesapeake Bay. In 2013, in Maryland, a dead dolphin was recovered entangled in large mesh gillnet gear. In 2015, in Virginia, another dead dolphin was recovered entangled in gillnet gear (this animal was also self-reported by the fisherman per the Marine Mammal Authorization Program). Both of these animals was were ascribed to the Northern and Southern Migratory Coastal stocks, and both are it is included in the annual human-caused mortality and serious injury total for this stock (Table 2b) as well as in the stranding database and stranding totals presented in Table 45 (Northeast Regional Marine Mammal Stranding Network; NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 13 August 2016). There is no observer coverage of this fishery within Maryland waters of Chesapeake Bay; however, within Virginia waters of Chesapeake Bay, there is a low level of observer coverage (<1%). No estimate of bycatch mortality is available for this fishery, and the documented interactions in this commercial gear represent a minimum known count of interactions in the last five years. Three other dead, stranded common bottlenose dolphins were recovered within Chesapeake Bay with markings indicative of interaction with gillnet gear (Read and Murray 2000), but no gear was attached to the carcasses and it is unknown whether the interactions with the gear contributed to the death of these animals. Two of these animals were ascribed to the Southern Migratory Coastal and NNSES stocks, and one was ascribed to the Northern and Southern Migratory Coastal and NNSES Stocks.

Virginia Pound Net

During 2011–2018, there were no documented mortalities or serious injuries involving pound net gear in Virginia. One stranded common bottlenose dolphin (mortality) was found entangled in pound net gear in Virginia. This animal was ascribed to the Northern and Southern Migratory Coastal stocks (in 2011, Northeast Regional Marine Mammal Stranding Network; NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 6 June 2016), and is included in the annual human caused mortality and serious injury total for this stock (Table 2b). However, during 2017–2018, an additional threeone dolphins stranded with twisted
twine markings indicative of interactions with pound net gear, but it is unknown whether the interactions with the gear contributed to the death of these animals, and these cases are not included in the annual human-caused mortality and serious injury total for this stock. This stranding was ascribed to the Southern Migratory Coastal and NNSES stocks. These three strandings were ascribed to the Southern and Northern Migratory Coastal stocks and the NNSES Stock. All of the strandings discussed here occurred inside estuarine waters near the mouth of the Chesapeake Bay in August, September, and October (Northeast Regional Marine Mammal Stranding Network; NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 13 August 2019). The overall impact of the Virginia pound net fishery on the Southern Migratory Coastal Stock is unknown due to the limited information on the stock’s movements. Because there is no systematic observer program for the Virginia pound net fishery, no estimate of bycatch mortality is available. An amendment to the BDTRP was implemented in 2015 requiring gear restrictions for VA pound nets in estuarine and coastal state waters of Virginia to reduce bycatch (https://www.fisheries.noaa.gov/national/marine-mammal-protection/bottlenose-dolphin-take-reduction-plan) and the documented interaction in this commercial gear represents a minimum known count of interactions in the last five years.

Atlantic Blue Crab Trap/Pot

During 2011–2014–2018–2015, stranding data documented nine cases of common bottlenose dolphins entangled in trap/pot gear that could be ascribed to the Southern Migratory Coastal Stock. One case was a mortalitisy, and two were serious injuries, and for the remaining three cases, it could not be determined whether the animals were seriously injured. These cases were ascribed to the Northern and Southern Migratory Coastal and NNSES stocks. The third case occurred during 2018 and was ascribed to the Southern Migratory Coastal and NNSES stocks. One serious injury occurred in 2014 in commercial blue crab trap/pot gear, and one occurred in 2015 in unidentified trap/pot gear. These cases were ascribed to the Southern Migratory Coastal and NNSES stocks. The remaining two serious injuries occurred in 2015 and 2017 in commercial blue crab trap/pot gear; one was ascribed to the Southern Migratory Coastal and South Carolina/Georgia Coastal stocks, and the other was ascribed to the Northern and Southern Migratory Coastal and NNSES stocks. The six mortalities and serious injuries all are included in the annual human-caused mortality and serious injury total for this stock (Table 2b). In addition, there were three cases where it could not be determined whether the animals were seriously injured. Two occurred in 2017. One case was ascribed to the Northern and Southern Migratory Coastal stocks, and the other was ascribed to the Southern Migratory Coastal and South Carolina/Georgia Coastal stocks. The third case occurred during 2018 and was ascribed to the Southern and Northern Migratory Coastal and NNSES stocks. These animals are also included in the stranding database and in the stranding totals presented in Table 43 (Northeast Regional Marine Mammal Stranding Network; Southeast Regional Marine Mammal Stranding Network; NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 21 May 2019; 18 May 2016; 21 May 2019). Details regarding the serious injury determinations can be found in Maze-Foley and Garrison (2020). Because there is no observer program, it is not possible to estimate the total number of mortalities associated with crab traps/pots and these documented interactions in this commercial gear represent a minimum known count of interactions with this fishery. Stranding data indicate that interactions with trap/pot gear occur at some unknown level in North Carolina (Byrd et al. 2014) and other regions of the southeast U.S. (Noke and Odell 2002; Burdett and McFee 2004).

North Carolina Roe Mullet Stop Net

During 2012–2014–2018–2015, stranding data there were no documented mortalities or serious injuries of common bottlenose dolphins in stop net gear. One mortality in which a common bottlenose dolphin was entangled in a stop net (this animal was also self-reported by the fisherman per the Marine Mammal Authorization Program). This mortality, which occurred during November 2013, was ascribed to the NNSES and Southern Migratory Coastal stocks and is included in the annual human-caused mortality and serious injury total for this stock (Table 2b). In addition, a dead stranded dolphin with line markings indicative of interactions with stop net gear in October 2015 ~300 yards from a stop net, but it is unknown whether the interaction with gear contributed to the death of this animal, and this case is not included in the annual human-caused mortality and serious injury total for this stock. This animal was ascribed to multiple stocks: the Southern Migratory Coastal, NNSES, and SNSES stocks. Both these mortalities are included in the stranding database and in the stranding totals presented in Table 43 (Southeast Regional Marine Mammal Stranding Network; NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 21 May 2019; 18 May 2016). No estimate of bycatch mortality is available for the stop net fishery, and the confirmed interaction in this commercial gear represents a minimum known count of interactions with this fishery.
in the last five years. This fishery has not had regular, ongoing federal or state observer coverage. However, the NMFS Beaufort laboratory observed this fishery in 2001–2002 (Byrd and Hohn 2010), and Duke University observed the fishery in 2005–2006 (Thayer et al. 2007). Entangled dolphins were not documented during these formal observations, but historical takes of dolphins entangled in stop nets occurred in 1993 and 1999 (Byrd and Hohn 2010).

**Mid-Atlantic Menhaden Purse Seine**

During 2011–2014–2018–2015, there were no documented mortalities or serious injuries in mid-Atlantic menhaden purse seine gear of common bottlenose dolphins that could be ascribed to the Southern Migratory Coastal Stock. The mid-Atlantic menhaden purse seine fishery historically reported an annual incidental take of one to five common bottlenose dolphins (NMFS 1991, pp. 5–73). There has been very limited federal observer coverage since 2008. No observer coverage was allocated to this fishery during 2014–2018 or during 2013–2015, and for 2012 only three trips were observed. Because there is no systematic observer program for this fishery, no estimate of bycatch mortality is available.

**Mid-Atlantic Haul/Beach Seine**

During 2011–2014–2018–2015, one serious injury of a common bottlenose dolphin occurred associated with the mid-Atlantic haul/beach seine fishery that could be ascribed to the Southern Migratory Coastal Stock. During 2014, a common bottlenose dolphin was found within a haul seine net in Virginia and released alive seriously injured (Maze-Foley and Garrison 2020). The animal was ascribed to the Northern and Southern Migratory Coastal and NNces stocks, and is included in the annual human-caused mortality and serious injury total for this stock (Table 2b) as well as in the stranding database and stranding totals presented in Table 43 (Northeast Regional Marine Mammal Stranding Network; NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 13 August 2019). The mid-Atlantic haul/beach seine fishery had limited observer coverage by the NEFOP in 2010–2011. No observer coverage was allocated to this fishery during 2012–2018–2015. No estimate of bycatch mortality is available for this fishery, and the documented interaction in this commercial gear represents a minimum known count of interactions in the last five years.

**Shrimp Trawl**

During 2011–2014–2018–2015, there were no documented mortalities or serious injuries of common bottlenose dolphins associated with the shrimp trawl fishery that could be ascribed to the Southern Migratory Coastal Stock. There has been very little systematic observer coverage of this fishery in the Atlantic during the last decade.

**Hook and Line (Rod and Reel)**

During 2011–2014–2018–2015, stranding data documented four mortalities and one serious injury that could be ascribed to the Southern Migratory Coastal Stock for which hook and line gear entanglement or ingestion were recorded. The serious injury (2017, Virginia) was ascribed to the Northern and Southern Coastal Migratory stocks (Maze-Foley and Garrison 2020). For one mortality, ascribed to the Southern Migratory Coastal and South Carolina/Georgia Coastal stocks, available evidence suggested the hook and line gear interaction contributed to the cause of death (2017, South Carolina; Maze-Foley et al. 2019). This serious injury and mortality are included in the annual human-caused mortality and serious injury total for this stock (Table 3b). For two of the remaining mortalities, evidence suggested the hook and line gear interactions were not a contributing factor to cause of death. Both of these mortalities occurred in 2016 (one in Virginia, one in North Carolina) and were ascribed to the Northern and Southern Migratory Coastal and NNces stocks (for the Virginia case, Urian in review assigned solely to the NNces Stock). For one mortality, evidence suggested the hook and line gear interaction was not a contributing factor to cause of death (2012, North Carolina). For two mortalities, it could not be determined if the hook and line gear interaction contributed to cause of death (2011, Virginia; 2011, South Carolina). All four mortalities were ascribed to multiple stocks: the Northern Migratory Coastal, South Carolina/Georgia Coastal Southern Migratory Coastal, and NNces stocks. For the final mortality, ascribed to the Southern Migratory Coastal and South Carolina/Georgia Coastal stocks, it could not be determined whether the hook and line gear interaction contributed to cause of death (2017, South Carolina; Maze-Foley et al. 2019). These mortalities All five cases were included in the stranding database and are included in the stranding totals presented in Table 43 (Northeast Regional Marine Mammal Stranding Network; Southeast Regional Marine Mammal Stranding Network; NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 21 May 2019–44 May 2016 (SER) and 13 August 2019–6 June 2016 (NER)). The 2012 mortality for which evidence suggested the gear contributed to the cause of death is included in the annual human-caused mortality and serious injury total for this stock (Table 2b).
It should be noted that, in general, it cannot be determined if rod and reel hook and line gear originated from a commercial (i.e., commercial fisherman, charter boat, or headboat) or recreational angler because the gear type used by both sources is typically the same. Also, it is not possible to estimate the total number of interactions with hook and line gear because there is no systematic observer program. The documented interactions in this commercial gear represent a minimum known count of interactions with this fishery.

**Other Mortality**

Historically, there have been occasional mortalities of common bottlenose dolphins during research activities (Waring et al. 2016); however, none were documented during 2011–2014 or 2015 that could be ascribed to the Southern Migratory Coastal Stock. All mortalities and serious injuries from known human-caused sources for the Southern Migratory Coastal Stock are summarized in Tables 2a, 2b, and 2c.

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

<table>
<thead>
<tr>
<th>Fishery</th>
<th>Years</th>
<th>Data Type</th>
<th>Observer Coverage</th>
<th>Observed Serious Injury</th>
<th>Observed Mortality</th>
<th>Mean Annual Estimated Mortality and Serious Injury (CV) Based on Observer Data</th>
</tr>
</thead>
<tbody>
<tr>
<td>Mid-Atlantic Gillnet</td>
<td>2011–2015, 2016–2018</td>
<td>Obs. Data Logbook</td>
<td>2.0, 2.6, 3.1, 3.6, 5.6, 9.8, 7.0, 6.4</td>
<td>0, 0, 0, 0, 0</td>
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<td>Southeastern U.S. Atlantic Shark Gillnet</td>
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Southeast Atlantic Gillnet 2011–2015 2014–2018 Obs. Data Logbook NA due to uncertainty in reported effort 0, 0, 0, 0, 0 0, 0, 0, 0, 0 No estimate

<table>
<thead>
<tr>
<th>Fishery</th>
<th>Years</th>
<th>Data Type</th>
<th>5-year Count Based on Stranding Data</th>
</tr>
</thead>
<tbody>
<tr>
<td>Chesapeake Bay Inshore Gillnet</td>
<td>2011–2015 2014–2018</td>
<td>Limited Observer and Stranding Data</td>
<td>Min=0 Max=21</td>
</tr>
<tr>
<td>Virginia Pound Net</td>
<td>2011–2015 2014–2018</td>
<td>Stranding Data</td>
<td>Min=0 Max=10</td>
</tr>
<tr>
<td>Atlantic Blue Crab Trap/Pot</td>
<td>2011–2015 2014–2018</td>
<td>Stranding Data</td>
<td>Min=0 Max=26</td>
</tr>
</tbody>
</table>

Mean Annual Mortality due to the observed commercial mid-Atlantic gillnet fishery (20112014–20182015)

Table 3b. Summary of the incidental mortality and serious injury of common bottlenose dolphins of the Southern Migratory Coastal Stock during 2014–2018 from commercial fisheries that do not have ongoing, systematic federal observer programs. Counts of mortality and serious injury based on stranding data are given. Minimum and maximum values are reported in individual cells when there is uncertainty in the assignment of mortalities to this particular stock due to spatial overlap with other common bottlenose dolphin stocks in certain areas and seasons. In addition, mortality due to research and other non-commercial fishery takes are included, as well as a total mean annual human caused mortality and serious injury summed from all sources.

<table>
<thead>
<tr>
<th>Fishery</th>
<th>Years</th>
<th>Data Type</th>
<th>5-year Count Based on Stranding Data</th>
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</thead>
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<td>Chesapeake Bay Inshore Gillnet</td>
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<td>Limited Observer and Stranding Data</td>
<td>Min=0 Max=21</td>
</tr>
<tr>
<td>Virginia Pound Net</td>
<td>2011–2015 2014–2018</td>
<td>Stranding Data</td>
<td>Min=0 Max=10</td>
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<tr>
<td>Atlantic Blue Crab Trap/Pot</td>
<td>2011–2015 2014–2018</td>
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</tbody>
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Table 2b. Summary of the incidental mortality and serious injury of common bottlenose dolphins of the Southern Migratory Coastal Stock during 2011–2015 for commercial fisheries that do not have ongoing, systematic federal observer programs. Counts of mortality and serious injury based on stranding data are given. Minimum and maximum values are reported in individual cells when there is uncertainty in the assignment of mortalities to this particular stock due to spatial overlap with other common bottlenose dolphin stocks in certain areas and seasons.
Table 3c. Summary of the incidental mortality and serious injury of common bottlenose dolphins of the Southern Migratory Coastal Stock during 2014–2018 from all sources, including observed commercial fisheries, unobserved commercial fisheries, and research and other takes. See the Annual Human-Caused Mortality and Serious Injury section for biases and limitations of mortality estimates.
Mean Annual Mortality due to the observed commercial mid-Atlantic gillnet fishery (2011–2015) (Table 23a)

<table>
<thead>
<tr>
<th></th>
<th>Min=0</th>
<th>Max=13.36.3</th>
</tr>
</thead>
</table>

Mean Annual Mortality due to unobserved commercial fisheries (2011–2015) (Table 23b)

<table>
<thead>
<tr>
<th></th>
<th>Min=0</th>
<th>Max=1.8</th>
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</thead>
</table>

Research Takes (5-year Min/Max Count)

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Other takes (5-year Min/Max Count)

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Mean Annual Mortality due to research and other takes (2011–2015)

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</thead>
</table>

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

<table>
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<tr>
<th></th>
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<th>Max=14.3</th>
</tr>
</thead>
</table>

Strandings

During 2011–2015, 965 common bottlenose dolphins stranded along the Atlantic coast between Florida and Virginia that could be ascribed to the Southern Migratory Coastal Stock (Table 4; Northeast Regional Marine Mammal Stranding Network; Southeast Regional Marine Mammal Stranding Network; NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 21 May 2019; 13 August 2019; 6 June 2016 (NER); Maze-Foley et al. 2019). There was evidence of human interaction for 59 of these strandings, of which 43 (73%) were fisheries interactions and 1 (2%) showed evidence of a boat strike (Table 4). No evidence of human interaction was detected for 121 strandings, and for the remaining 385 strandings, it could not be determined if there was evidence of human interaction. It could not be determined if there was evidence of human interaction for 677 of these strandings, and for 203 it was determined there was no evidence of human interaction. The remaining 85 showed evidence of human interactions, of which 57 (67%) were fisheries interactions and four (5%) showed evidence of a boat strike (Table 4). It should be recognized that evidence of human interaction does not indicate cause of death, but rather only that there was evidence of interaction with a fishery (e.g., line marks, net marks) or evidence of a boat strike, gunshot wound, mutilation, etc., at some point. Stranding data probably underestimate the extent of human and fishery-related mortality and serious injury because not all of the dolphins that die or are seriously injured in human interactions wash ashore, or, if they do, they are not all recovered (Peltier et al. 2012; Wells et al. 2015). Additionally, not all carcasses will show evidence of human interaction, entanglement or other fishery-related interaction due to decomposition, scavenger damage, etc. (Byrd et al. 2014). Finally, the level of technical expertise to recognize signs of human interaction varies among stranding network personnel.

The assignment of animals to a single stock is impossible in some seasons and regions due to spatial and temporal overlap among several common bottlenose dolphin stocks (Maze-Foley et al. 2019). Due to its migratory behavior, the Southern Migratory Coastal Stock can overlap with other common bottlenose dolphin stocks in every season. None of the 965 strandings ascribed to the Southern Migratory Coastal Stock were ascribed solely to this stock. Therefore, the counts in Table 4 likely include animals from other stocks and therefore overestimate the number of strandings attributable to the Southern Migratory Coastal Stock. Those strandings that could not be definitively ascribed to the Southern Migratory Coastal Stock alone are also included in the counts for these other stocks as appropriate. In addition, stranded carcasses are not routinely identified to either the offshore or coastal
morphotype of common bottlenose dolphin, therefore it is possible that some of the reported strandings were of the offshore form though that number is likely to be low (Byrd et al. 2014).

This stock has also been impacted by three unusual mortality events (UMEs). Two events, one in 1987–1988 and one in 2013–2015, have been attributed to morbillivirus epidemics (Lipscomb et al. 1994; Morris et al. 2015). Both UMEs included deaths of dolphins in spatiotemporal locations that apply to the Southern Migratory Coastal Stock. When the impacts of the 1987–1988 UME were being assessed, only a single coastal stock of common bottlenose dolphin was thought to exist along the U.S. eastern seaboard from New York to Florida (Scott et al. 1988), so impacts to the Southern Migratory Coastal Stock alone are not known. However, it was estimated that between 10 and 50% of the coast-wide stock died as a result of this UME (Scott et al. 1988; Eguchi 2002). The total number of stranded common bottlenose dolphins from New York through North Florida (Brevard County) during the 2013–2015 UME was 1,614–1,827 (https://www.fisheries.noaa.gov/national/marine-life-distress/2013-2015-bottlenose-dolphin-unusual-mortality-event-mid-atlantic http://www.nmfs.noaa.gov/pr/health/mmume/midatldolphins2013.html, accessed 13 November 2019). Most strandings and morbillivirus positive animals have been recovered from the ocean side beaches rather than from within the estuaries, suggesting that coastal stocks have been more impacted by this UME than estuarine stocks (Morris et al. 2015). The number of dolphins from the Southern Migratory Coastal Stock that died in this event is unknown. An analysis of trends in abundance for common bottlenose dolphins coast-wide (New Jersey to Florida) indicated a statistically significant decline in population size between 2011 and 2016 (Garrison et al. 2017a). A weighted generalized linear model was used to evaluate trends in coast-wide population size based on aerial surveys conducted between 2002 and 2016 (see Population Size above for survey descriptions). The model included a linear term for survey year and an interaction term to test for a difference in slope between 2002–2011 and 2011–2016. Estimates were weighted by the inverse of their standard error to reduce the influence of less certain estimates. There was no significant trend in population size between 2002 and 2011; however, there was a statistically significant (p=0.0308) change in slope between 2011 and 2016, indicating a decline in population size. The coast-wide inverse variance weighted average estimate for coastal common bottlenose dolphins during 2011 was 41,456 (CV=0.30) while the estimate during 2016 was 19,470 (CV=0.23; Garrison et al. 2017a). It is possible that this apparent decline in common bottlenose dolphin abundance in coastal waters along the eastern seaboard is a result of the 2013–2015 UME. An assessment of the impacts of the 2013–2015 UME on common bottlenose dolphin stocks in the wNA is ongoing. Finally, a UME was declared in South Carolina during February–May 2011. Six strandings assigned to the Southern Migratory Coastal Stock were considered to be part of the UME. The cause of this UME was undetermined.

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

<table>
<thead>
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<th>State</th>
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<tr>
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<td>CBD</td>
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<td>7</td>
<td>2</td>
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</table>
Table 3. Strandings of common bottlenose dolphins during 2011–2015 from Maryland to Florida that were ascribed to the Southern Migratory Coastal Stock, as well as number of strandings for which evidence of human interaction (HI) was detected and number of strandings for which it could not be determined (CBD) if there was evidence of HI. Assignments to stock were based upon the understanding of the seasonal movements of this stock; however, there is likely overlap with other stocks throughout the year. Data are from the NOAA National Marine Mammal Health and Stranding Response Database (unpublished data, accessed 18 May 2016 (SER) and 6 June 2016 (GAR)). Please note HI does not necessarily mean the interaction caused the animal’s death.

<table>
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<td>(Jan–Feb)</td>
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<td>104</td>
<td>86</td>
<td>513</td>
<td>144</td>
<td>118</td>
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Strandings from Virginia and Maryland were ascribed to stock based upon location and time of year with most occurring between May and September that could be ascribed to the Southern Migratory Coastal Stock. Some of these strandings could also be ascribed to the Northern Migratory Coastal Stock or NNCES Stock.

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

HABITAT ISSUES

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

Common bottlenose dolphins in the western North Atlantic are not listed as threatened or endangered under the Endangered Species Act, but the Southern Migratory Coastal Stock is a strategic stock due to its designation as depleted under the MMPA. From 1995 to 2001, NMFS recognized only the western North Atlantic Coastal Stock of common bottlenose dolphins in the western North Atlantic, and this stock was listed as depleted as a result of a UME in 1988–1989 (64 FR 17789, April 6, 1993). The stock structure was revised in 2008, 2009, and 2010, to recognize resident estuarine stocks and migratory and resident coastal stocks. The Southern Migratory Coastal Stock retains the depleted designation as a result of its origin from the western North Atlantic Coastal Stock. This stock is presumed to be below OSP due to its designation as depleted. PBR for the Southern Migratory Coastal Stock is 2423 and so the zero mortality rate goal, 10% of PBR, is 2.42-3. The documented mean annual human-caused mortality for this stock for 2014-2018 ranged between a minimum of 0 and a maximum of 18.344.3. However, these estimates are biased low for the following reasons: 1) the total U.S. human-caused mortality and serious injury for the Southern Migratory Coastal Stock cannot be directly estimated because of the spatial overlap of this stock with several other stocks of common bottlenose dolphins resulting in uncertainty in the stock assignment of takes, 2) there are several commercial fisheries operating within this stock’s boundaries that have little to no observer coverage, and 3) this mortality estimate incorporates a count of verified human-caused deaths and serious injuries and should be considered a minimum (NMFS 2016). Given these biases and uncertainties, there is insufficient information to determine whether or not the total fishery-related mortality and serious injury is approaching a zero mortality and serious injury rate. The impacts of two large UMEs on the status of this stock are unknown. Although there was no statistically significant difference in abundance for this stock between the 2010–2011 and 2016 surveys, a statistically significant decline in population size of all common bottlenose dolphins in coastal waters from New Jersey to Florida between 2010–2011 and 2016 was detected (Garrison et al. 2017a), concurrent with a large UME in the area; however, there is limited power to evaluate trends given uncertainty in stock distribution, lack of precision in abundance estimates, and a limited number of surveys.

REFERENCES CITED


Garrison, L.P., A.A. Hohn and L.J. Hansen. 2017b. Seasonal movements of Atlantic common bottlenose dolphin stocks based on tag telemetry data. Southeast Fisheries Science Center, Protected Resources and Biodiversity Division, 75 Virginia Beach Dr., Miami, FL 33140. PRBD Contribution # PRBD-2017-027, XX pp.


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

STOCK DEFINITION AND GEOGRAPHIC RANGE

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

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

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

**POPULATION SIZE**

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

**Earlier abundance estimates (>8 years old)**

Read et al. (2003) provided the first abundance estimate of common bottlenose dolphins that occur within the estuarine portion of the NNCES Stock range. This estimate, 919 (CV=0.13, 95% CI: 730–1,190), was based on a July 2000 photo-ID mark-recapture survey of a portion of North Carolina waters inshore of the barrier islands. However, the portion of the stock that may have occurred in coastal waters (≤1 km from shore) was not accounted for in this survey. Aerial survey data from 2002 (Garrison et al. 2016) were therefore used to account for this portion of the stock in coastal waters. The abundance estimate for the NNCES Stock during 2000–2002 was the combined abundance from estuarine and coastal waters. This combined estimate was 1,387 (CV=0.17). Because the survey did not sample all of the estuarine waters where dolphins are known to occur, the estimate of abundance may be negatively biased. Positive bias may have been introduced through the aerial survey data because Southern Migratory Coastal Stock dolphins may have been present in the coastal strip.
A photo-ID mark-recapture study was conducted by Urian et al. (2013) in July 2006 by Urian et al. (2013) using similar methods to those in Read et al. (2003) and included estuarine waters of North Carolina from, and including, the Little River Inlet estuary (near the North Carolina/South Carolina border) to, and including, Pamlico Sound. This survey also included coastal waters up to Cape Hatteras extending up to 1 km from shore. In order to estimate the abundance for the NNCES Stock, only sightings north of 34°46' N in central Core Sound were used (Urian et al. 2013). The resulting abundance estimate was 950 animals (CV=0.23, 95% CI: 516–1,384) and included a correction for the proportion of dolphins in the population with non-distinct fins (Urian et al. 2013). Because the survey did not include estuarine waters of Albemarle or Currituck Sounds or more northern estuarine and coastal waters, it is likely that some portion of the NNCES Stock was outside of the boundaries of the survey. Thus, the 2006 abundance estimate was most likely negatively biased.

Recent surveys and abundance estimates

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

Minimum Population Estimate

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

Current Population Trend

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

CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

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

POTENTIAL BIOLOGICAL REMOVAL

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

Table 1. Best and minimum abundance estimates for the Northern North Carolina Estuarine System Stock of common bottlenose dolphins with Maximum Productivity Rate (Rmax), Recovery Factor (Fr) and PBR.

<table>
<thead>
<tr>
<th>Nbest</th>
<th>CV</th>
<th>Nmin</th>
<th>Fr</th>
<th>Rmax</th>
<th>PBR</th>
</tr>
</thead>
<tbody>
<tr>
<td>823</td>
<td>0.06</td>
<td>782</td>
<td>0.50</td>
<td>0.04</td>
<td>7.8</td>
</tr>
</tbody>
</table>

ANNUAL HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

The total annual human-caused mortality and serious injury for the NNCES Stock during 2014–2015 is unknown. The mean annual fishery-related mortality and serious injury for observed fisheries, for strandings, and for at-sea observations identified as fishery-related ranged between 0.27 and 47.6/29.8. Additional mean annual
mortality and serious injury due to other human-caused sources (fishery research, at-sea entanglements in debris, unidentified gear) was 0.60. The minimum total mean annual human-caused mortality and serious injury for this stock during 2011–2014 and 2015 therefore ranged between 0.87 and 18.230 (Tables 12a, 12b and 12c). This range reflects several sources of uncertainty and is a minimum because 1) not all fisheries that could interact with this stock are observed and/or observer coverage is very low, 2) stranding data are used as an indicator of fishery-related interactions and not all dead animals are detected or recovered by the stranding network (Peltier et al. 2012; Wells et al. 2015), 3) cause of death is not (or cannot be) routinely determined for stranded carcasses, 4) the estimate includes an actual count of verified human-caused deaths and serious injuries and should be considered a minimum (NMFS 2016), and 5) the spatiotemporal overlap between the NNCES Stock and other common bottlenose dolphin stocks introduces uncertainty in assignment of mortalities to stock. In the sections below, dolphin mortalities were assigned to a stock or stocks by comparing the time and geographic location of the mortality to the stock boundaries and geographic range delimited for each stock (Lyssikatos and Garrison 2018).

Fishery Information

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

Note: Animals reported in the sections to follow were ascribed to a stock or stocks of origin following methods described in Maze-Foley et al. (2019). These include strandings, observed takes (through an observer program), fisherman self-reported takes (through the Marine Mammal Authorization Program), research takes, and opportunistic at-sea observations. For several cases in this report, the stock identity of a stranded animal was supported via sighting history in the Mid-Atlantic Bottlenose Dolphin Catalog (Urian in review).

Mid-Atlantic Gillnet

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

Mortality estimation for this stock is difficult because 1) observed takes are statistically rare events, 2) the NNCES, Northern Migratory Coastal, Southern Migratory Coastal, and SNCES common bottlenose dolphin stocks overlap in coastal waters of North Carolina at different times of the year, and therefore it is not always possible to definitively assign every observed mortality, or extrapolated bycatch estimate, to a specific stock, and 3) the low levels of federal observer coverage in state waters are likely insufficient to consistently detect rare bycatch events (Lyssikatos and Garrison 2018). To help address the first problem, two different analytical approaches were used to estimate common bottlenose dolphin bycatch rates during the period 2014–2018: 1) a simple annual ratio estimator of catch per unit effort (CPUE = observed catch/observed effort) per year based directly upon the observed data; and 2) a pooled CPUE approach (where all observer data from the most recent 5 years were combined into one sample to
During the most recent five-year time period, 2011–2014–2015–2016–2017, the combined average Northeast (NEFOP) and Southeast (SEFOP) Fisheries Observer Program observer coverage (measured in trips) for this fishery was 2.67 ± 5.16% in state waters (0–3 miles from shore) and 5.46 ± 6.95% in federal waters (3–200 miles from shore) (Lyssikatos in review and Garrison 2018). During this timeframe, two/three mortalities and one non-serious injury two cases where it could not be determined whether the animal was seriously injured were observed (Lyssikatos in review; Lyssikatos and Garrison 2018). In February 2017 and again in May 2018, the NEFOP observed an animal entangled in a small-mesh gillnet off the coast of North Carolina that was released alive but it could not be determined whether the animal was seriously injured and therefore, it was not included in the bycatch estimate. The entangled animal from 2018 was ascribed to the NNces Stock, and the animal from 2017 was ascribed to the NNces and Northern Migratory Coastal stocks. In July 2017, one mortality was observed by the NEFOP off North Carolina entangled in a small-mesh gillnet and was ascribed to the NNces and Southern Migratory Coastal stocks. In January 2015, one mortality was observed by the NEFOP off Hatteras, North Carolina, entangled in a medium-mesh gillnet within 0.23 km of shore and was ascribed to the NNces and Northern Migratory Coastal stocks (Lyssikatos and Garrison 2018). The animal was also self-reported by the fisherman per the Marine Mammal Authorization Program. The second/third mortality was observed by the SEFOP off the coast of northern North Carolina in September 2014, and this animal was ascribed to the NNces and Southern Migratory Coastal stocks (Lyssikatos and Garrison 2018). The animal was observed entangled in a small-mesh gillnet. In February 2013, the NEFOP observed an animal entangled in a small-mesh gillnet off the coast of North Carolina that was released alive without serious injury, and, therefore, not included in the bycatch estimate (Wenzel et al. 2015). This animal was ascribed to the NNces and Northern Migratory Coastal stocks. The most recent five-year mean minimum and maximum mortality estimates (2011–2014–2018–2015) were 6.6 (CV=0.32) and 28.2 (CV=0.15) animals per year, respectively (Table 12a; Lyssikatos in review and Garrison 2018).

However, based on documented serious injury and mortality in this fishery from both federal observer coverage and other data sources, the mean annual minimum mortality is likely not zero. Historical stranding data have documented multiple cases of dead, stranded dolphins recovered with gillnet gear attached (Byrd et al. 2014; Waring et al. 2015). During 2011–2014–2018–2015, stranding data documented two mortalities entangled in a single medium-mesh gillnet off of North Carolina, and these animals were ascribed to the NNces and Northern Migratory Coastal stocks (these animals were also self-reported by the fisherman per the Marine Mammal Authorization Program). One mortality that was recovered in Roanoke Sound with medium mesh gillnet gear entangled around its rostrum and flipper. The gear entangled around its flipper was attributed to the North Carolina inshore gillnet fishery and gear entangled around the rostrum was attributed to the mid-Atlantic gillnet fishery. This mortality is therefore reported in both fishery sections. The mortality is included within the annual human-caused mortality and serious injury total for the North Carolina inshore gillnet fishery (Table 1b). It may also be accounted for in the observer-based fishery bycatch estimate for the mid-Atlantic gillnet fishery. Four Eight other dead, stranded common bottlenose dolphins were recovered with markings indicative of interaction with gillnet gear (Read and Murray 2000), but no gear was attached to the carcasses and it is unknown whether the interactions with the gear contributed to the death of these animals. One of the four eight cases was ascribed to the NNces Stock alone, one was four were ascribed to both the NNces and Southern Migratory Coastal stocks, and two/three cases were ascribed to both the NNces and Northern Migratory Coastal stocks multiple stocks including the Northern and Southern Migratory Coastal stocks and NNces Stock. Overall, the low level of observer coverage, rarity of observed takes, and the inability to definitively assign each observed take to stock are sources of uncertainty in the bycatch estimates for this fishery (Lyssikatos and Garrison 2018).

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

North Carolina Inshore Gillnet

During 2014–2018, two mortality dead dolphin strandings ascribed to the NNCES Stock were observed in inshore waters with markings indicative of interaction with gillnet gear (Read and Murray 2000). Observers from the North Carolina Division of Marine Fisheries (NCDMF) recorded this incident in November 2017 (McConnaughey et al. 2019). This animal was recovered during 2011 in Roanoke Sound with two different types of medium-mesh gillnet gear entangled around its rostrum and flipper. The gear entangled around its flipper was attributed to the North Carolina inshore gillnet fishery, and gear entangled around the rostrum was attributed to the mid-Atlantic gillnet fishery. This mortality is therefore reported in both fishery sections. The mortality is included within the annual human-caused mortality and serious injury total for the North Carolina inshore gillnet fishery (Table 12b). It may also be accounted for in the observer-based fishery bycatch estimate for the mid-Atlantic gillnet fishery. This mortality was included in the stranding database and in the stranding totals presented in Table 1 (Northeast Regional Marine Mammal Stranding Network; NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 6 June 2016). No estimate of bycatch mortality is available for this fishery and the documented interaction in commercial gear represents a minimum known count of interactions with this fishery in the last five years. Five other dead, stranded common bottlenose dolphins were recovered in inshore waters with markings indicative of interaction with gillnet gear (Read and Murray 2000), but no gear was attached to the carcasses and it is unknown whether the interactions with the gear contributed to the death of these animals. Four of the five cases were ascribed to the NNCES Stock alone (one case supported by Urian in review), and one case was ascribed to both the NNCES and SNCES stocks.

Previously, current information on interactions between common bottlenose dolphins and the North Carolina inshore gillnet fishery was based solely on stranding data as no bycatch had been observed by state and federal observer programs. There was limited federal observer coverage (0.28%) of this fishery from May 2010 through March 2012, when NMFS observed this fishery. No common bottlenose dolphin bycatch was recorded. However, the low level of federal observer coverage in internal waters where the NNCES Stock largely resides is likely insufficient to detect bycatch events of common bottlenose dolphins if they were to occur in the inshore commercial gillnet fishery.

The North Carolina Division of Marine Fisheries (NCDMF) has operated their own observer program since 2000 due to sea turtle bycatch in inshore gillnets. The NCDMF applied for and obtained an Incidental Take Permit (ITP) in September 2013 that covers gillnet fisheries in all internal state waters. This ITP requires monitoring of gillnets statewide in internal waters with at least 7% observer coverage of large-mesh nets during spring, summer, and fall, and at least 1% observer coverage of small mesh nets during the same seasons (U.S. Dept. of Commerce 2013, Notice of permit issuance, Fed. Register 78: 57132–57133). In November 2017 NCDMF observers recorded their first bycatch event of a common bottlenose dolphin since they began monitoring in 2000 (McConnaughey et al. 2019). No common bottlenose dolphin bycatch was recorded by NCDMF during 2018 (McConnaughey et al. 2019; Byrd et al. 2020). No bycatch of common bottlenose dolphins had been recorded by state observers since they began monitoring in 2000.

North Carolina Long Haul Seine

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

Mid-Atlantic Haul/Beach Seine

During 2014–2018, stranding data documented one serious injury involving a common bottlenose dolphin and the mid-Atlantic haul/beach seine fishery in Virginia (Maze-Foley and Garrison 2017). The animal was ascribed to the Northern and Southern Migratory Coastal and NNCE Stock. The serious injury occurred during
October 2014, and is included in the annual human-caused mortality and serious injury total for this stock (Table 42b) as well as in the stranding database and in the stranding totals presented in Table 3 (Northeast Regional Marine Mammal Stranding Network; NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 13 August 2019). The mid-Atlantic haul/beach seine fishery had limited observer coverage by the NEFOP in 2010–2011. No observer coverage was allocated to this fishery during 2012–2014. No estimate of bycatch mortality is available for this fishery, and the documented interaction in this commercial gear represents a minimum known count of interactions in the last five years.

Virginia Pound Net

During 2011–2014–2018–2015, there were no documented mortalities or serious injuries in pound net gear of common bottlenose dolphins that could be ascribed to the NNCES Stock (Northeast Regional Marine Mammal Stranding Network; NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 13 August 2019). During 2015, two dolphin mortalities were and it was included in the stranding database and in the stranding totals presented in Table 3. Because there is no systematic observer program for the Virginia pound net fishery, no estimate of bycatch mortality is available. The overall impact of the Virginia Pound Net fishery on the NNCES Stock is unknown due to limited information on the extent to which the stock occurs within waters inside the mouth of the Chesapeake Bay. An amendment to the BDTRP was implemented in 2015 requiring gear restrictions for VA pound nets in estuarine and coastal state waters of Virginia to reduce bycatch (https://www.fisheries.noaa.gov/national/marine-mammal-protection/bottlenose-dolphin-take-reduction-plan).

North Carolina Roe Mullet Stop Net

During 2011–2014–2018–2015, there were no stranding data documented mortalities or serious injuries of common bottlenose dolphins in stop net gear. One mortality in which a common bottlenose dolphin was found entangled and dead in a stop net (this animal was also self-reported by the fisherman per the Marine Mammal Authorization Program). This mortality occurred during November 2013, and the animal was ascribed to the NNCES and Southern Migratory Coastal stocks. Both strandings were ascribed to multiple stocks: the Northern and Southern Migratory Coastal and NNSES stocks. Both mortalities were and it was included in the annual human-caused mortality and serious injury total for this stock (Table 1b). In addition, in 2015 a dead dolphin with line markings indicative of interaction with stop net gear was recovered ~300 yards from a stop net, but it is unknown whether the interaction with gear contributed to the death of this animal, and this case is therefore not included in the annual human-caused mortality and serious injury total for this stock. This animal was ascribed to the NNSES, SNCES, and Southern Migratory Coastal stocks. Both mortalities were and it was included in the stranding database and in the stranding totals presented in Table 3. Because there is no systematic observer program for the Virginia pound net fishery, no estimate of bycatch mortality is available. The overall impact of the Virginia Pound Net fishery on the NNSES Stock is unknown due to limited information on the extent to which the stock occurs within waters inside the mouth of the Chesapeake Bay. An amendment to the BDTRP was implemented in 2015 requiring gear restrictions for VA pound nets in estuarine and coastal state waters of Virginia to reduce bycatch (https://www.fisheries.noaa.gov/national/marine-mammal-protection/bottlenose-dolphin-take-reduction-plan).

Atlantic Blue Crab Trap/Pot

During 2011–2014–2018–2015, stranding data documented seven cases of common bottlenose dolphins entangled in trap/pot gear that could be ascribed to the NNCES Stock. Two cases were mortalities, and two were serious injuries, and for the remaining case, it could not be determined whether the animal was seriously injured. One mortality occurred during 2016 in unidentified trap/pot gear, and the other mortality occurred during 2015 in commercial blue crab trap/pot gear. Both of the mortalities were ascribed to the NNSES and Southern Migratory Coastal stocks. One serious injury occurred in 2018 in commercial blue crab trap/pot gear, and was ascribed solely to the NNSES Stock. Two additional One serious
injuries occurred in 2014 in commercial blue crab trap/pot gear and one occurred in 2015 in unidentified trap/pot gear. Both of these All three cases were ascribed to the Southern Migratory Coastal and NNCES stocks. The remaining serious injury occurred in 2017 in commercial blue crab trap/pot gear, and was ascribed to the Northern and Southern Migratory Coastal and NNCES stocks. The two mortalities and four serious injuries are included in the annual human-caused mortality and serious injury total for this stock (Table 1 2b). In addition, during 2017 an animal was disentangled and released alive from commercial blue crab trap/pot gear, but it could not be determined whether the animal was seriously injured. During 2018, an animal was disentangled from unidentified trap/pot gear, released alive, and considered not seriously injured following the disentanglement. Both of these animals were ascribed to the NNCES, Northern Migratory Coastal and Southern Migratory Coastal stocks. All of the cases These animals were included in the stranding database and in the stranding totals presented in Table 3 (Northeast Regional (NER) Marine Mammal Stranding Network; Southeast Regional (SER) Marine Mammal Stranding Network; NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 21 May 2019 18 May 2016 (SER) and 13 August 2019 6 June 2016 (NER)). Details regarding the serious injury determinations can be found in Maze-Foley and Garrison (2020). Because there is no observer program, it is not possible to estimate the total number of mortalities associated with crab traps/pots. However, stranding data indicate that interactions with trap/pot gear occur at some unknown level in North Carolina (Byrd et al. 2014) and other regions of the southeast U.S. (Noke and Odell 2002; Burdett and McFee 2004).

North Carolina Pound Net

During 2011–2014–20182015, there were no documented mortalities or serious injuries in North Carolina pound net gear of common bottlenose dolphins that could be ascribed to the NNCES Stock (NorSoutheast Regional Marine Mammal Stranding Network; NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 21 May 2019 6 June 2016). The North Carolina pound net fishery is included within the Category III U.S. mid-Atlantic mixed species stop seine/weir/pound net fishery. The pound net is a common fishing gear used in portions of North Carolina’s estuarine waters. However, the level of interaction with common bottlenose dolphins is unknown. Between 1997 and 2018, there has only been one documented mortality (2008) in North Carolina pound net gear, and this came from stranding data (Byrd et al. 2014). Because there is no systematic observer program, it is not possible to estimate the total number of interactions or mortalities associated with this commercial gear.

Hook and Line (Rod and Reel)

During 2011–2014–20182015, stranding data included two mortalities that could be ascribed to the NNCES Stock for which hook and line gear entanglement or ingestion were documented. For one of the Both mortalities occurred in 2016, and for both, the stranding data suggested the hook and line gear interaction was not a contributing factor to cause of death (Maze-Foley et al. 2019) (2012, North Carolina). For one mortality, it could not be determined whether the hook and line gear interaction contributed to cause of death (2011, Virginia). One mortality was ascribed to the NNCES and Southern Migratory Coastal stocks, and the other was ascribed to the NNCES, Northern Migratory Coastal and Southern Migratory Coastal stocks (Urian in review assigned solely to the NNCES Stock). None of these mortalities is included in the annual human-caused mortality and serious injury total for this stock (Table 1 2b).

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

Other Mortality

Historically, there have been occasional mortalities of common bottlenose dolphins during research activities (Waring et al. 2016); however, none were documented during 2014–2018 that were ascribed to the NNCES Stock. During 2015, a live animal was documented entangled in a sport toy flying ring (e.g., Aerobie or similar flying ring), and this animal was considered seriously injured (Maze-Foley and Garrison in 2020). This animal was ascribed to the NNCES Stock alone (supported by Urian in review), and it is included in the annual human-caused mortality and serious injury total for this stock (Table 2c). This animal was also included within the stranding database and in the stranding totals presented in Table 3 (Southeast Regional Marine Mammal Stranding Network; NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 21 May 2019 (SER)).

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

In addition to animals included in the stranding database, during 2011–2014 and 2018–2015, there were one at-sea observations in the NNCES Stock area of a live common bottlenose dolphins entangled in unidentified line/fishing gear in two cases and one entangled in a sport toy flying ring (e.g., Aerobie or similar flying ring) in the third case. This observation occurred in 2014, and it could not be determined if the animal was seriously injured (Maze-Foley and Garrison 2020). This animal was ascribed to the NNCES and SNCES stocks. One observation occurred during 2014 and one during 2015, and both of these animals were considered seriously injured. Both were ascribed to the NNCES Stock alone and are included in the annual human-caused mortality and serious injury total for this stock (Table 1c). The remaining observation occurred during 2014 and it could not be determined if the animal was seriously injured (see Maze-Foley and Garrison 2017 for details on serious injury determinations). The 2014 observation was ascribed to the NNCES and SNCES stocks.

All mortalities and serious injuries from known sources for the NNCES Stock are summarized in Tables 1a, 1b, and 1c.

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

<table>
<thead>
<tr>
<th>Fishery</th>
<th>Years</th>
<th>Data Type</th>
<th>Observer Coverage</th>
<th>Observed Serious Injury</th>
<th>Observed Mortality</th>
<th>Mean Annual Estimated Mortality and Serious Injury (CV) Based on Observer Data</th>
</tr>
</thead>
<tbody>
<tr>
<td>Mid-Atlantic Gillnet</td>
<td>2011–2015, 2014–2018</td>
<td>Obs. Data Logbook</td>
<td>2.0, 2.6, 3.1, 3.6, 5.6, 9.8, 7.0, 6.4</td>
<td>0, 0, 0, 0, 1, 0, 1</td>
<td>0, 0, 0, 1, 0</td>
<td>Min=0.6 (0.32) Max=16.428.2 (0.1522)</td>
</tr>
</tbody>
</table>
Mean Annual Mortality due to the observed mid-Atlantic gillnet commercial fishery
(2011–2015\textup{2014–2018})

Min=0.6 (0.32)
Max=16.428.2 (0.1522)

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

<table>
<thead>
<tr>
<th>Fishery</th>
<th>Years</th>
<th>Data Type</th>
<th>5-year Count Based on Stranding Data</th>
</tr>
</thead>
<tbody>
<tr>
<td>Chesapeake Bay Inshore Gillnet</td>
<td>2014–2018</td>
<td>Limited Observer and Stranding Data</td>
<td>0</td>
</tr>
<tr>
<td>North Carolina Long Haul Seine</td>
<td>2011–2015\textup{2014–2018}</td>
<td>Stranding Data</td>
<td>0</td>
</tr>
<tr>
<td>Mid-Atlantic Haul/Beach Seine</td>
<td>2011–2015\textup{2014–2018}</td>
<td>Limited Observer and Stranding Data</td>
<td>Min=0\textup{Max}=1</td>
</tr>
<tr>
<td>Virginia Pound Net</td>
<td>2011–2015\textup{2014–2018}</td>
<td>Stranding Data</td>
<td>0</td>
</tr>
<tr>
<td>North Carolina Roe Mullet Stop</td>
<td>2011–2015\textup{2014–2018}</td>
<td>Stranding Data</td>
<td>Min=0\textup{Max}=10</td>
</tr>
</tbody>
</table>
Table 2c. Summary of the incidental mortality and serious injury of common bottlenose dolphins of the Northern North Carolina Estuarine System Stock during 2014–2018 from all sources, including observed commercial fisheries, unobserved commercial fisheries, and research and other takes. See the Annual Human-Caused Mortality and Serious Injury section for biases and limitations of mortality estimates.

<table>
<thead>
<tr>
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</tr>
</thead>
<tbody>
<tr>
<td>Atlantic Blue Crab</td>
<td>2011–2015</td>
<td>2014–2018</td>
<td>Stranding Data</td>
<td>Min=0</td>
<td>Max=6</td>
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<tr>
<td>Trap/Pot</td>
<td></td>
<td></td>
<td></td>
<td></td>
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<tr>
<td>Hook and Line*</td>
<td>2011–2015</td>
<td>2014–2018</td>
<td>Stranding Data</td>
<td>0</td>
<td></td>
</tr>
</tbody>
</table>

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

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

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

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

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

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

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

Table 1c. Summary of the incidental mortality and serious injury of common bottlenose dolphins of the Northern North Carolina Estuarine System Stock during 2011–2015 from all sources, including observed commercial fisheries, unobserved commercial fisheries, and research and other takes. See the Annual Human-Caused Mortality and Serious Injury section for biases and limitations of mortality estimates.
Mean Annual Mortality due to the observed commercial mid-Atlantic gillnet fishery (2011–2015 2014–2018) (Table 2a)  
Min=0.66 (0.32)  
Max=16.4—(0.22) 28.2 (0.15)

Mean Annual Mortality due to unobserved commercial fisheries (2011–2015 2014–2018) (Table 2b)  
Min=0.2 0.4  
Max=1.21.6

Research Takes (5-year Min/Max Count)  
10

Other takes (5-year Min/Max Count)  
21

Mean Annual Mortality due to research and other takes (2011–2015 2014–2018)  
0.6 0.2

Min=0.87.2  
Max=18.230.0

Strandings

Between 2011–2014 and 2018–2015, 895,476 common bottlenose dolphins stranded along coastal and estuarine waters of North Carolina, Virginia, and Maryland that could be assigned to the NNCES Stock (Table 3; Northeast Regional Marine Mammal Stranding Network, Southeast Regional Marine Mammal Stranding Network; NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 21 May 2019 48 May 2016 (SER) and 13 August 2019 6 June 2016 (NER); Maze-Foley et al. 2019). There was evidence of human interaction for 66 of these strandings (Table 3). No evidence of human interaction was detected for 83 strandings, and for the remaining 331 strandings, it could not be determined if there was evidence of human interaction. It could not be determined if there was evidence of human interaction (HI) for 65 of these strandings, and for 187 strandings it was determined there was no evidence of human interaction. The remaining 736 showed evidence of human interactions (Table 2). Wells et al. (2015) estimated only one-third of common bottlenose dolphin carcasses in estuarine environments are recovered. In most cases, it was not possible to determine if an human interaction HI had occurred due to the decomposed state of the stranded animal. Of the 4217 (of 456,144) estuarine strandings positive for human interaction HI, seven11 (88.5%) of them exhibited evidence of fisheries entanglement (e.g., entanglement lesions, attached gear), and two were incidental takes from research gillnet gear (described above). Evidence of human interaction does not indicate cause of death, but rather only that there was evidence of interaction with a fishery (e.g., line marks, net marks) or evidence of a boat strike, gunshot wound, mutilation, etc., at some point. Stranding data probably underestimate the extent of human and fishery-related mortality and serious injury because not all of the dolphins that die or are seriously injured in human interactions wash ashore, or, if they do, they are not all recovered (Peltier et al. 2012; Wells et al. 2015). Additionally, not all carcasses will show evidence of human interaction, entanglement, or other fishery-related interaction due to decomposition, scavenger damage, etc. (Byrd et al. 2014). Finally, the level of technical expertise to recognize signs of human interaction varies among stranding network personnel.

The assignment of animals to a single stock is impossible in some seasons and regions where stocks overlap, particularly in coastal waters of North Carolina and Virginia, and estuarine waters near Beaufort Inlet (Maze-Foley et al. 2019). Of the 895,476 strandings ascribed to the NNCES Stock, 456,140 were ascribed solely to this stock. It is
likely, therefore, that the counts in Table 32 include some animals from the Southern Migratory Coastal, Northern Migratory Coastal, and SNCES stocks, and thereby overestimate the number of strandings for the NNCES Stock; those strandings that could not be definitively ascribed to the NNCES Stock were also included in the counts for these other stocks as appropriate. Stranded carcasses are not routinely identified to either the offshore or coastal morphotype of common bottlenose dolphin, therefore it is possible that some of the reported strandings were of the offshore form, though that number is likely to be low (Byrd et al. 2014).

This stock has also been impacted by two unusual mortality events (UMEs), one in 1987–1988 and one in 2013–2015, both of which have been attributed to morbillivirus epidemics (Lipscomb et al. 1994; Morris et al. 2015). Both UMEs included deaths of dolphins in spatiotemporal locations that apply to the NNCES Stock. When the impacts of the 1987–1988 UME were being assessed, only a single coastal stock of common bottlenose dolphin was thought to exist along the U.S. eastern seaboard from New York to Florida (Scott et al. 1988) and it was estimated that 10 to 50% of the coast-wide stock died as a result of this UME (Scott et al. 1988; Eguchi 2002). Impacts to the NNCES Stock alone are not known. However, Scott et al. (1988) indicated that the observed mortalities from this event affected primarily coastal dolphins. The total number of stranded common bottlenose dolphins from New York through North Florida (Brevard County) during the 2013–2015 UME was 1614–1827 (https://www.fisheries.noaa.gov/national/marine-life-distress/2013-2015-bottlenose-dolphin-unusual-mortality-event-mid-atlantic, accessed 13 November 2019). Most strandings and morbillivirus positive animals have been recovered from the ocean side beaches rather than from within the estuaries, again suggesting that coastal stocks may have been more impacted by this UME than estuarine stocks (Morris et al. 2015). However, the habitat of the NNCES stock includes more nearshore coastal waters (in winter) than many estuarine stocks and so it may have been more heavily impacted by this UME than other estuarine stocks. An assessment of the impacts of the 2013–2015 UME to common bottlenose dolphin stocks in the wNA is ongoing.

Table 32. Strandings of common bottlenose dolphins during 2014–2018 from North Carolina, Virginia, and Maryland that were ascribed to the Northern North Carolina Estuarine System (NNCES) Stock, including the number of strandings for which evidence of human interaction (HI) was detected and number of strandings for which it could not be determined (CBD) if there was evidence of HI. Strandings observed in North Carolina are separated into those occurring within the Pamlico Sound estuarine system (Estuary) vs. coastal waters. Assignments to stock were based upon the understanding of the seasonal movements of this stock. However, particularly in coastal waters, there is likely overlap between the NNCES Stock and other common bottlenose dolphin stocks. Data are from the NOAA National Marine Mammal Health and Stranding Response Database (unpublished data, accessed 21 May 2019 (SER) and 13 August 2019 (NER)). Please note HI does not necessarily mean the interaction caused the animal’s death.

<table>
<thead>
<tr>
<th>State</th>
<th>2014</th>
<th>2015</th>
<th>2016</th>
<th>2017</th>
<th>2018</th>
<th>Total</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>HI Yes</td>
<td>HI No</td>
<td>CBD</td>
<td>HI Yes</td>
<td>HI No</td>
<td>CBD</td>
</tr>
<tr>
<td>North Carolina - Estuary</td>
<td>2</td>
<td>3</td>
<td>35</td>
<td>4</td>
<td>3</td>
<td>18</td>
</tr>
<tr>
<td>North Carolina - Coastal</td>
<td>2</td>
<td>22</td>
<td>27</td>
<td>7</td>
<td>15</td>
<td>25</td>
</tr>
<tr>
<td>Virginia*</td>
<td>5</td>
<td>3</td>
<td>16</td>
<td>3</td>
<td>3</td>
<td>22</td>
</tr>
<tr>
<td>Maryland*</td>
<td>0</td>
<td>0</td>
<td>4</td>
<td>0</td>
<td>0</td>
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<tr>
<td>Total</td>
<td>119</td>
<td>100</td>
<td>89</td>
<td>74</td>
<td>98</td>
<td>480</td>
</tr>
</tbody>
</table>

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

<table>
<thead>
<tr>
<th>State</th>
<th>2011</th>
<th>2012</th>
<th>2013</th>
<th>2014</th>
<th>2015</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>Type</strong></td>
<td>HI</td>
<td>HI</td>
<td>HI</td>
<td>HI</td>
<td>HI</td>
</tr>
<tr>
<td><strong>North Carolina—Estuary</strong></td>
<td>HI</td>
<td>HI</td>
<td>HI</td>
<td>HI</td>
<td>HI</td>
</tr>
<tr>
<td>1*</td>
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<tr>
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<td>4</td>
<td>5</td>
<td>6</td>
<td>7</td>
<td>8</td>
</tr>
<tr>
<td><strong>North Carolina—Coastal</strong></td>
<td>HI</td>
<td>HI</td>
<td>HI</td>
<td>HI</td>
<td>HI</td>
</tr>
<tr>
<td>3*</td>
<td>12</td>
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<td>15</td>
<td>12</td>
<td>12</td>
</tr>
<tr>
<td>4*</td>
<td>9</td>
<td>14</td>
<td>21</td>
<td>22</td>
<td>15</td>
</tr>
<tr>
<td><strong>Virginia</strong></td>
<td>HI</td>
<td>HI</td>
<td>HI</td>
<td>HI</td>
<td>HI</td>
</tr>
<tr>
<td>5*</td>
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<td>1</td>
<td>6</td>
<td>4</td>
<td>3</td>
<td>0</td>
</tr>
<tr>
<td><strong>Maryland</strong></td>
<td>HI</td>
<td>HI</td>
<td>HI</td>
<td>HI</td>
<td>HI</td>
</tr>
<tr>
<td>0*</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
</tr>
<tr>
<td><strong>Annual Total</strong></td>
<td>8</td>
<td>8</td>
<td>5</td>
<td>11</td>
<td>9</td>
</tr>
</tbody>
</table>
Includes 1 FI, an entanglement interaction with commercial gillnet gear (mortality, North Carolina inshore gillnet fishery).

Includes 2 entanglement interactions in research sink gillnet gear (1 mortality; 1 released alive, could not be determined if seriously injured) and 1 FI.

Includes 2 FIs.

Includes 1 FI.

Includes 2 FIs, 1 of which had markings indicative of interactions with gillnet gear (mortality).

Includes 4 FIs.

Includes 9 FIs, 1 of which involved ingestion of hook and line gear (mortality), and 3 of which had markings indicative of interactions with gillnet gear (mortalities).

Includes 4 FIs, 1 of which was an entanglement in a stop net (mortality, North Carolina roe mullet stop net fishery). Also includes 2 mortalities with evidence of a boat strike.

Includes 2 FIs, 1 of which had markings indicative of interactions with gillnet gear (mortality).

Includes 6 FIs, 1 of which had markings indicative of an entanglement in a stop net (mortality, North Carolina roe mullet stop net fishery).

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

Includes 5 FIs, 1 of which was an entanglement in hook and line gear (mortality). Two FIs were mortalities with twisted twine markings indicative of interaction with Virginia pound net gear.

A mortality with evidence of a boat strike.

Includes 7 FIs.

Includes 3 FIs. One animal was released alive seriously injured following entanglement in commercial crab trap/pot gear. Another animal was released alive seriously injured following capture in a haul seine. Also includes 1 mortality with evidence of a boat strike.

Includes 2 FIs. One FI was an entanglement in commercial blue crab trap/pot gear (mortality), and the other was an entanglement in unidentified trap/pot gear (released alive seriously injured).

Includes 1 FI, an entanglement interaction with commercial gillnet gear (mortality, North Carolina inshore gillnet fishery).

HABITAT ISSUES

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

STATUS OF STOCK

Common bottlenose dolphins in the western North Atlantic are not listed as threatened or endangered under the Endangered Species Act. However, this stock is considered strategic under the MMPA. PBR for the NNCES Stock is 7.8 and so the zero mortality rate goal, 10% of PBR, is 0.8. The documented mean annual human-caused mortality for this stock for 2011-2014–2015 ranged between a minimum of 0.872 and a maximum of 18.230.0. However, these estimates are biased low for the following reasons: 1) the total U.S. human-caused mortality and serious injury for this stock cannot be directly estimated because of the spatial overlap of several stocks of common bottlenose dolphins in North Carolina and Virginia resulting in uncertainty in the stock assignment of some takes, 2) there are several commercial fisheries operating within this stock’s boundaries that have little to no observer coverage, and 3) this
mortality estimate incorporates a count of verified human-caused deaths and serious injuries and should be considered a minimum (NMFS 2016). Given these uncertainties, and the fact that the maximum mean annual human-caused mortality and serious injury exceeds PBR, NMFS considers this stock strategic under the MMPA. The total fishery-related mortality and serious injury for this stock is not less than 10% of the calculated PBR and therefore, cannot be considered to be insignificant and approaching a zero mortality and serious injury rate. The impact of the 2013–2015 UME on the status of this stock is unknown. The status of this stock relative to OSP is unknown. There are insufficient data to determine the population trends for this stock.

REFERENCES CITED


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NMFS 2016. Guidelines for preparing stock assessment reports pursuant to the 1994 amendments to the MMPA. NMFS Instruction 02-204-01. 24 pp.


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

STOCK DEFINITION AND GEOGRAPHIC RANGE

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

The Southern North Carolina Estuarine System (SNCES) Stock is best defined as animals occupying estuarine and nearshore coastal waters (≤3 km from shore) between the Little River Inlet estuary (33.9°N), inclusive of the estuary (near the North Carolina/South Carolina border), and the New River (34.5°N) during cold water months (best defined as January and February). Members of this stock do not undertake large-scale migratory movements. Instead, they expand their range only slightly northward during warmer months into estuarine waters and nearshore waters (≤3 km from shore) of southern North Carolina as far as central Core Sound and southern Pamlico Sound (Garrison *et al.* 2017b) (Figure 1). These movements and the
range of this stock have been inferred from a combination of telemetry, photo-ID, and genetic data (Read et al. 2003; Rosel et al. 2009; Garrison et al. 2017b). Two animals tagged at Holden Beach, North Carolina, just south of Cape Fear during November 2004, remained within waters of southern and central North Carolina throughout the nine-month period their tags were operational (Garrison et al. 2017b). Eight animals tagged and/or freeze-branded near Beaufort, North Carolina, between 1995 and 2006 were documented, using long-term photo-ID studies, to have moved south and occupied estuarine and coastal waters near Cape Fear, south of the New River during cold water months (Garrison et al. 2017b). A photo-ID mark-recapture survey (Read et al. 2003) found little movement of marked animals between the northern portion of the survey area (northern Pamlico Sound, Roanoake Sound, Albemarle Sound, and Currituck Sound) and the southern portion (Southport, Cape Fear River, New River, and Bogue Sound). The authors suggested that movement patterns, differences in group sizes, and habitats are consistent with two stocks of animals occupying estuarine waters of North Carolina (Read et al. 2003). SNCES Stock animals have not been observed to move north of Cape Lookout in coastal waters nor into the main northern and central portion of Pamlico Sound during warm water months (Garrison et al. 2017b). Finally, genetic analysis of samples from animals in waters of southern North Carolina (including known SNCES animals based on live captures and strandings of unknown stock origin between Cape Lookout and the North Carolina/South Carolina border) demonstrated significant genetic differentiation from animals occupying waters from Virginia and further north and estuarine waters of South Carolina (Rosel et al. 2009).

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

**POPULATION SIZE**

The current population size of the SNCES Stock is unknown because the survey data are more than eight years old (Wade and Angliss 1997; Table 1).

**Earlier abundance estimates (>8 years old)**

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

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

**Recent surveys and abundance estimates**

Silva et al. (2020) performed photo-identification (photo-ID) capture-mark-recapture (CMR) surveys in summer
and winter 2014 within the estuarine waters of the SNCES stock and nearshore coastal waters. The estimated abundance in the winter survey, when the least amount of spatial overlap with other stocks is expected, was 206 (95% CI 100–423, CV=0.38). Each survey consisted of a single mark and recapture session and had low resight rates during the recapture session (five resights in summer, three in winter). Both surveys required extended periods of time to complete the original mark (15–20 days) and single recapture (10–30 days). In addition, the length of time between the end of the initial summer season mark and the start of the single recapture session was 19 days. These prolonged periods of time likely lead to violation of the assumption of population closure in CMR analysis as noted by the authors in particular for the summer estimate. For the winter survey, the authors note that the spatial coverage of the survey was reduced and that the distribution of the dolphins expanded outside of the survey area potentially resulting in a negative bias. Finally, the survey did not include multiple recapture sessions as suggested for CMR studies to be used for stock assessment reports (Rosel et al. 2011). Due to the potential bias and uncertainty associated with these estimates, the study results were not used to provide an estimate of abundance for the SNCES stock.

Minimum Population Estimate

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

Current Population Trend

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

CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

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

POTENTIAL BIOLOGICAL REMOVAL

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

<table>
<thead>
<tr>
<th>Nbest</th>
<th>Nbest CV</th>
<th>Nmin</th>
<th>Fr</th>
<th>Rmax</th>
<th>PBR</th>
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<tr>
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<td>-</td>
<td>Unknown</td>
<td>0.5</td>
<td>0.04</td>
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</table>

ANNUAL HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

The total annual human-caused mortality and serious injury for the SNCES Stock during 2011–2014–2018–2015 is unknown. The mean annual fishery-related mortality and serious injury estimated from observed fisheries and strandings identified as fishery-related was 0.4 ranged between 0.4 and 0.6. No additional mortality and serious injury was documented from other human-caused sources (e.g., fishery research) and therefore, the minimum total mean annual human-caused mortality and serious injury for this stock during 2011–2014–2018–2015 was 0.4 ranged between 0.4 and 0.6 (Tables 12a, 12b and 12c). This range estimate reflects several sources of uncertainty and is a minimum because 1) not all fisheries that could interact with this stock are observed and/or observer coverage is very low, 2) stranding data are used as an indicator of fishery-related interactions and not all dead animals are recovered by the stranding network (Peltier et al. 2012; Wells et al. 2015), 3) cause of death is not (or cannot be) routinely determined for stranded carcasses, 4) the estimate includes an actual count of verified human-caused deaths and serious injuries and should be considered a minimum (NMFS 2016), and 5) the spatiotemporal overlap between the SNCES Stock and other common bottlenose dolphin stocks introduces uncertainty in assignment of mortalities to stock. In the sections below, dolphin mortalities were assigned to a stock or stocks by comparing the time and geographic location of the mortality to the stock boundaries and geographic range delimited for each stock (Lyssikatos and Garrison 2018).
Fishery Information

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

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

Mid-Atlantic Gillnet

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

Mortality estimation for this stock is difficult because 1) observed takes are statistically rare events, 2) the Northern Migratory, Southern Migratory, NNCES, and SNCES common bottlenose dolphin stocks overlap in coastal waters off North Carolina and Virginia at different times of the year, and therefore it is not always possible to definitively assign every observed mortality, or extrapolated bycatch estimate, to a specific stock, and 3) the low levels of federal observer coverage in state waters are likely insufficient to consistently detect rare bycatch events (Lyssikatos and Garrison 2018). To help address the first problem, two different analytical approaches were used to estimate common bottlenose dolphin bycatch rates during the period 2011–2014–2015: 1) a simple annual ratio estimator of catch per unit effort (CPUE = observed catch/observed effort) per year based directly upon the observed data; and 2) a pooled CPUE approach (where all observer data from the most recent five years were combined into one sample to estimate CPUE) (Lyssikatos and Garrison 2018). In each case, the annual reported fishery effort (defined as a fishing trip) was multiplied by the estimated bycatch rate to develop annual estimates of fishery-related mortality. Next, the two model estimates (and the associated uncertainty) were averaged, in order to account for the uncertainty in the two approaches, to produce an estimate of the mean mortality of common bottlenose dolphins for this fishery (Lyssikatos and Garrison 2018). To help address the second problem, minimum and maximum mortality estimates were calculated per stock to indicate the range of uncertainty in assigning observed takes to stock (Lyssikatos and Garrison 2018). Uncertainties and potential biases are described in Lyssikatos and Garrison (2018).

During the most recent five-year reporting period, 2011–2014–2015, the combined average Northeast (NEFOP) and Southeast (SEFOP) Fisheries Observer Program observer coverage (measured in trips) for this fishery was 2.67–5.35% in state waters (0–3 miles from shore) and 5.36–9.95% in federal waters (3–200 miles from shore), respectively (Lyssikatos in review and Garrison 2018). This low level of observer coverage may result in small-sample bias in the bycatch estimate because the stock is small and PBR may be less than four (NMFS 2016; Lyssikatos and Garrison 2018). During this timeframe, no common bottlenose dolphin mortalities or serious injuries that could be attributed to the SNCES Stock were observed by the NEFOP or SEFOP. The most recent five-year mean minimum and maximum mortality estimates (2011–2014–2018) were, therefore, both unknown zero (Table 4.2; Lyssikatos...
Carolina. This animal was considered seriously injured (Maze-Foley and Garrison 2020) and was ascribed to the SNCES and Southern Migratory Coastal stocks. Because there is no systematic stranding database in which a common bottlenose dolphin was entangled in commercial blue crab trap/pot gear. The data, accessed 21 May 201918 May 2016). The most recent documented interaction was a 2009 mortality within the Southeast Regional Marine Mammal Stranding Network; NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 21 May 201918 May 2016). However, there were two one cases documented in which a carcass stranded with markings indicative of interaction with gillnet gear (Read and Murray 2000) but no gear was attached to the carcass and it is unknown whether the interactions with the gear contributed to the deaths of these animals. These cases occurred in 2012 and 2015; and both were ascribed to the SNCES and NNCES stocks. Neither of these mortalities are This mortality is not included in the annual human-caused mortality and serious injury total for this stock since bycatch estimates for this stock based on observer program data were zero (Table 12a). Overall, the low level of observer coverage, rarity of observed takes, and the inability to definitively assign each observed take to stock are sources of uncertainty in the bycatch estimates for this fishery.

**North Carolina Inshore Gillnet**

During 2011-2014–20182015, there were no documented mortalities or serious injuries involving inshore gillnet gear of common bottlenose dolphins that could be ascribed to the SNCES Stock (Southeast Regional Marine Mammal Stranding Network; NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 21 May 201918 May 2016). However, there were two one cases documented in which a carcass stranded with markings indicative of interaction with gillnet gear (Read and Murray 2000) but no gear was attached to the carcass and it is unknown whether the interactions with the gear contributed to the deaths of these animals. These cases occurred in 2012 and 2015; and both were ascribed to the SNCES and NNCES stocks. Neither of these mortalities are This mortality is not included in the annual human-caused mortality and serious injury total for this stock (Table 12b), but it is included in the stranding database and in the stranding totals presented in Table 3. In addition, during 2014–2018, there was one at-sea observation of a live common bottlenose dolphin entangled in gillnet gear (in 2018) which was ascribed to the SNCES Stock, and this animal was considered seriously injured (McConnaughey et al. 2020). This serious injury is included in the annual human-caused mortality and serious injury total for this stock (Table 2b).

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

**Atlantic Blue Crab Trap/Pot**

During 2011-2014–20182015, there were no documented mortalities or serious injuries in commercial blue crab trap/pot gear of common bottlenose dolphins that could be ascribed to the SNCES Stock (Southeast Regional Marine Mammal Stranding Network; NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 21 May 201918 May 2016). The most recent documented interaction was a 2009 mortality within the stranding database in which a common bottlenose dolphin was entangled in commercial blue crab trap/pot gear. The 2009 mortality was ascribed to the SNCES and Southern Migratory Coastal stocks. Because there is no systematic
observer program, it is not possible to estimate the total number of interactions or mortalities associated with crab traps/pots. However, stranding data indicate that interactions occur at some unknown level in North Carolina (Byrd et al. 2014) and other regions of the southeast U.S. (Nokoe and Odell 2002; Burdett and McFee 2004).

**North Carolina Long Haul Seine Fishery**

There have been no documented interactions between common bottlenose dolphins of the SNCES Stock and the North Carolina long haul seine fishery during 2011–2014. The fishery includes fishing with long haul seine gear to target any species in waters off North Carolina, including estuarine waters in Pamlico and Core Sounds and their tributaries. There has not been federal observer coverage of this fishery.

**North Carolina Roe Mullet Stop Net**

During 2011–2014, stranding data indicate that interactions occur at some unknown level in North Carolina (Byrd et al. 2014) and other regions of the southeast U.S. (Noke and Odell 2002; Burdett and McFee 2004).

**North Carolina Roe Mullet Stop Net**

During 2011–2014, stranding data indicate that interactions occur at some unknown level in North Carolina (Byrd et al. 2014) and other regions of the southeast U.S. (Noke and Odell 2002; Burdett and McFee 2004).

**Hook and Line (Rod and Reel)**

During 2011–2014, there were no documented mortalities or serious injuries of common bottlenose dolphins involving hook and line gear that could be ascribed to the SNCES Stock. However, a dead stranded dolphin with line markings indicative of interaction with stop net gear was recovered in October 2015. This animal was ascribed to the SNCES, NNCES, and Southern Migratory Costal stocks. This mortality is included in the stranding database and in the stranding totals presented in Table 32 (Southeast Regional Marine Mammal Stranding Network; NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 21 May 2019). No estimate of bycatch mortality is available for the stop net fishery. This fishery has not had regular, ongoing federal or state observer coverage. However, the NMFS Beaufort laboratory observed this fishery in 2001–2002 (Byrd and Hohn 2010), and Duke University observed the fishery in 2005–2006 (Thayer et al. 2007). Entangled dolphins were not documented during these formal observations, but two mortalities of dolphins due to entanglement in stop nets occurred in 1993 and 1999 and were documented by the stranding network in North Carolina (Byrd and Hohn 2010).

**Other Mortality**

Historically, there have been occasional mortalities of common bottlenose dolphins during research activities (Waring et al. 2016); however, none were documented during 2011–2014 that were ascribed to the SNCES Stock.

In addition to animals included in the stranding database and the at-sea observation mentioned above (under Hook and Line North Carolina Inshore Gillnet), during 2011–2014, there was one at-sea observation of a live common bottlenose dolphin entangled in unidentified line (in 2014). It could not be determined if this animal was seriously injured or not (Maze-Foley and Garrison 2017). This animal was ascribed to the SNCES stock alone and determined to have been seriously injured (Maze-Foley and Garrison in review). This serious injury was included in the annual human-caused mortality and serious injury total for this stock (Table 1b).

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

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

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

<table>
<thead>
<tr>
<th>Fishery</th>
<th>Years</th>
<th>Data Type</th>
<th>Observer Coverage</th>
<th>Observed Serious Injury</th>
<th>Observed Mortality</th>
<th>Mean Annual Estimated Mortality and Serious Injury (CV) Based on Observer Data</th>
</tr>
</thead>
<tbody>
<tr>
<td>Mid-Atlantic Gillnet</td>
<td>2011–2015</td>
<td>Obs. Data Logbook</td>
<td>2.0, 2.6, 3.1, 3.6, 5.6, 9.8, 7.7, 6.7</td>
<td>0, 0, 0, 0, 0</td>
<td>0, 0, 0, 0, 0</td>
<td>0Unknown</td>
</tr>
<tr>
<td></td>
<td>2014–2018</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>5-year Count Based on Stranding Data and Fisherman Self-Reported Takes via the Marine Mammal Authorization Program</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td>Min=1 Max=2.1</td>
</tr>
<tr>
<td>Mean Annual Mortality due to the observed mid-Atlantic gillnet commercial fishery (2011–2015, 2014–2018)</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td>Min=0.2 Max=0.4</td>
</tr>
</tbody>
</table>

Table 2b. Summary of the incidental mortality and serious injury of common bottlenose dolphins of the Southern North Carolina Estuarine System Stock during 2014–2018 from commercial fisheries that do not have ongoing, systematic federal observer programs. Counts of mortality and serious injury based on stranding data are given. Minimum and maximum values are reported in individual cells when there is uncertainty in the assignment of mortalities to this particular stock due to spatial overlap with other common bottlenose dolphin stocks in certain areas and seasons.

<table>
<thead>
<tr>
<th>Fishery</th>
<th>Years</th>
<th>Data Type</th>
<th>5-Year Count Based on Stranding Data and At-Sea Observations</th>
</tr>
</thead>
</table>

<table>
<thead>
<tr>
<th>Fishery</th>
<th>Years</th>
<th>Data Type</th>
<th>5-year Count Based on Stranding Data</th>
</tr>
</thead>
<tbody>
<tr>
<td>North Carolina Inshore Gillnet(^a)</td>
<td>2014–2018</td>
<td>Limited Federal and State Observers, Stranding Data, and At-Sea Observation</td>
<td>1</td>
</tr>
<tr>
<td>Atlantic Blue Crab Trap/Pot</td>
<td>2014–2018</td>
<td>Stranding Data</td>
<td>0</td>
</tr>
<tr>
<td>North Carolina Long Haul Seine</td>
<td>2014–2018</td>
<td>Stranding Data</td>
<td>0</td>
</tr>
<tr>
<td>North Carolina Roe Mullet Stop Net(^b)</td>
<td>2014–2018</td>
<td>Stranding Data</td>
<td>0</td>
</tr>
<tr>
<td>Hook and Line(^c)</td>
<td>2014–2018</td>
<td>Stranding Data</td>
<td>0</td>
</tr>
<tr>
<td>Mean Annual Mortality Due to Unobserved Commercial Fisheries (2014–2018)</td>
<td></td>
<td></td>
<td>0.2</td>
</tr>
</tbody>
</table>

\(^a\) North Carolina inshore gillnet interactions are included if the animal was found entangled in gillnet gear. Strandings with line markings indicative of interaction with gillnet gear are not included within the table. See "North Carolina Inshore Gillnet" text for more details.

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

\(^c\) Hook and line interactions are counted here if the available evidence suggested the hook and line gear contributed to the cause of death. See "Hook and Line" text for more details.
<table>
<thead>
<tr>
<th>Source</th>
<th>Period</th>
<th>Method</th>
<th>Number</th>
</tr>
</thead>
<tbody>
<tr>
<td>North Carolina Long-Haul Seine</td>
<td>2011–2015</td>
<td>Stranding Data</td>
<td>0</td>
</tr>
<tr>
<td>North Carolina Roe Mullet Stop Net</td>
<td>2011–2015</td>
<td>Stranding Data</td>
<td>0</td>
</tr>
<tr>
<td>Hook and Line</td>
<td>2011–2015</td>
<td>Stranding Data and At-Sea Observation</td>
<td>1</td>
</tr>
</tbody>
</table>

Mean Annual Mortality due to unobserved commercial fisheries (2011–2015) 0.2

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

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

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

Table 2c. Summary of the incidental mortality and serious injury of common bottlenose dolphins of the Southern North Carolina Estuarine System Stock during 2014–2018 from all sources, including observed commercial fisheries, unobserved commercial fisheries, and research and other takes. See the Annual Human-Caused Mortality and Serious Injury section for biases and limitations of mortality estimates.
Mean Annual Mortality due to unobserved commercial fisheries (2011–2015/2014–2018) (Table 12b)

<p>| | |</p>
<table>
<thead>
<tr>
<th></th>
<th></th>
</tr>
</thead>
<tbody>
<tr>
<td>Research Takes (5-year Min/Max Count)</td>
<td>0</td>
</tr>
<tr>
<td>Other takes (5-year Min/Max Count)</td>
<td>0</td>
</tr>
</tbody>
</table>

Mean Annual Mortality due to research and other takes (2011–2015/2014–2018)

<p>| | |</p>
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</thead>
</table>


|                             | Min=0.4 | Max=0.6 |

Strandings

Between 2014 and 2018, common bottlenose dolphins stranded along coastal and estuarine waters of North Carolina that could be ascribed to the SNCES Stock (Table 3; Southeast Regional Marine Mammal Stranding Network; NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed May 2019; Maze-Foley et al. 2019). There was evidence of human interaction for five of these strandings, all of which were fisheries interactions (Table 3). No evidence of human interaction was detected for the remaining 18 strandings, and it could not be determined if there was evidence of human interaction. It could not be determined if there was evidence of human interaction for 2618 of these strandings, and for 3720 it was determined there was no evidence of human interaction. The remaining 175 showed evidence of human interactions, all of which were including 16 fisheries interactions (FIs). One FI occurred in 2011 and involved a dolphin entangled in gillnet gear and reported to the stranding network, who recovered the carcass. The gillnet was targeting spot, and this take is included under the mid-Atlantic gillnet fishery (Table 1a). The remaining FIs could not be assigned to a specific fishery. It should be recognized that evidence of human interaction does not always indicate cause of death, but rather only that there was evidence of interaction with a fishery (e.g., line marks, net marks) or evidence of a boat strike, gunshot wound, mutilation, etc., at some point. Stranding data probably underestimate the extent of human and fishery-related mortality and serious injury because not all of the dolphins that die or are seriously injured in human interactions wash ashore, or, if they do, they are not all recovered (Peltier et al. 2012; Wells et al. 2015). Additionally, not all carcasses will show evidence of human interaction, entanglement, or other fishery-related interaction due to decomposition, scavenger damage, etc. (Byrd et al. 2014). Finally, the level of technical expertise to recognize signs of human interaction varies among stranding network personnel.

As described in the Stock Definition and Geographic Range section, there is spatiotemporal overlap between the SNCES Stock and the Southern Migratory Coastal Stock in coastal waters of southern North Carolina when the Southern Migratory Coastal Stock makes its seasonal migrations north and south. There is also overlap in waters from southern Pamlico Sound to Bogue Sound with the NNCES Stock during late summer and early fall. Therefore, assignment of animals to a single stock is impossible in some seasons and regions (Maze-Foley et al. 2019). Of the 8053 strandings ascribed to the SNCES Stock, 1211 were ascribed solely to this stock and two—one of those was identified as having evidence of both a fishery interaction and boat collision. It is likely that the counts in Table 3 include some animals from the Southern Migratory Coastal and/or NNCES Stock and therefore overestimate the number of strandings for the SNCES Stock; those strandings that could not be solely ascribed to the SNCES Stock were also included in the counts for these other stocks as appropriate. In addition, stranded carcasses are not routinely identified to either the offshore or coastal morphotype of common bottlenose dolphin. Therefore, it is possible that some of the reported strandings recorded along the coast were of the offshore form, although that number is likely to be low (Byrd et al. 2014).
This stock has been impacted by two unusual mortality events (UMEs), one in 1987–1988 and one in 2013–2015, both of which have been attributed to morbillivirus epidemics (Lipscomb et al. 1994; Morris et al. 2015). Both UMEs included deaths of dolphins in spatiotemporal locations that apply to the SNCES Stock. When the impacts of the 1987–1988 UME were being assessed, only a single coastal stock of common bottlenose dolphin was thought to exist along the U.S. eastern seaboard from New York to Florida (Scott et al. 1988) and it was estimated that 10 to 50% of the coast-wide stock died as a result of this UME (Scott et al. 1988; Eguchi 2002). Impacts to the SNCES Stock alone are not known. However, Scott et al. (1988) indicated that the observed mortalities from this event affected primarily coastal rather than estuarine dolphins. The total number of stranded common bottlenose dolphins from New York through North Florida (Brevard County) during the 2013–2015 UME was 1614–1827 (https://www.fisheries.noaa.gov/national/marine-life-distress/2013-2015-bottlenose-dolphin-unusual-mortality-event-mid-atlantic http://www.nmfs.noaa.gov/pr/health/mmume/midatldolphins2013.html, accessed 13 November 2016). Most strandings and morbillivirus positive animals have been recovered from the ocean side beaches rather than from within the estuaries, suggesting that coastal stocks may have been more impacted by this UME than estuarine stocks (Morris et al. 2015). However, the habitat of the SNCES Stock includes more nearshore coastal waters than many estuarine stocks and so it may have been more heavily impacted by this UME than other estuarine stocks. An assessment of the impacts of the 2013–2015 UME to common bottlenose dolphin stocks in the wNA is ongoing.

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

<table>
<thead>
<tr>
<th>State</th>
<th>2014</th>
<th>2015</th>
<th>2016</th>
<th>2017</th>
<th>2018</th>
<th>Total</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>HI</td>
<td>HI</td>
<td>CBD</td>
<td>HI</td>
<td>HI</td>
<td>CBD</td>
</tr>
<tr>
<td></td>
<td>Yes</td>
<td>No</td>
<td>CBD</td>
<td>Yes</td>
<td>No</td>
<td>CBD</td>
</tr>
<tr>
<td>North Carolina: Estuary</td>
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<td>1</td>
<td>3</td>
<td>1</td>
<td>0</td>
<td>0</td>
</tr>
<tr>
<td>North Carolina: Coastal</td>
<td>0</td>
<td>2</td>
<td>1</td>
<td>3</td>
<td>4</td>
<td>1</td>
</tr>
<tr>
<td>Total</td>
<td>14</td>
<td>9</td>
<td>16</td>
<td>10</td>
<td>6</td>
<td>38</td>
</tr>
</tbody>
</table>

Table 2. Strandings of common bottlenose dolphins during 2011–2015 from North Carolina that were ascribed to the Southern North Carolina Estuarine System (SNCES) Stock, including the number of strandings for which evidence of human interaction (HI) was detected and number of strandings for which it could not be determined (CBD) if there was evidence of HI. Strandings observed in North Carolina are separated into those occurring within estuaries vs. coastal waters. Assignments to stock were based upon the understanding of the seasonal movements of this stock. However, particularly in coastal waters, there is likely overlap between the SNCES Stock and other common bottlenose dolphin stocks. Data are from the NOAA National Marine Mammal Health and Stranding Response Database (unpublished data, accessed 18 May 2016). Please note HI does not necessarily mean the interaction caused the animal’s death.
<table>
<thead>
<tr>
<th>State</th>
<th>2011</th>
<th>2012</th>
<th>2013</th>
<th>2014</th>
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</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>HI</td>
<td>HI</td>
<td>CBD</td>
<td>HI</td>
<td>HI</td>
</tr>
<tr>
<td></td>
<td>Yes</td>
<td>No</td>
<td>CBD</td>
<td>Yes</td>
<td>No</td>
</tr>
<tr>
<td>North Carolina - Coastal</td>
<td>5^a</td>
<td>4</td>
<td>3</td>
<td>2</td>
<td>4</td>
</tr>
<tr>
<td>North Carolina - Estuary</td>
<td>0</td>
<td>4</td>
<td>4</td>
<td>0</td>
<td>4</td>
</tr>
<tr>
<td>Annual Total</td>
<td>14</td>
<td>10</td>
<td>33</td>
<td>14</td>
<td>9</td>
</tr>
</tbody>
</table>

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

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

^c Includes 3 FIs.

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

^e Includes 2 FIs.

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

**HABITAT ISSUES**

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

Common bottlenose dolphins in the western North Atlantic are not listed as threatened or endangered under the Endangered Species Act. NMFS considers the SNCES Stock to be a strategic stock under the MMPA because while the abundance of the SNCES Stock is currently unknown, based on the restricted range of the stock and previous abundance estimates it is likely small and therefore relatively few mortalities and serious injuries per year would exceed PBR. An annual average of 0.4 carcasses showing evidence of fishery interaction (primarily gillnet interactions, Table 2) were recovered within this stock’s range during 2014–2018. However, this estimate is biased low for the following reasons: 1) the total U.S. human-caused mortality and serious injury for this stock cannot be directly estimated because of the spatial overlap of several stocks of bottlenose dolphins in this area resulting in uncertainty in the stock assignment of takes, and 2) there are several commercial fisheries operating within this stock’s boundaries and these fisheries have little to no observer coverage. In addition, the number of stranded dolphins showing evidence of fishery interactions is nearly 10% of the total number of strandings, suggesting more fishery interactions occur than are observed. Finally, Wells et al. (2015) estimated that only one-third of bottlenose dolphin carcasses in estuarine environments are recovered, indicating significantly more mortalities may occur than are recorded. Therefore, the documented mortalities must be considered minimum estimates of total fishery-related mortality and are of concern given the stock’s restricted range and likely small abundance. There is insufficient information available to determine whether the total fishery-related mortality and serious injury for this stock is insignificant and approaching a zero mortality and serious injury rate. The status of this stock relative to OSP is unknown. There are insufficient data to determine the population trends for this stock. An unbiased abundance estimate for this stock is unavailable, but the stock size is likely less than 200 given the restricted range of the stock and the best available abundance estimate (Urian et al. 2013). An annual average of 3.2 carcasses showing evidence of fishery interaction (primarily gillnet interactions, Table 2) are recovered within this stock’s range. This high number is of concern, particularly in light of Wells et al. (2015) who estimated that only one-third of common bottlenose dolphin carcasses in estuarine environments are recovered. This suggests that annual human-caused mortality could approach 16 animals per year. While it is likely that not every dolphin with evidence of fishery interaction died as a result of that interaction, only five mortalities per year would place the stock at or above PBR if the minimum abundance (Nmin) is anything less than 500. Therefore, given the likely small stock size and the probable negative bias in the estimated total human-caused mortality, this stock is listed as strategic. The status of this stock relative to OSP is unknown. There is insufficient information available to determine whether the total fishery-related mortality and serious injury for this stock is insignificant and approaching a zero mortality and serious injury rate. The abundance of this stock is currently unknown and there are insufficient data to determine population trends for this stock. The impact of the 2013–2015 UME to the status of this stock is unknown.

REFERENCES CITED


Garrison, L.P., A.A. Hohn and L.J. Hansen. 2017b. Seasonal movements of Atlantic common bottlenose dolphin stocks based on tag telemetry data. Southeast Fisheries Science Center, Protected Resources and Biodiversity Division, 75 Virginia Beach Dr., Miami, FL 33140. PRBD Contribution # PRBD-2017-01, XX pp.


HARBOR PORPOISE (Phocoena phocoena phocoena): Gulf of Maine/Bay of Fundy Stock

STOCK DEFINITION AND GEOGRAPHIC RANGE

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

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 subdivision in either sex (Rosel et al. 1999a). These patterns may be indicative of female philopatry coupled with dispersal of males. Both mitochondrial DNA and microsatellite analyses indicate that the Gulf of Maine/Bay of Fundy stock is not the sole contributor to the aggregation of porpoises found off the mid-Atlantic states during winter (Rosel et al. 1999a; Hiltunen 2006). Mixed-stock analyses using twelve microsatellite loci in both Bayesian and likelihood frameworks indicate that the Gulf of Maine/Bay of Fundy is the largest contributor (~60%), followed by Newfoundland (~25%) and then the Gulf of St. Lawrence (~12%), with Greenland making a small contribution (<3%). For Greenland, the lower confidence interval of the likelihood analysis includes zero. For the Bayesian analysis, the lower 2.5% posterior quantiles include zero for both Greenland and the Gulf of St. Lawrence. Intervals that reach zero provide the possibility that these populations contribute no animals to the mid-Atlantic aggregation.

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

POPULATION SIZE

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

Earlier abundance estimates

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

Recent surveys and abundance estimates

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

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

An abundance estimate of 75,079 (CV=0.38) harbor porpoises was generated from a U.S. shipboard and aerial survey conducted during 27 June–28 September 2016 (Table 1; Palka 2020) in a region covering 425,192 km². The aerial portion included 11,782 km of tracklines that were over waters north of New Jersey from the coastline to the 100-m depth contour, throughout the U.S. waters. The shipboard portion included 4,351 km of tracklines that were in waters offshore of central Virginia to Massachusetts (waters that were deeper than the 100-m depth contour out to beyond the outer limit of the U.S. EEZ). Both sighting platforms used a two-team data collection procedure, which allows estimation of abundance to correct for perception bias of the detected species (Laake and Borchers, 2004). The estimates were also corrected for availability bias.

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

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

<table>
<thead>
<tr>
<th>Month/Year</th>
<th>Area</th>
<th>N_best</th>
<th>CV</th>
</tr>
</thead>
<tbody>
<tr>
<td>Jul–Aug 2011</td>
<td>Central Virginia to lower Bay of Fundy</td>
<td>79,883</td>
<td>0.32</td>
</tr>
<tr>
<td>Jun–Sep 2016</td>
<td>Central Virginia to Maine</td>
<td>75,079</td>
<td>0.38</td>
</tr>
<tr>
<td>Aug–Sep 2016</td>
<td>Gulf of St. Lawrence/Bay of Fundy/Scotian Shelf</td>
<td>20,464</td>
<td>0.39</td>
</tr>
<tr>
<td>Jun–Sep 2016</td>
<td>Central Virginia to Gulf of St. Lawrence/Bay of Fundy/Scotian Shelf -COMBINED</td>
<td>95,543</td>
<td>0.31</td>
</tr>
</tbody>
</table>

**Minimum Population Estimate**

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

**Current Population Trend**

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

**CURRENT AND MAXIMUM NET PRODUCTIVITY RATES**

Several attempts have been made to estimate potential population growth rates. Barlow and Boveng (1991), who used a re-scaled human life table, estimated the upper bound of the annual potential growth rate to be 9.4%. Woodley and Read (1991) used a re-scaled Himalayan tahr life table to estimate a likely annual growth rate of 4%. In an attempt to estimate a potential population growth rate that incorporates many of the uncertainties in survivorship and reproduction, Caswell et al. (1998) used a Monte Carlo method to calculate a probability distribution of growth rates. The median potential annual rate of increase was approximately 10%, with a 90% confidence interval of 3–15%. This analysis underscored the considerable uncertainty that exists regarding the potential rate of increase in this population. Moore and Read (2008) conducted a Bayesian population modeling analysis to estimate the potential population growth of harbor porpoise in the absence of bycatch mortality. Their method used fertility data, in combination with age-at-death data from stranded animals and animals taken in gillnets, and was applied under two scenarios to correct for possible data bias associated with observed bycatch of calves. Demographic parameter estimates were ‘model averaged’ across these scenarios. The Bayesian posterior median estimate for potential natural growth rate was 0.046. This last, most recent, value will be the one used for the purpose of this assessment.

Key uncertainties in the estimate of the maximum net productivity rate for this stock were discussed in Moore and Read (2008), which included the assumption that the age structure is stable, and the lack of data to estimate the probability of survivorship to maximum age. The authors considered the effects of these uncertainties on the estimated potential natural growth rate to be minimal.

**POTENTIAL BIOLOGICAL REMOVAL**

Potential Biological Removal (PBR) is the product of minimum population size, one-half the maximum productivity rate, and a recovery factor (MMPA Sec. 3. 16 U.S.C. 1362; Wade and Angliss 1997). The minimum population size is 74,034. The maximum productivity rate is 0.046. The recovery factor is 0.5 because stock’s status relative to OSP is unknown and the CV of the average mortality estimate is less than 0.3 (Wade and Angliss 1997). PBR for the Gulf of Maine/Bay of Fundy harbor porpoise is 851.

**ANNUAL HUMAN-CAUSED MORTALITY AND SERIOUS INJURY**

The total annual estimated average human-caused mortality and serious injury is 217–150 harbor porpoises per year (CV=0.15) from U.S. fisheries using observer data. Canadian bycatch information is not available.

**TABLE 2.** Best and minimum abundance estimates for the Gulf of Maine/Bay of Fundy harbor porpoise (Phocoena phocoena phocoena) with Maximum Productivity Rate (Rmax), Recovery Factor (Fr.), and PBR.

<table>
<thead>
<tr>
<th>Nbest</th>
<th>CV</th>
<th>Nmin</th>
<th>Fr</th>
<th>Rmax</th>
<th>PBR</th>
</tr>
</thead>
<tbody>
<tr>
<td>95,543</td>
<td>0.31</td>
<td>74,034</td>
<td>0.5</td>
<td>0.046</td>
<td>851</td>
</tr>
</tbody>
</table>

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

<table>
<thead>
<tr>
<th>Years</th>
<th>Source</th>
<th>Annual Avg.</th>
<th>CV</th>
</tr>
</thead>
<tbody>
<tr>
<td>2014–2018</td>
<td>U.S. fisheries using observer data</td>
<td>150</td>
<td>0.14</td>
</tr>
</tbody>
</table>

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

**Fishery Information**

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

**Earlier Interactions**

See Appendix V for more information on historical takes.

**U.S. Northeast Sink Gillnet**

Harbor porpoise bycatch in the northern Gulf of Maine occurs primarily from June to September, while in the southern Gulf of Maine and south of New England, bycatch occurs from January to May and September to December. Annual bycatch is estimated using ratio estimator techniques that account for the use of pingers (Hatch and Orphanides 2015–2016; Orphanides and Hatch 2017; Orphanides 2019, 2020, in press). See Table 2 for bycatch estimates and observed mortality and serious injury for the current 5-year period, and Appendix V for historical bycatch information.

**Mid-Atlantic Gillnet**

Harbor porpoise bycatch in Mid-Atlantic waters occurs primarily from December to May in waters off New Jersey and less frequently in other waters ranging farther south, from New Jersey to North Carolina. Annual bycatch is estimated using ratio estimator techniques (Hatch and Orphanides 2015–2016; Orphanides and Hatch 2017; Orphanides 2019, 2020, in press). See Table 2 for bycatch estimates and observed mortality and serious injury for the current 5-year period, and Appendix V for historical bycatch information.

**Northeast Bottom Trawl**

Since 1989, harbor porpoise mortalities have been observed in the northeast bottom trawl fishery, but many of these were not attributable to this fishery because decomposed animals are presumed to have been dead prior to being taken by the trawl. Those infrequently caught freshly dead harbor porpoises have been caught during January to April on Georges Bank or in the southern Gulf of Maine. Fishery-related bycatch rates were estimated using an annual stratified ratio-estimator (Lyssikatos et al. 2020). See Table 2 for bycatch estimates and observed mortality and serious injury for the current 5-year period, and Appendix V for historical bycatch information.

**Canada**

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

*Table 2. From observer program data, summary of the incidental mortality of Gulf of Maine/Bay of Fundy harbor porpoise (Phocoena phocoena phocoena) by commercial fishery including the years sampled, the type of data used, the annual observer coverage, the mortalities and serious injuries recorded by on-board observers, the estimated annual serious injury and mortality, the estimated CV of the annual mortality, and the mean annual combined mortality (CV in parentheses).*
<table>
<thead>
<tr>
<th></th>
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<th></th>
<th></th>
<th></th>
<th></th>
<th></th>
<th></th>
</tr>
</thead>
<tbody>
<tr>
<td>Northeast Sink Gillnet</td>
<td>2013</td>
<td>Obs. Data, Trip</td>
<td>0.11</td>
<td>0</td>
<td>20</td>
<td>0</td>
<td>300</td>
<td>300</td>
<td>0.33</td>
<td>193-132 (0.1615)</td>
</tr>
<tr>
<td></td>
<td>2014</td>
<td>Obs. Data, Trip</td>
<td>0.18</td>
<td>0</td>
<td>28</td>
<td>0</td>
<td>128</td>
<td>128</td>
<td>0.27</td>
<td></td>
</tr>
<tr>
<td></td>
<td>2015</td>
<td>Logbook, Allocated Data</td>
<td>0.14</td>
<td>0</td>
<td>23</td>
<td>0</td>
<td>177</td>
<td>177</td>
<td>0.28</td>
<td></td>
</tr>
<tr>
<td></td>
<td>2016</td>
<td>Obs. Data</td>
<td>0.10</td>
<td>0</td>
<td>11</td>
<td>0</td>
<td>125</td>
<td>125</td>
<td>0.34</td>
<td></td>
</tr>
<tr>
<td></td>
<td>2017</td>
<td>Obs. Data, Dealer</td>
<td>0.12</td>
<td>1</td>
<td>18</td>
<td>7</td>
<td>129</td>
<td>136</td>
<td>0.28</td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td>Mean</td>
<td>0.11</td>
<td>0</td>
<td>9</td>
<td>2</td>
<td>92</td>
<td>92</td>
<td>0.52</td>
<td></td>
</tr>
<tr>
<td>Mid-Atlantic Gillnet</td>
<td>2013</td>
<td>Obs. Data, Weighout</td>
<td>0.03</td>
<td>0</td>
<td>1</td>
<td>0</td>
<td>10</td>
<td>10</td>
<td>1.06</td>
<td>21-174 (0.4055)</td>
</tr>
<tr>
<td></td>
<td>2014</td>
<td>Obs. Data, Weighout</td>
<td>0.05</td>
<td>0</td>
<td>1</td>
<td>0</td>
<td>22</td>
<td>22</td>
<td>1.03</td>
<td></td>
</tr>
<tr>
<td></td>
<td>2015</td>
<td>Obs. Data, Weighout</td>
<td>0.06</td>
<td>2</td>
<td>2</td>
<td>27</td>
<td>33</td>
<td>60</td>
<td>1.16</td>
<td></td>
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<tr>
<td></td>
<td>2016</td>
<td>Obs. Data, Weighout</td>
<td>0.08</td>
<td>2</td>
<td>2</td>
<td>23</td>
<td>23</td>
<td>60</td>
<td>0.64</td>
<td></td>
</tr>
<tr>
<td></td>
<td>2017</td>
<td>Obs. Data, Weighout</td>
<td>0.09</td>
<td>0</td>
<td>1</td>
<td>0</td>
<td>9</td>
<td>9</td>
<td>0.95</td>
<td></td>
</tr>
<tr>
<td>Northeast Bottom Trawl</td>
<td>2013</td>
<td>Obs. Data, Weighout</td>
<td>0.15</td>
<td>0</td>
<td>1</td>
<td>0</td>
<td>5.5</td>
<td>5.5</td>
<td>0.86</td>
<td>2.2-1.1 (0.5386)</td>
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<tr>
<td></td>
<td>2014</td>
<td>Obs. Data, Weighout</td>
<td>0.17</td>
<td>0</td>
<td>1</td>
<td>0</td>
<td>4.4</td>
<td>4.4</td>
<td>0.490</td>
<td></td>
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<tr>
<td></td>
<td>2015</td>
<td>Obs. Data, Weighout</td>
<td>0.19</td>
<td>0</td>
<td>4</td>
<td>0</td>
<td>7.7</td>
<td>7.7</td>
<td>0.490</td>
<td></td>
</tr>
<tr>
<td></td>
<td>2016</td>
<td>Obs. Data, Weighout</td>
<td>0.12</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td>2017</td>
<td>Obs. Data, Weighout</td>
<td>0.12</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td></td>
<td></td>
</tr>
<tr>
<td>TOTAL</td>
<td></td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>217-150 (0.4314)</td>
<td></td>
</tr>
</tbody>
</table>

a Observer data (Obs. Data) are used to measure bycatch rates and the data are collected within the Northeast Fisheries Observer Program. NEFSC collects Weighout (Weighout) landings data that are used as a measure of total effort for the U.S. gillnet fisheries. Mandatory vessel trip report (VTR; Trip Logbook) data are used to determine the spatial distribution of fishing effort in the Northeast sink gillnet fishery.
b Observer coverage for the U.S. Northeast and mid-Atlantic coastal gillnet fisheries is based on tons of fish landed. Northeast bottom trawl fishery coverages are ratios based on trips.
c Serious injuries were evaluated for the 2013–2017 period and include both at-sea monitor and traditional observer data (Josephson et al. 2019).

Other Mortality

United States

There is evidence that harbor porpoises were harvested by natives in Maine and Canada before the 1960s, and the meat was used for human consumption, oil, and fish bait (NMFS 1992). The extent of these past harvests is unknown, though it is believed to have been small. Up until the early 1980s, small kills by native hunters (Passamaquoddy Indians) were reported. It was believed to have nearly stopped (Polacheck 1989) until media reports in September 1997 depicted a Passamaquoddy tribe member dressing out a harbor porpoise. Further articles describing use of porpoise products for food and other purposes were timed to coincide with ongoing legal action in state court.


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

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

<table>
<thead>
<tr>
<th>Area</th>
<th>2013</th>
<th>2014</th>
<th>2015</th>
<th>2016</th>
<th>2017</th>
<th>2018</th>
<th>Total</th>
</tr>
</thead>
<tbody>
<tr>
<td>Maine, New Brunswick, New Jersey</td>
<td>7</td>
<td>5</td>
<td>2</td>
<td>5</td>
<td>8</td>
<td>8</td>
<td>27-28</td>
</tr>
<tr>
<td>New Hampshire</td>
<td>4</td>
<td>1</td>
<td>0</td>
<td>1</td>
<td>2</td>
<td>2</td>
<td>5</td>
</tr>
<tr>
<td>Area</td>
<td>2013</td>
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<td>2015</td>
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<td>2017</td>
<td>2018</td>
<td>Total</td>
</tr>
<tr>
<td>---------------------------</td>
<td>------</td>
<td>------</td>
<td>------</td>
<td>------</td>
<td>------</td>
<td>------</td>
<td>-------</td>
</tr>
<tr>
<td>Massachusetts&lt;sup&gt;a, b, c, e, f&lt;/sup&gt;</td>
<td>40</td>
<td>22</td>
<td>18</td>
<td>8</td>
<td>29</td>
<td>13</td>
<td>117</td>
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<tr>
<td>Rhode Island&lt;sup&gt;d, e&lt;/sup&gt;</td>
<td>3</td>
<td>0</td>
<td>2</td>
<td>2</td>
<td>0</td>
<td>0</td>
<td>24</td>
</tr>
<tr>
<td>Connecticut&lt;sup&gt;b&lt;/sup&gt;</td>
<td>1</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>1</td>
<td>1</td>
</tr>
<tr>
<td>New York&lt;sup&gt;c, d&lt;/sup&gt;</td>
<td>12</td>
<td>1</td>
<td>3</td>
<td>1</td>
<td>12</td>
<td>2</td>
<td>32</td>
</tr>
<tr>
<td>New Jersey&lt;sup&gt;c&lt;/sup&gt;</td>
<td>8</td>
<td>4</td>
<td>2</td>
<td>5</td>
<td>14</td>
<td>5</td>
<td>33</td>
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<tr>
<td>Delaware</td>
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<td>0</td>
<td>0</td>
<td>6</td>
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<tr>
<td>Maryland</td>
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<td>0</td>
<td>0</td>
<td>2</td>
<td>0</td>
<td>5</td>
</tr>
<tr>
<td>Virginia&lt;sup&gt;c, d&lt;/sup&gt;</td>
<td>12</td>
<td>3</td>
<td>3</td>
<td>2</td>
<td>5</td>
<td>1</td>
<td>28</td>
</tr>
<tr>
<td>North Carolina&lt;sup&gt;d&lt;/sup&gt;</td>
<td>2</td>
<td>11</td>
<td>14</td>
<td>1</td>
<td>1</td>
<td>3</td>
<td>34</td>
</tr>
<tr>
<td>TOTAL U.S.</td>
<td>402</td>
<td>39</td>
<td>47</td>
<td>44</td>
<td>25</td>
<td>79</td>
<td>297</td>
</tr>
<tr>
<td>Nova Scotia/Prince Edward Island&lt;sup&gt;g&lt;/sup&gt;</td>
<td>24</td>
<td>9</td>
<td>13</td>
<td>16</td>
<td>22</td>
<td>20</td>
<td>81</td>
</tr>
<tr>
<td>Newfoundland and New Brunswick&lt;sup&gt;b&lt;/sup&gt;</td>
<td>3</td>
<td>0</td>
<td>2</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>5</td>
</tr>
<tr>
<td>GRAND TOTAL</td>
<td>426</td>
<td>48</td>
<td>56</td>
<td>59</td>
<td>161</td>
<td>101</td>
<td>383</td>
</tr>
</tbody>
</table>

- Three Massachusetts live strandings were taken to rehab in 2013 and 1 Maine animal was released alive. In 2016, one animal in Maine and one animal in New Jersey were responded to and released alive. Ten animals were released alive in 2017, 6 of them in Massachusetts, 2 in Maine and 2 in New York.
- Ten total HI cases in 2013 (MA-3, ME-2, NY-3, NJ-1, CT-1), including one released alive (ME). Three of these were considered fishery interactions, including one entangled in gear in Maine.
- Five total HI cases in 2014: 2 in Maine, 1 each in Massachusetts, New Jersey and Virginia. The Virginia case was recorded as a fishery interaction.
- Two HI cases in 2015: 1 in Rhode Island and 1 in New York.
- Two HI cases in 2016: 1 in Rhode Island and 1 in Virginia. The Virginia case was coded as a fishery interaction.
- 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.
- Two HI cases in 2018; both in Massachusetts. One was coded as a fishery interaction.
- Data supplied by Nova Scotia Marine Animal Response Society (pers. comm.). Not included in count for 2014 are at least 8 animals released alive from weirs. One of the 2015 animals a suspected fishery interaction.

**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 2013, 2014, 2015, 2017, 2018; Table 35).

**HABITAT ISSUES**

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

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

STATUS OF STOCK

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

REFERENCES CITED


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

Ledwell, W., J. Huntington and E. Sacrey 2013. Incidental entrapments in fishing gear and strandings reported to and responded to by the Whale Release and Strandings Group in Newfoundland and Labrador and a summary of the Whale Release and Strandings program during 2013. Report to the Department of Fisheries and Oceans Canada, St. John's, Newfoundland, Canada. 19 pp.


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

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

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

Tagging studies of adult harbor seals demonstrate that adults can make long-distance migrations through the mid-Atlantic and Gulf of Maine (Waring et al. 2006, Jones et al. 2018). Prior to these studies, it was believed that the majority of seals moving into southern New England and mid-Atlantic waters were subadults and juveniles (Whitman and Payne 1990; Katona et al. 1993). The more recent studies demonstrate that various age classes utilize habitat along the eastern seaboard throughout the year. Reconnaissance flights for pupping south of Maine would help confirm the extent of the current pupping range.

Although the stock structure of western North Atlantic harbor seals is unknown, it is thought that harbor seals found along the eastern U.S. and Canadian coasts represent one population (Temte et al. 1991; Andersen and Olsen 2010). However, uncertainty in the single stock designation is suggested by multiple sources, both in this population and by inference from other populations. Stanley et al. (1996) demonstrated some genetic differentiation in Atlantic Canada harbor seal samples. Gilbert et al. (2005) noted regional differences in pup count trends along the coast of Maine. Goodman (1998) observed high degrees of philopatry in eastern North Atlantic populations. In addition, multiple lines of evidence have suggested fine-scaled sub-structure in Northeast Pacific harbor seals (Westlake and O’Corry-Crowe 2002; O’Corry-Crowe et al. 2003, Huber et al. 2010).

POPULATION SIZE
The best current abundance estimate of harbor seals is 75,834 (CV=0.15) which is from a 2012 survey (Waring...
Aerial photographic surveys and radio tracking of harbor seals on ledges along the Maine coast were conducted during the pupping period in late May 2012. Twenty-nine harbor seals (20 adults and 9 juveniles) were captured and radio-tagged prior to the aerial survey. Of these, 18 animals were available during the survey to develop a correction factor for the fraction of seals not observed. A key uncertainty is that the area from which the samples were drawn in 2012 may not have included the area the entire population occupied in late May and early June. Additionally, since the most current estimate dates from a survey done in 2012, the ability for that estimate to accurately represent the present population size has become increasingly uncertain. A population survey was conducted in 2018 to provide updated abundance estimates and these data are in the process of being analyzed.

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

<table>
<thead>
<tr>
<th>Month/Year</th>
<th>Area</th>
<th>$N_{best}$</th>
<th>CV</th>
</tr>
</thead>
<tbody>
<tr>
<td>May/June 2012</td>
<td>Maine coast</td>
<td>75,834</td>
<td>0.15</td>
</tr>
</tbody>
</table>

**Minimum Population Estimate**

The minimum population estimate is the lower limit of the two-tailed 60% confidence interval of the log-normally distributed best abundance estimate. This is equivalent to the 20th percentile of the log-normal distribution as specified by Wade and Angliss (1997). The best estimate of abundance for harbor seals is 75,834 (CV=0.15). The minimum population estimate is 66,884 based on corrected available counts along the Maine coast in 2012.

**Current Population Trend**

A trend analysis is currently underway using the 2018 survey data combined with historical data, but the results are not yet available. The statistical power to detect a trend in abundance for this stock is poor due to the relatively imprecise abundance estimates and long survey interval. For example, the power to detect a precipitous decline in abundance (i.e., 50% decrease in 15 years) with estimates of low precision (e.g., CV>0.30) remains below 80% (alpha=0.30) unless surveys are conducted on an annual basis (Taylor et al. 2007).

Although the 2012 population estimate was lower than the previous estimate of 99,340 obtained from a survey in 2001 (Gilbert et al. 2005), Waring et al. (2015) did not consider the population to be declining because the two estimates were not significantly different and there was uncertainty over whether some fraction of the population was not in the survey area. This was due to the fact that 31.4% of the count was pups, a percentage that is biologically unlikely. The estimated number of harbor seal pups did not differ significantly between 2001 and 2012. In 2001, there were an estimated 23,722 (CV=0.096) pups in the study area (Gilbert et al. 2005); in 2012 there were an estimated 23,830 (CV=0.159) pups in the study area. Therefore some non-pups in the population may not have been available to be counted because they were outside the study area of Coastal Maine. Some seals could have remained farther south in New England, more northerly in Canada, or offshore. Therefore, a decline in the apparent abundance of harbor seals could be explained by changing distributions and/or different survey coverage over time. Other lines of evidence provide that support for an apparent decline in abundance and/or changing distributions. In 2001 the population was estimated to be 99,340 (95% CI: 83,118 - 121,397) (Gilbert et al. 2005). While the estimated population size was lower in 2012, Waring et al. (2015) did not consider the population to be declining because the 2012 and 2001 estimates were not significantly different and there was uncertainty over whether some fraction of the population was not in the survey area. In southeastern Massachusetts, counts of harbor seals progressively declined after 2009 (Pace et al. 2019), and reduced population size has been hypothesized from declining rates of stranded and bycaught animals (Johnston et al. 2015). However, the occupancy patterns of harbor seals at haul-out sites has also changed through time in relation to the growth of the sympatric gray seal population (Pace et al. 2019), so inferences about abundance could reflect a sampling and monitoring plan that needs to be revisited. If juvenile seals are redistributing to new areas they may be missed during population surveys, designed around historical pupping habitat. This may have explained differences in the estimated size of the population between 2001 and 2012 (Waring et al. 2015).

**CURRENT AND MAXIMUM NET PRODUCTIVITY RATES**

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

**POTENTIAL BIOLOGICAL REMOVAL**

Potential Biological Removal (PBR) is the product of minimum population size, one-half the maximum productivity rate, and a recovery factor (MMPA Sec. 3. 16 U.S.C. 1362; Wade and Angliss 1997). The minimum population size is 66,884 animals. The maximum productivity rate is 0.12, the default value for pinnipeds. The recovery factor (Fr) is 0.5, the default value for stocks of unknown status relative to optimum sustainable population (OSP) and with the CV of the average mortality estimate less than 0.3 (Wade and Angliss 1997). PBR for the western North Atlantic stock of harbor seals is 2,006.

**Table 2. Best and minimum abundance estimates for the Western North Atlantic harbor seal (Phoca vitulina vitulina) with Maximum Productivity Rate (Rmax), Recovery Factor (Fr) and PBR.**

<table>
<thead>
<tr>
<th>Best (Nbest)</th>
<th>CV</th>
<th>Minimum (Nmin)</th>
<th>Fr</th>
<th>Rmax</th>
<th>PBR</th>
</tr>
</thead>
<tbody>
<tr>
<td>75,834</td>
<td>0.15</td>
<td>66,883</td>
<td>0.5</td>
<td>0.12</td>
<td>2,006</td>
</tr>
</tbody>
</table>

**ANNUAL HUMAN-CAUSED SERIOUS INJURY AND MORTALITY**

For the period 2014–2018, the annual average estimated human-caused mortality and serious injury to harbor seals in the U.S. is 365 (Table 3). Mortality in U.S. fisheries is are explained in further detail below.

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

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

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

<table>
<thead>
<tr>
<th>Years</th>
<th>Source</th>
<th>Annual Avg.</th>
<th>CV</th>
</tr>
</thead>
<tbody>
<tr>
<td>2014–2018</td>
<td>U.S. fisheries using observer data</td>
<td>351</td>
<td>0.12</td>
</tr>
<tr>
<td>2014–2018</td>
<td>non-fishery human interaction stranding mortalities</td>
<td>14.2</td>
<td></td>
</tr>
<tr>
<td>2014–2018</td>
<td>research mortalities</td>
<td>0</td>
<td></td>
</tr>
<tr>
<td>TOTAL</td>
<td></td>
<td>365.2</td>
<td></td>
</tr>
</tbody>
</table>

**Fishery Information**

Detailed fishery information is given in Appendix III.

**U.S. Northeast Sink Gillnet:**

The Northeast sink gillnet fishery is a Category I fishery. The average annual observed mortality from 2014–2018 was 51 animals, and the average annual total mortality was 319 (CV=0.13) (Hatch and Orphanides 2015, 2016, Orphanides and Hatch 2017; Orphanides 2019, 2020, in review, Josephson et al. 2019). 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. Harbor seal bycatch is observed year-round, most frequently in the summer in groundfish trips.
occurring between Boston, Massachusetts, and Maine in coastal Gulf of Maine waters. See Table 2 for bycatch estimates and observed mortality and serious injury for the current 5-year period, and Appendix V for historical bycatch information. Analysis methodology and results can be found in Orphanides (2019, 2020), Hatch and Orphanides (2015, 2016), Orphanides and Hatch (2017), and Josephson et al. (2019).

Mid-Atlantic Gillnet

The Mid-Atlantic sink gillnet fishery is a Category I fishery. The average annual observed mortality from 2014–2018 was 2 animals, and the average annual total mortality was 23 (CV=0.34) (Hatch and Orphanides 2015, 2016; Orphanides and Hatch 2019, 2020, in review; Josephson et al. 2019). 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. Harbor seal bycatch has been observed in this fishery in waters off Massachusetts and New Jersey and rarely further south. See Table 2 for bycatch estimates and observed mortality and serious injury for the current 5-year period, and Appendix V for historical bycatch information. Analysis methodology and results can be found in Orphanides (2019, 2020), Hatch and Orphanides (2015, 2016), and Orphanides and Hatch (2017).

Northeast Bottom Trawl

The Northeast bottom trawl fishery is a Category II fishery. The average annual observed mortality from 2014–2018 was <1 animal, and the average annual total mortality was 4 (CV=0.54) (Lyssikatos et al. in press). 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. Harbor seals are occasionally observed taken in this fishery. See Table 2 for bycatch estimates and observed mortality and serious injury for the current 5-year period, and Appendix V for historical bycatch information. Analysis methodology and results can be found in (Lyssikatos et al. 2020).

Mid-Atlantic Bottom Trawl

The Mid-Atlantic bottom trawl fishery is a Category II fishery. The average annual observed mortality from 2014–2018 was <1 animal, and the average annual total mortality was 5 (CV=0.57) (Lyssikatos et al. in press). 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. Harbor seals are rarely observed taken in this fishery. Annual harbor seal mortalities were estimated using annual stratified ratio-estimator methods (Lyssikatos et al. 2020). See Table 2 for bycatch estimates and observed mortality and serious injury for the current 5-year period, and Appendix V for historical bycatch information.

Northeast Mid-water Trawl Fishery (Including Pair Trawl)

The Northeast mid-water and pair trawl fisheries are Category II fisheries. The average annual observed mortality from 2014–2018 was <1 animal. An expanded bycatch estimate has not been calculated for the current 5-year period. See Table 4 for observed mortality and serious injury during the current 5-year period, and Appendix V for historical bycatch information. Harbor seals are occasionally observed taken in this fishery. An extended bycatch rate has not been calculated for the current 5-year period. Until this bycatch estimate can be developed, the average annual fishery-related mortality and serious injury for 2013–2017 is calculated as 0.8 animals (4 animals/5 years). See Table 2 for bycatch estimates and observed mortality and serious injury for the current 5-year period, and Appendix V for historical bycatch information.

Gulf of Maine Atlantic Herring Purse Seine Fishery

The Gulf of Maine Atlantic Herring Purse Seine Fishery is a Category III fishery. This fishery was not observed until 2003. No mortalities have been observed in this fishery and no harbor seals were captured and released alive in 2013 and 0 in 2014–2017. In addition, 0 seals of unknown species were captured and released alive in 2013–2014, 2 in 2015, 1 in 2016, and 0 in 2017. None of the seals captured alive in herring purse seine during 2013–2017 were designated as serious injuries (Josephson et al. 2019).

Currently, scant data are available on bycatch in Atlantic Canada fisheries due to limited observer programs (Baird 2001). An unknown number of harbor seals have been taken in Newfoundland, Labrador, Gulf of St. Lawrence and Bay of Fundy groundfish gillnets; Atlantic Canada and Greenland salmon gillnets; Atlantic Canada cod traps; and in Bay of Fundy herring weirs (Read 1994; Cairns et al. 2000). Furthermore, some of these mortalities (e.g., seals trapped in herring weirs) are the result of direct shooting under nuisance permits.
Table 24. Summary of the incidental mortality of harbor seals (Phoca vitulina vitulina) by commercial fishery including the years sampled (Years), the number of vessels active within the fishery (Vessels), the type of data used (Data Type), the annual observer coverage (Observer Coverage), the mortalities recorded by on-board observers (Observed Mortality), the estimated annual mortality (Estimated Mortality), the estimated CV of the annual mortality (Estimated CVs) and the mean annual mortality (CV in parentheses).

<table>
<thead>
<tr>
<th>Fishery</th>
<th>Years</th>
<th>Data Type</th>
<th>Observer Coverage</th>
<th>Observed Serious Injury</th>
<th>Observed Mortality</th>
<th>Estimated Serious Injury</th>
<th>Estimated Mortality</th>
<th>Estimated Combined Mortality</th>
<th>Estimated CVs</th>
<th>Mean Annual Mortality</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>Northeast Sink Gillnet</strong></td>
<td>2013</td>
<td>Obs. Data, Weighout, Logbooks</td>
<td>0.11</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0.51</td>
</tr>
<tr>
<td>2014</td>
<td></td>
<td></td>
<td>0.18</td>
<td>0</td>
<td>59</td>
<td>0</td>
<td>390</td>
<td>390</td>
<td>390</td>
<td>0.39</td>
</tr>
<tr>
<td>2015</td>
<td></td>
<td></td>
<td>0.14</td>
<td>0</td>
<td>87</td>
<td>0</td>
<td>474</td>
<td>474</td>
<td>474</td>
<td>0.17</td>
</tr>
<tr>
<td>2016</td>
<td></td>
<td></td>
<td>0.10</td>
<td>0</td>
<td>36</td>
<td>0</td>
<td>245</td>
<td>245</td>
<td>245</td>
<td>0.29</td>
</tr>
<tr>
<td>2017</td>
<td></td>
<td></td>
<td>0.12</td>
<td>0</td>
<td>63</td>
<td>0</td>
<td>298</td>
<td>298</td>
<td>298</td>
<td>0.36</td>
</tr>
<tr>
<td>2018</td>
<td></td>
<td></td>
<td>0.11</td>
<td>0</td>
<td>87</td>
<td>0</td>
<td>474</td>
<td>474</td>
<td>474</td>
<td>0.17</td>
</tr>
<tr>
<td><strong>Mid-Atlantic Gillnet</strong></td>
<td>2013</td>
<td>Obs. Data, Weighout</td>
<td>0.03</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
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<tr>
<td>2014</td>
<td></td>
<td></td>
<td>0.05</td>
<td>0</td>
<td>1</td>
<td>0</td>
<td>19</td>
<td>19</td>
<td>19</td>
<td>0.63</td>
</tr>
<tr>
<td>2015</td>
<td></td>
<td></td>
<td>0.06</td>
<td>0</td>
<td>5</td>
<td>0</td>
<td>48</td>
<td>48</td>
<td>48</td>
<td>0.52</td>
</tr>
<tr>
<td>2016</td>
<td></td>
<td></td>
<td>0.08</td>
<td>0</td>
<td>2</td>
<td>0</td>
<td>18</td>
<td>18</td>
<td>18</td>
<td>0.95</td>
</tr>
<tr>
<td>2017</td>
<td></td>
<td></td>
<td>0.09</td>
<td>0</td>
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<td>0</td>
<td>3</td>
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</tr>
<tr>
<td>2018</td>
<td></td>
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<td>0.09</td>
<td>0</td>
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<td>26</td>
<td>26</td>
<td>26</td>
<td>0.52</td>
</tr>
<tr>
<td><strong>Northeast Bottom Trawl</strong></td>
<td>2013</td>
<td>Obs. Data, Weighout</td>
<td>0.15</td>
<td>0</td>
<td>1</td>
<td>0</td>
<td>4</td>
<td>4</td>
<td>4</td>
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<tr>
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<td>0.17</td>
<td>0</td>
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<td>11</td>
<td>0.63</td>
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<td>2015</td>
<td></td>
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<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
</tr>
<tr>
<td><strong>Mid-Atlantic Bottom Trawl</strong></td>
<td>2013</td>
<td>Obs. Data, Dealer</td>
<td>0.06</td>
<td>0</td>
<td>1</td>
<td>0</td>
<td>11</td>
<td>11</td>
<td>11</td>
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<tr>
<td>2014</td>
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<td>0</td>
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<td>0</td>
<td>7</td>
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<td>1</td>
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<tr>
<td>2015</td>
<td></td>
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<td>0</td>
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<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
</tr>
<tr>
<td><strong>Northeast Mid-water Trawl - Including Pair Trawl</strong></td>
<td>2013</td>
<td>Obs. Data, Weighout, Trip Logbook</td>
<td>0.37</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
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<tr>
<td>2014</td>
<td></td>
<td></td>
<td>0.42</td>
<td>0</td>
<td>1</td>
<td>0</td>
<td>na</td>
<td>na</td>
<td>na</td>
<td>na</td>
</tr>
<tr>
<td>2015</td>
<td></td>
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<td>0</td>
<td>na</td>
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<tr>
<td>2016</td>
<td></td>
<td></td>
<td>0.27</td>
<td>0</td>
<td>1</td>
<td>0</td>
<td>na</td>
<td>na</td>
<td>na</td>
<td>na</td>
</tr>
<tr>
<td>2017</td>
<td></td>
<td></td>
<td>0.16</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
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<td>0</td>
</tr>
<tr>
<td>2018</td>
<td></td>
<td></td>
<td>0.16</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
</tr>
<tr>
<td><strong>TOTAL</strong></td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>-</td>
</tr>
</tbody>
</table>

a. Observer data (Obs. Data) are used to measure bycatch rates, and the data are collected within the Northeast Fisheries Observer Program. NEFSC collects landings data (Weighout), and total landings are used as a measure of total effort for the sink gillnet fishery. Mandatory logbook (Logbook) data are used to determine the spatial distribution of fishing effort in the northeast sink gillnet fishery.

b. The observer coverages for the northeast sink gillnet fishery and the mid-Atlantic gillnet fisheries are ratios based on tons of fish landed and coverages for the bottom and mid-water trawl fisheries are ratios based on trips. Total observer coverage reported for bottom trawl gear and gillnet gear in the years 2013-2014, 2016-2017, and 2018 includes samples collected from traditional fisheries observers in addition to fishery monitors through the Northeast Fisheries Observer Program (NEFOP).

c. Serious injuries were evaluated for the years 2013-2014, 2016-2017, and 2018 and include both at-sea monitor and traditional observer data (Josephson et al. 2019 in press).

**Other Mortality**

**United States**

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

Harbor seals strand each year throughout their migratory range. Stranding data provide insight into some of these sources of mortality. Tables 5 and 6 present summaries of harbor seal stranding mortalities as reported to the NOAA National Marine Mammal Health and Stranding Response Database (accessed 20 November 2019). From 2013 to 2017, 1,214 harbor seal stranding mortalities were reported between Maine and Florida (Table 3; NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 23 October 2018). Seventy (5.8%) of the dead harbor seals stranded during this five year period showed signs of human interaction (15 in 2013, 11 in 2014, 18 in 2015, 16 in 2016, and 10 in 2017), with 10 (0.8%) having some sign of fishery interaction (3 in 2013, 2 in 2014, 2 in 2015, 3 in 2016, and 1 in 2017). Three harbor seals during this period were reported as having been shot. Seven harbor seal mortalities were reported with indications of vessel strike. In an analysis of mortality causes of stranded marine mammals on Cape Cod and southeastern Massachusetts between 2000 and 2006, Bogomolni et al. (2010) reported that 13% of harbor seal stranding mortalities were attributed to human interaction.

A number of Unusual Mortality Events (UMEs) have affected harbor seals over the past decade. A UME was declared for harbor seals in northern Gulf of Maine waters in 2003 and continued into 2004. No consistent cause of death could be determined. The UME was declared over in spring 2005 (MMC 2006). NMFS declared another UME in the Gulf of Maine in autumn 2006 based on infectious disease. A UME was declared in November of 2011 that involved 567 harbor seal stranding mortalities between June 2011 and October 2012 in Maine, New Hampshire, and Massachusetts. The UME was declared closed in February 2013 (https://www.fisheries.noaa.gov/national/marine-life-distress/active-and-closed-unusual-mortality-events). Another UME was declared by the NMFS beginning in July 2018 due to increased numbers of harbor and gray seal strandings along the U.S. coasts of Maine, New Hampshire, and Massachusetts. Strandings remained elevated over the summer and the UME area was expanded to include nine states from Maine to Virginia with strandings continuing into 2019. From July to December 2018, 1,100 harbor seals stranded predominantly in Maine, New Hampshire and Massachusetts. The preliminary cause of the UME was attributed to a phocine distemper outbreak (https://www.fisheries.noaa.gov/new-england-mid-atlantic/marine-life-distress/2018-2020-pinniped-unusual-mortality-event-along).

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

CANADA

Aquaculture operations in eastern Canada can be licensed to shoot nuisance seals, but the number of seals killed is unknown (Jacobs and Terhune 2000; Baird 2001). Small numbers of harbor seals are taken in subsistence hunting in northern Canada (DFO 2011).

Table 35. Harbor seal (Phoca vitulina vitulina) stranding mortalities along the U.S. Atlantic coast (2013-2018) with subtotals of animals recorded as pups in parentheses.

<table>
<thead>
<tr>
<th>State</th>
<th>2013</th>
<th>2014</th>
<th>2015</th>
<th>2016</th>
<th>2017</th>
<th>2018</th>
<th>Total</th>
</tr>
</thead>
<tbody>
<tr>
<td>Maine</td>
<td>99 (74)</td>
<td>127 (94)</td>
<td>73 (47)</td>
<td>76 (58)</td>
<td>120 (84)</td>
<td>819 (75)</td>
<td>495 (1,215) (352 344)</td>
</tr>
<tr>
<td>New Hampshire</td>
<td>16 (6)</td>
<td>38 (22)</td>
<td>56 (43)</td>
<td>45 (27)</td>
<td>26 (20)</td>
<td>113 (60)</td>
<td>181 (278) (418 171)</td>
</tr>
<tr>
<td>Massachusetts</td>
<td>95 (39)</td>
<td>58 (15)</td>
<td>81 (24)</td>
<td>55 (19)</td>
<td>78 (29)</td>
<td>204 (58)</td>
<td>367 (476) (126 145)</td>
</tr>
<tr>
<td>Rhode Island</td>
<td>9 (3)</td>
<td>7 (1)</td>
<td>8 (0)</td>
<td>5 (1)</td>
<td>9 (3)</td>
<td>9 (0)</td>
<td>38 (85)</td>
</tr>
<tr>
<td>State</td>
<td>2013</td>
<td>2014</td>
<td>2015</td>
<td>2016</td>
<td>2017</td>
<td>2018</td>
<td>Total</td>
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<td>------</td>
<td>-------</td>
</tr>
<tr>
<td>Connecticut</td>
<td>2 (1)</td>
<td>0</td>
<td>2 (1)</td>
<td>1 (0)</td>
<td>2 (0)</td>
<td>2 (1)</td>
<td>7 (2)</td>
</tr>
<tr>
<td>New York</td>
<td>11 (2)</td>
<td>13 (4)</td>
<td>21 (0)</td>
<td>1 (0)</td>
<td>11 (0)</td>
<td>12 (1)</td>
<td>57 (5)</td>
</tr>
<tr>
<td>New Jersey</td>
<td>4 (0)</td>
<td>2 (1)</td>
<td>9 (4)</td>
<td>4 (0)</td>
<td>9 (3)</td>
<td>14 (2)</td>
<td>28 (10)</td>
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<tr>
<td>Delaware</td>
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<td>1 (1)</td>
<td>1 (0)</td>
<td>2 (1)</td>
<td>6 (4)</td>
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<td>Maryland</td>
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<td>0</td>
<td>0</td>
<td>1 (0)</td>
<td>4 (0)</td>
<td>4 (0)</td>
</tr>
<tr>
<td>Virginia</td>
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<td>2 (0)</td>
<td>1 (0)</td>
<td>1 (0)</td>
<td>2 (0)</td>
<td>1 (0)</td>
<td>8 (0)</td>
</tr>
<tr>
<td>North Carolina</td>
<td>3 (0)</td>
<td>3 (1)</td>
<td>5 (2)</td>
<td>4 (2)</td>
<td>4 (4)</td>
<td>7 (2)</td>
<td>19 (11)</td>
</tr>
<tr>
<td>South Carolina</td>
<td>0</td>
<td>1 (0)</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>1 (0)</td>
</tr>
<tr>
<td><strong>Total</strong></td>
<td><strong>245</strong></td>
<td><strong>256</strong></td>
<td><strong>257</strong></td>
<td><strong>193</strong></td>
<td><strong>263</strong></td>
<td><strong>1,187</strong></td>
<td><strong>2,156 (635)</strong></td>
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<tr>
<td>Unspecified seals (all states)</td>
<td>25</td>
<td>38</td>
<td>31</td>
<td>13</td>
<td>86</td>
<td>92</td>
<td>193 (260)</td>
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<table>
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<tr>
<th>Cause</th>
<th>2014</th>
<th>2015</th>
<th>2016</th>
<th>2017</th>
<th>2018</th>
<th>Total</th>
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<td>3</td>
<td>1</td>
<td>5</td>
<td>13</td>
</tr>
<tr>
<td>Boat Strike</td>
<td>2</td>
<td>1</td>
<td>5</td>
<td>3</td>
<td>2</td>
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<td>0</td>
<td>0</td>
<td>1</td>
</tr>
<tr>
<td>Human Interaction - Other</td>
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<td>15</td>
<td>8</td>
<td>6</td>
<td>22</td>
<td>57</td>
</tr>
<tr>
<td><strong>TOTAL</strong></td>
<td><strong>11</strong></td>
<td><strong>18</strong></td>
<td><strong>16</strong></td>
<td><strong>10</strong></td>
<td><strong>29</strong></td>
<td><strong>84</strong></td>
</tr>
</tbody>
</table>

**STATUS OF STOCK**

Harbor seals are not listed as threatened or endangered under the Endangered Species Act, and the western North Atlantic stock is not considered strategic under the Marine Mammal Protection Act. The 2013–2017 average annual human-caused mortality and serious injury does not exceed PBR. The status of the western North Atlantic harbor seal stock, relative to OSP, in the U.S. Atlantic EEZ is unknown. Total fishery-related mortality and serious injury for this stock is not less than 10% of the calculated PBR and, therefore, cannot be considered to be insignificant and approaching zero mortality and serious injury rate.

**REFERENCES CITED**


Lyssikatos, M.C., Chavez-Rosales, S., and J. Hatch. in press. Estimates of cetacean and pinniped bycatch in northeast and mid-Atlantic bottom trawl fisheries, 2014–2018


GRAY SEAL (*Halichoerus grypus atlantica*): Western North Atlantic Stock

**STOCK DEFINITION AND GEOGRAPHIC RANGE**

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

After near extirpation due to bounties, which ended in the 1960s, in the mid-1980s, small numbers of animals and pupping pups were observed on several isolated islands along the Maine coast and in Nantucket-Vineyard Sound, Massachusetts (Katona *et al.* 1993; Rough 1995; Gilbert *et al.* 2005). In December 2001, NMFS initiated aerial surveys to monitor gray seal pup production on Muskeget Island and adjacent sites in Nantucket Sound, and Green and Seal Islands off the coast of Maine (Wood *et al.* 2007). Tissue samples collected from Canadian and U.S. populations were examined for genetic variation using mitochondrial and nuclear DNA (Wood *et al.* 2011). All individuals were identified as belonging to one population, confirming the new U.S. population was recolonized by Canadian gray seals. The genetic evidence (Boskovic *et al.* 1996; Wood *et al.* 2011) provides a high degree of certainty that the Western North Atlantic stock of gray seals is comprise a single stock. Further supporting evidence comes from sightings of seals in the U.S. that had been branded on Sable Island, resights of tagged animals, and satellite tracks of tagged animals (Puryear *et al.* 2016). However, the percentage of time that individuals are resident in U.S. waters is unknown.

**POPULATION SIZE**
The size of the western Northwest Atlantic gray seal population is estimated separately for the portion of the population in Canada versus the U.S., and mainly reflects the size of the breeding population in each respective country (Table 1). Currently there is a lack of information on the rate of exchange between animals in the U.S. and Canada, which influences seasonal changes in abundance throughout the range of this transboundary stock as well as life history parameters in population models. Total pup production in 2016 at breeding colonies in Canada was estimated to be 98,650 pups (CV=0.10) (den Heyer 2017; DFO 2017). Production at Sable Island, Gulf of St. Lawrence, and Coastal Nova Scotia colonies accounted for 85%, 11% and 4%, respectively, of the estimated total number of pups born. Population models, incorporating estimates of age-specific reproductive rates and removals, are fit to these pup production estimates to estimate total population levels in Canada. The total Canadian gray seal population in 2016 was estimated to be 424,300 (95% CI=263,600 to 578,300) (DFO 2017). Uncertainties in the population estimate derive from uncertainties in life history parameters such as mortality rates and sex ratios (DFO 2017).

In U.S. waters, the number of pupping sites has increased from 1 in 1988 to 9 in 2019, and are located in Maine and Massachusetts (Wood et al. 2019). Although white-coated pups have stranded on eastern Long Island beaches in New York, no pupping colonies have been detected in that region. A minimum of 6,308 of pups were born in 2016 at U.S. breeding colonies (Wood et al. 2019), approximately 6% of the total pup production over the entire range of the stock population (denHeyer et al. 2017). The percentage of pup production in the U.S. is considered a minimum because pup counts are single day counts that have not been adjusted to account for pups born after the survey, or that left the colony prior to the survey. Mean rates of increase in the number of pups born at various times since 1988 at 4 of the more frequently surveyed pupping sites (Muskeget, Monomoy, Seal, and Green Islands) ranged from -0.2% (95%CI: -2.3–1.9%) to 26.3% (95%CI: 21.6–31.4%) (Wood et al. 2019). These high rates of increase provide further support that seals from other areas are continually supplementing the breeding population in U.S. waters. Table 2 summarizes single-day pup counts from U.S. pupping colonies from 2001/2002 to 2015/2016 pupping periods. Aerial survey data from these sites indicate that pup production is increasing (Table 2), although aerial survey quality and coverage has varied significantly among surveys. In U.S. waters, gray seals primarily pup at four established colonies: Muskeget and Monomoy islands in Massachusetts, and Green and Seal islands in Maine. Gray seals have been observed using the historic pupping site on Muskeget Island in Massachusetts since 1988. Pupping has taken place on Seal and Green Islands in Maine since at least the mid-1990s. Since 2010 pupping has also been observed at Noman’s Island in Massachusetts and Wooden Ball and Matinicus Rock in Maine. Although white-coated pups have stranded on eastern Long Island beaches in New York, no pupping colonies have been detected in that region.

Using Canadian population models, the number of pups born at U.S. breeding colonies can be used to approximate the total size (pups and adults) of the gray seal population in U.S. waters, based on the ratio of total best population size to pups in Canadian waters (4.3:1) (den Heyer et al. 2017; DFO 2017). Although not yet measured for U.S. waters, this ratio falls within the range of other adult to pup ratios suggested for pinniped populations (Harwood and Prime 1978; Thomas et al. 2019). Using this approach, the population estimate in U.S. waters is 27,131 (CV=0.19, 95% CI: 18,768–39,221) animals. The CV and CI around this estimate is based on CVs and CIs from Canadian population estimates, rather than using a default CV when the variance is unknown (Wade and Angliss 1997). There is further uncertainty in this abundance level in the U.S. because life history parameters that influence the ratio of pups to total individuals in this portion of the population are unknown. It also does not reflect seasonal changes in stock abundance in the Northeast region for a transboundary stock. For example, roughly 24,000 seals were observed in southeastern Massachusetts alone in 2015 (Pace et al. 2019), and yet an estimated 28,000–40,000 gray seals
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.

<table>
<thead>
<tr>
<th>Month/Year</th>
<th>Area</th>
<th>N&lt;sub&gt;best&lt;/sub&gt;</th>
<th>CI</th>
</tr>
</thead>
<tbody>
<tr>
<td>2012&lt;sup&gt;b&lt;/sup&gt;</td>
<td>Gulf of St Lawrence + Nova Scotia Eastern Shore + Sable Island</td>
<td>331,000</td>
<td>263,000–458,000</td>
</tr>
<tr>
<td>2014&lt;sup&gt;c&lt;/sup&gt;</td>
<td>Gulf of St Lawrence + Nova Scotia Eastern Shore + Sable Island</td>
<td>505,000</td>
<td>329,000–682,000</td>
</tr>
<tr>
<td>2016&lt;sup&gt;d&lt;/sup&gt;</td>
<td>Gulf of St Lawrence + Nova Scotia Eastern Shore + Sable Island</td>
<td>424,300</td>
<td>263,600–578,300</td>
</tr>
<tr>
<td>2016</td>
<td>U.S</td>
<td>27,131&lt;sup&gt;e&lt;/sup&gt;</td>
<td>18,768–39,221</td>
</tr>
</tbody>
</table>

*These are model-based estimates derived from pup surveys.
<sup>b</sup> DFO 2013
<sup>c</sup> DFO 2014
<sup>d</sup> DFO 2017
<sup>e</sup>This is derived from total population size to pup ratios in Canada, applied to U.S. pup counts.

Table 2. Single day pup counts from five U.S. pupping colonies during 2001–2016 from aerial surveys. *= Surveys need further evaluation before reporting. As single day pup counts, these counts do not represent the entire number of pups born in a pupping season.

<table>
<thead>
<tr>
<th>Pupping Season</th>
<th>Massachusetts</th>
<th>Maine</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Muskeget Island</td>
<td>Monomoy Island</td>
</tr>
<tr>
<td>2001-02</td>
<td>883</td>
<td>Not surveyed</td>
</tr>
<tr>
<td>2002-03</td>
<td>509</td>
<td>Not surveyed</td>
</tr>
<tr>
<td>2003-04</td>
<td>824</td>
<td>Not surveyed</td>
</tr>
<tr>
<td>2004-05</td>
<td>592</td>
<td>1</td>
</tr>
<tr>
<td>2005-06</td>
<td>868</td>
<td>8</td>
</tr>
<tr>
<td>2006-07</td>
<td>4,704</td>
<td>0</td>
</tr>
<tr>
<td>2007-08</td>
<td>2,095</td>
<td>2</td>
</tr>
</tbody>
</table>
The minimum population estimate is the lower limit of the two-tailed 60% confidence interval of the log-normally distributed best abundance estimate. This is equivalent to the 20th percentile of the log-normal distribution as specified by Wade and Angliss (1997). Based on an estimated U.S. population in 2016 of 27,131 (CV=0.19), the minimum population estimate in U.S. waters is 23,158 (Table 2). Similar to the best abundance estimate, there is uncertainty in this minimum abundance level in the U.S. because life history parameters that influence the ratio of pups to total individuals in this population are unknown.

Current Population Trend

In the U.S., the mean rate of increase in the number of pups born differs across all U.S. the pupping colonies from 1991-2016 is currently being evaluated. From 1988–2019, the estimated mean rate of increase in the number of pups born was 12.8% on Muskeget Island, 26.3% on Monomoy Island, 11.5% on Seal Island, and -0.2% on Green Island (Wood et al. 2019). These rates only reflect new recruits to the population and do not reflect changes in total population growth resulting from Canadian seals migrating to the region. More data on movements of animals between Canada and the U.S. is needed—particularly the number of adult breeding females recruiting into U.S. colonies each year—to separate out intrinsic rates of increase from the overall annual growth rate.

The 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. The population in eastern Canada was greatly reduced by hunting and bounty programs, and in the 1950s the gray seal was considered rare (Lesage and Hammill 2001). The Sable Island, Nova Scotia, population was less affected and has been increasing for several decades. Pup production on Sable Island increased exponentially at a rate of 12.8% per year between the 1970s and 1997 (Stobo and Zwanenburg 1990; Mohn and Bowen 1996; Bowen et al. 2003; Trzcinski et al. 2005; Bowen et al. 2007; DFO 2011). Pupping also occurs on Hay Island off Nova Scotia, in colonies off southwestern Nova Scotia, and in the Gulf of St. Lawrence. Since 1997, the rate of increase has been slowed (Bowen et al. 2011, den Heyer et al. 2017), supporting the hypothesis that density-dependent changes in vital rates may be limiting population growth. While slowing, pup production is still increasing on Sable Island and in southwest Nova Scotia, and stabilizing on Hay Island in the Gulf of St. Lawrence (DFO 2017, den Heyer et al. 2017). In the Gulf of St. Lawrence, the proportion of pups born on the ice has declined from 100% in 2004 to 1% in 2016 due to a decline in winter ice cover in the area, and seals have responded by pupping on nearby islands (DFO 2017).
The projected population trends for all Canadian aggregations are still increasing. The model projections in 2016 differed from previous analyses due to changes in adult sex ratio and adult mortality rates (DFO 2017). Uncertainties in the population abundance estimates and mortality could have impacts on the abundance trends.

**CURRENT AND MAXIMUM NET PRODUCTIVITY RATES**

Current and maximum net productivity rates are unknown for this stock. Recent studies estimated the current annual rate of increase at 4.5% for the combined breeding aggregations in Canada (DFO 2014), continuing a decline in the rate of increase (Trzcinski et al. 2005; Bowen et al. 2007; Thomas et al. 2011; DFO 2014). For purposes of this assessment, the maximum net productivity rate was assumed to be 0.12. This value is based on theoretical modeling showing that pinniped populations may not grow at rates much greater than 12% given the constraints of their reproductive life history (Barlow et al. 1995).

**POTENTIAL BIOLOGICAL REMOVAL**

Potential Biological Removal (PBR) is the product of minimum population size, one-half the maximum productivity rate, and a recovery factor (MMPA Sec. 3. 16 U.S.C. 1362; Wade and Angliss 1997). The minimum population size for the portion of the stock residing in U.S. waters is 23,158. The maximum productivity rate is 0.12, the default value for pinnipeds. 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 western North Atlantic stock of gray seals residing in U.S. waters is 1,389 animals (Table 2). Uncertainty in the PBR level arises from the same sources of uncertainty in calculating a minimum abundance estimate in U.S. waters.

**Table 2.** Best and minimum abundance estimates for the western North Atlantic gray seal (*Halichoerus grypus atlantica*) with Maximum Productivity Rate (Rmax), Recovery Factor (Fr) and PBR.

<table>
<thead>
<tr>
<th>Nbest</th>
<th>CV</th>
<th>Nmin</th>
<th>Fr</th>
<th>Rmax</th>
<th>PBR</th>
</tr>
</thead>
<tbody>
<tr>
<td>27,131</td>
<td>0.19</td>
<td>23,153</td>
<td>1</td>
<td>0.12</td>
<td>1,389</td>
</tr>
</tbody>
</table>

**ANNUAL HUMAN-CAUSED MORTALITY AND SERIOUS INJURY**

For the period 2014–2018, the average annual estimated human-caused mortality and serious injury to gray seals in the U.S. and Canada was 5,410,4,729 (946–953 U.S./4,464–3,776 Canada) per year. The average was derived from six components: 1) 940 (CV=0.09) (Table 3) from the 2013–2017 U.S. observed fisheries; 2) 5.6 from average 2013–2017 non-fishery related, human interaction stranding and shooting mortalities in the U.S.; 3) 0.8 from U.S. research mortalities; 4) 672 from the average 2013–2017 Canadian commercial harvest; 5) 55 from the average 2013–2017 DFO scientific collections; and 6) 3,737 removals of nuisance animals in Canada (DFO 2017, Mike Hammill pers. comm). 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*).**

<table>
<thead>
<tr>
<th>Years</th>
<th>Source</th>
<th>Annual Avg.</th>
<th>CV</th>
</tr>
</thead>
<tbody>
<tr>
<td>2014-2018</td>
<td>U.S. fisheries using observer data</td>
<td>946</td>
<td>0.11</td>
</tr>
<tr>
<td>2014-2018</td>
<td>U.S. non-fishery human interaction stranding mortalities</td>
<td>6.2</td>
<td></td>
</tr>
<tr>
<td>2014-2018</td>
<td>U.S. research mortalities</td>
<td>1.2</td>
<td></td>
</tr>
<tr>
<td>2014-2018</td>
<td>Canadian commercial harvest</td>
<td>636</td>
<td></td>
</tr>
<tr>
<td>2014-2018</td>
<td>DFO Canada scientific collections</td>
<td>62</td>
<td></td>
</tr>
<tr>
<td>2014-2018</td>
<td>Canadian removals of nuisance animals</td>
<td>3,078</td>
<td></td>
</tr>
</tbody>
</table>
A source of unquantified human-caused mortality or serious injury may not be able to be quantified for this stock is the fact that Observed serious injury rates are lower than would be expected from the anecdotally observed numbers of gray seals living with ongoing entanglements. Estimated rates of entanglement in gillnet gear, for example, may be biased low because 100% of observed animals are dead when they come aboard the vessel (Josephson et al., 2019in review); therefore, rates do not reflect the number of live animals that may have broken free of the gear and are living with entanglements. For example, mean prevalence of live entangled gray seals ranged from roughly 1 to 4% at haul-out sites in Massachusetts and Isle of Shoals (Iruzun Martins et al. 2019). Reports of seal shootings and other non-fishery-related human interactions are minimum counts. Canadian reporting of nuisance seal removal is known to be incomplete and there is also limited information on Canadian fishery bycatch (DFO 2017).

**Fishery Information**

Detailed fishery information is given in Appendix III.

**U.S. United States**

**Northeast Sink Gillnet**

The Northeast sink gillnet fishery is a Category I fishery. The average annual observed mortality from 2014 to 2018 was 199 animals, and the average annual estimated total mortality was 896 (CV=0.11) Gray seal bycatch in the northeast sink gillnet fishery was usually observed in the first half of the year in waters to the east and south of Cape Cod, Massachusetts in 12-inch gillnets fishing for skates and monkfish (Hatch and Orphanides 2015, 2016, Orphanides and Hatch 2017; Orphanides 2019, 2020, in press). See Table 3-4 for bycatch estimates and observed mortality and serious injury for the current 5-year period, and Appendix V for historical bycatch information.

**Mid-Atlantic Gillnet**

The Mid-Atlantic sink gillnet fishery is a Category I fishery. The average annual observed mortality from 2014-2018 was <1 animal, and the average annual total mortality was 9 (CV=0.67) Gray seal interactions were first observed in this fishery in 2010, since then, when they are observed, it is usually in waters off New Jersey in gillnets that have mesh sizes ≥ 7 in (Hatch and Orphanides 2015, 2016; Orphanides and Hatch 2017; Orphanides 2019, 2020, in press). See Table 3-4 for bycatch estimates and observed mortality and serious injury for the current 5-year period, and Appendix V for historical bycatch information.

**Northeast Mid-Water Trawl**

One gray seal mortality was observed in 2013 in this fishery. An expanded bycatch estimate has not been generated. Until this bycatch estimate can be developed, the average annual fishery-related mortality and serious injury for 2013-2017 is calculated as 0.2 animals (1 animal/5 years). See Table 3 for bycatch estimates and observed mortality and serious injury for the current 5-year period, and Appendix V for historical bycatch information.

**Gulf of Maine Atlantic Herring Purse Seine Fishery**

The Gulf of Maine Atlantic Herring Purse Seine Fishery is a Category III fishery. This fishery was not observed until 2003, and was not observed in 2006. No mortalities have been observed in this fishery, but during this time period, 2 seals were captured and released alive in 2013, 2 in 2014, 0 in 2015, 5 in 2016 and 0 in 2017 and 1 in 2018. In addition, during this time period 2 seals of unknown species were captured and released alive in 2015 and 1 in 2016 (Josephson et al. 2019 in press).

**Northeast Bottom Trawl**

The Northeast bottom trawl fishery is a Category II fishery. The average annual observed mortality from 2014-2018 was 3 animals, and the average annual total mortality was 18 (CV=0.22) Vessels in the North Atlantic bottom trawl fishery, a Category III fishery under MMPA, were observed in order to meet fishery management, rather than marine mammal management needs. Five gray seal mortalities were observed in this fishery in 2013, 4 in 2014, 4 in...
2015, 0 in 2016 and 2 in 2017 (Lyssikatos et al. 2020 in press). See Table 3-4 for bycatch estimates and observed mortality and serious injury for the current 5-year period, and Appendix V for historical bycatch information.

**Mid-Atlantic Bottom Trawl**

The Mid-Atlantic bottom trawl fishery is a Category II fishery. The average annual observed mortality from 2014-2018 was 2 animals, and the average annual total mortality was 23 (CV=0.33). Two gray seal mortalities were observed in this fishery in 2013, 1 in 2014, none in 2015, 3 in 2016 and 5 in 2017 (Lyssikatos et al. 2020 in press). See Table 3-4 for bycatch estimates and observed mortality and serious injury for the current 5-year period, and Appendix V for historical bycatch information.

**Northeast Mid-Water and Pair Trawl**

The Northeast mid-water and pair trawl fisheries are Category II fisheries. Only 1 gray seal was observed in these fisheries from 2014-2018 and an expanded bycatch estimate has not been generated. See Table 4 for observed mortality and serious injury for during the current 5-year period, and Appendix V for historical bycatch information.

**CANADA - Canada**

There is limited information on Canadian fishery bycatch (DFO 2017). Historically, an unknown number of gray seals have been taken in Newfoundland and Labrador, Gulf of St. Lawrence, and Bay of Fundy groundfish gillnets; Atlantic Canada and Greenland salmon gillnets; Atlantic Canada cod traps, and Bay of Fundy herring weirs (Read 1994).
Table 34. Summary of the incidental serious injury and mortality of gray seal (Halichoerus grypus atlantica) by commercial fishery including the years sampled, the type of data used (Data Type), the annual observer coverage (Observer Coverage), the mortalities recorded by on-board observers (Observed Mortality), the estimated annual mortality (Estimated Mortality), the estimated CV of the annual mortality (Estimated CVs) and the mean annual mortality (CV in parentheses).

<table>
<thead>
<tr>
<th>Fishery</th>
<th>Years</th>
<th>Data Type *</th>
<th>Observer Coverage</th>
<th>Observed Serious Injury *</th>
<th>Observed Mortality</th>
<th>Est. Serious Injury</th>
<th>Est. Mortality</th>
<th>Est. Comb. Mortality</th>
<th>Est. CVs</th>
<th>Mean Annual Combine Mortality</th>
</tr>
</thead>
<tbody>
<tr>
<td>Northeast Sink Gillnet</td>
<td>2013</td>
<td>Obs.</td>
<td>0.24</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0.20</td>
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<tr>
<td></td>
<td>2014</td>
<td>Data, Weighout</td>
<td>0.18</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
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<tr>
<td></td>
<td>2015</td>
<td>ut, Trip</td>
<td>0.14</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0.47</td>
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<tr>
<td></td>
<td>2016</td>
<td>Logbook</td>
<td>0.10</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
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</tr>
<tr>
<td></td>
<td>2017</td>
<td></td>
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<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0.32</td>
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<tr>
<td></td>
<td>2018</td>
<td></td>
<td>0.11</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
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<td>0</td>
</tr>
<tr>
<td>Mid-Atlantic Gillnet</td>
<td>2012</td>
<td>Obs. Data, Trip</td>
<td>0.03</td>
<td>0</td>
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<td>0</td>
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<td>0</td>
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<td>2014</td>
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<td>0</td>
<td>0</td>
<td>0.37</td>
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<tr>
<td></td>
<td>2015</td>
<td>Logbook, Allocated Dealer Data</td>
<td>0.06</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
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<tr>
<td></td>
<td>2016</td>
<td></td>
<td>0.08</td>
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<tr>
<td></td>
<td>2017</td>
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<tr>
<td>Northeast Bottom Trawl</td>
<td>2012</td>
<td>Obs. Data, Trip</td>
<td>0.04</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0.32</td>
</tr>
<tr>
<td></td>
<td>2014</td>
<td>Data, Trip</td>
<td>0.04</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0.33</td>
</tr>
<tr>
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<td>2015</td>
<td>Logbook</td>
<td>0.12</td>
<td>0</td>
<td>0</td>
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<td>0</td>
<td>0</td>
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<td>0.33</td>
</tr>
<tr>
<td></td>
<td>2016</td>
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<td>2017</td>
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<td>2018</td>
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<td></td>
</tr>
<tr>
<td>Mid-Atlantic Bottom Trawl</td>
<td>2012</td>
<td>Obs. Data, Trip</td>
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<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0.32</td>
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<tr>
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<td>Data, Trip</td>
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<td>Logbook</td>
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<td></td>
<td>2016</td>
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<td>2018</td>
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</tr>
<tr>
<td>Northeast Mid-water Trawl – Incl. Pair Trawl</td>
<td>2013</td>
<td>Obs. Data, Trip</td>
<td>0.04</td>
<td>0</td>
<td>0</td>
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<td>0</td>
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<td></td>
<td>2014</td>
<td>Data, Trip</td>
<td>0.04</td>
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<tr>
<td></td>
<td>2016</td>
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<td>2017</td>
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<tr>
<td>TOTAL</td>
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<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td>0.34</td>
</tr>
</tbody>
</table>

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).


Other Mortality

U.S. States

Gray seals, like harbor seals, were hunted for bounty in New England waters until the late 1960s (Katona et al. 1993; Lelli et al. 2009). This hunt may have severely depleted this stock in U.S. waters (Rough 1995; Lelli et al. 2009). Other sources of mortality include human interactions, storms, abandonment by the mother, disease, and shark predation. Mortalities caused by human interactions include research mortalities, boat strikes, fishing gear interactions, power plant entainment, oil spill/exposure, harassment, and shooting. Seals entangled in netting are common at haul-out sites in the Gulf of Maine and Southeastern Massachusetts.
Tables 5 and 6 present summaries of from 2013 to 2017, 603 gray seal stranding mortalities were recorded, extending from Maine to North Carolina (Table 4: NOAA National Marine Mammal Health and Stranding Response Database, accessed 23 October 2018). Most stranding mortalities were in Massachusetts, which is the center of gray seal abundance in U.S. waters. Sixty-three (10%) of the total stranding mortalities showed signs of human interaction (17 in 2013, 8 in 2014, 20 in 2015, 1 in 2016 and 17 in 2017). A gray seal is recorded in the stranding database during the 2013 to 2017 period as having been shot—in Maine in 2015. Another gray seal mortality due to shooting in Maine in 2016 was prosecuted by NOAA law enforcement. In an analysis of mortality causes of stranded marine mammals on Cape Cod and southeastern Massachusetts between 2000 and 2006, Bogomolni et al. (2010) reported that 45% of gray seal stranding mortalities were attributed to human interaction.

A UME was declared in November of 2011 that involved at least 137 gray seal stranding mortalities between June 2011 and October 2012 in Maine, New Hampshire, and Massachusetts. The UME was declared closed in February 2013 (https://www.fisheries.noaa.gov/national/marine-life-distress/active-and-closed-unusual-mortality-events).

CANADA

Between 2013-2014 and 2017-2018, the average annual human-caused mortality and serious injury to gray seals in Canadian waters from commercial harvest was is 672-636 per year though more are permitted (up to 60,000 seals/year are permitted) see http://www.dfo-mpo.gc.ca/decisions/fm-2015-gp/atl-001-eng.htm. This included: 243 in 2013, 82 in 2014, 1,381 in 2015, 1,588 in 2016, and 64 in 2017, and 66 in 2018 (DFO 2017, Mike Hammill/Courtney D’aust pers. comm.). In addition, between 2013-2014 and 2017-2018, an average of 3,732-3,078 nuisance animals per year were killed. This included, 3,757 in 2013, and 3,732 annually in 2014–2017 (DFO 2017) and 461 in 2018 based on the total number of licenses that were issued (Courtney D’Aoust, pers. comm). Nuisance animals in 2017 were not available as of March 2019, so the average number of nuisance animals from 2014-2016 were used for 2017. Lastly, DFO took 58 animals in 2013, 83 animals in 2014, 42 animals in 2015, 30 animals in 2016, and 60 animals in 2017, and 96 animals in 2018 for scientific collections, for an annual average of 55-62 animals (DFO 2017, Mike Hammill/Samuel Mongrain pers. comm).

Table 45. Gray seal (Halichoerus grypus atlantica) stranding mortalities along the U.S. Atlantic coast (2013-2014-2017-2018) with subtotals of animals recorded as pups in parentheses.

<table>
<thead>
<tr>
<th>State</th>
<th>2013</th>
<th>2014</th>
<th>2015</th>
<th>2016</th>
<th>2017</th>
<th>2018</th>
<th>Total</th>
</tr>
</thead>
<tbody>
<tr>
<td>ME Maine</td>
<td>9 (4)</td>
<td>3 (1)</td>
<td>5</td>
<td>6(0)</td>
<td>14 (1)</td>
<td>25 (0)</td>
<td>37</td>
</tr>
<tr>
<td>NH</td>
<td>1 (0)</td>
<td>3 (2)</td>
<td>2</td>
<td>0</td>
<td>3 (0)</td>
<td>9 (3)</td>
<td>17</td>
</tr>
<tr>
<td>MA</td>
<td>82 (8)</td>
<td>62 (6)</td>
<td>77 (3)</td>
<td>54(0)</td>
<td>135 (21)</td>
<td>261 (29)</td>
<td>440 (89)</td>
</tr>
<tr>
<td>RI</td>
<td>11 (2)</td>
<td>8 (1)</td>
<td>7 (1)</td>
<td>4(0)</td>
<td>16 (5)</td>
<td>20 (3)</td>
<td>46 (55)</td>
</tr>
<tr>
<td>CT</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>3 (0)</td>
<td>10 (0)</td>
<td>1</td>
</tr>
<tr>
<td>NY</td>
<td>18 (5)</td>
<td>12 (4)</td>
<td>10</td>
<td>1 (1)</td>
<td>57 (0)</td>
<td>25 (1)</td>
<td>82 (105)</td>
</tr>
<tr>
<td>NJ</td>
<td>7 (2)</td>
<td>7 (6)</td>
<td>7 (6)</td>
<td>3 (1)</td>
<td>4 (3)</td>
<td>14 (10)</td>
<td>28 (35)</td>
</tr>
<tr>
<td>DE</td>
<td>0</td>
<td>3 (3)</td>
<td>3 (3)</td>
<td>0</td>
<td>1 (0)</td>
<td>4 (2)</td>
<td>11</td>
</tr>
<tr>
<td>MD</td>
<td>0</td>
<td>1 (0)</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>1 (1)</td>
<td>12</td>
</tr>
<tr>
<td>VA</td>
<td>0</td>
<td>0</td>
<td>3</td>
<td>0</td>
<td>0</td>
<td>1 (1)</td>
<td>14</td>
</tr>
<tr>
<td>NC</td>
<td>0</td>
<td>2 (2)</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>5 (2)</td>
<td>27</td>
</tr>
</tbody>
</table>
Table 6. Documented gray seal (Halichoerus grypus atlantica) human-interaction related stranding mortalities along the U.S. Atlantic coast (2014–2018) by type of interaction. “Fishery interactions” are subsumed in the total estimated mortality calculated from observer data.

<table>
<thead>
<tr>
<th>Cause</th>
<th>2014</th>
<th>2015</th>
<th>2016</th>
<th>2017</th>
<th>2018</th>
<th>Total</th>
</tr>
</thead>
<tbody>
<tr>
<td>Fishery Interaction</td>
<td>2</td>
<td>14</td>
<td>0</td>
<td>10</td>
<td>10</td>
<td>36</td>
</tr>
<tr>
<td>Boat Strike</td>
<td>3</td>
<td>3</td>
<td>0</td>
<td>4</td>
<td>2</td>
<td>12</td>
</tr>
<tr>
<td>Shot</td>
<td>0</td>
<td>1</td>
<td>1</td>
<td>0</td>
<td>0</td>
<td>2</td>
</tr>
<tr>
<td>Human Interaction - Other</td>
<td>3</td>
<td>2</td>
<td>0</td>
<td>3</td>
<td>9</td>
<td>17</td>
</tr>
<tr>
<td>TOTAL</td>
<td>8</td>
<td>20</td>
<td>1</td>
<td>17</td>
<td>21</td>
<td>67</td>
</tr>
</tbody>
</table>

STATUS OF STOCK

Gray seals are not listed as threatened or endangered under the Endangered Species Act, and the western North Atlantic stock is not considered strategic under the Marine Mammal Protection Act. The U.S. portion of 2013–2017 average annual human-caused mortality and serious injury during 2014–2018 in U.S. waters does not exceed the portion of PBR in of the U.S. watersportion of the stocks. The status of the gray seal population relative to OSP in U.S. Atlantic EEZ waters is unknown, but the stock’s abundance appears to be increasing in Canadian and U.S. waters. Total fishery-related mortality and serious injury for this stock is not less than 10% of the calculated PBR and, therefore, cannot be considered to be insignificant and approaching zero mortality and serious injury rate.

Uncertainties described in the above sections could have an effect on the designation of the status of this stock in U.S. waters.

REFERENCES CITED


SPERM WHALE (Physeter macrocephalus):
Northern Gulf of Mexico Stock

STOCK DEFINITION AND GEOGRAPHIC RANGE

Sperm whales are found throughout the world's oceans in deep waters from the tropics to the edge of the ice at both poles (Leatherwood and Reeves 1983; Rice 1989; Whitehead 2002). Sperm whales were commercially hunted in the Gulf of Mexico by American whalers from sailing vessels until the early 1900s (Townsend 1935; Reeves et al. 2011). In the northern Gulf of Mexico (i.e., U.S. Gulf of Mexico), systematic aerial and ship surveys indicate that sperm whales inhabit continental slope and oceanic waters where they are widely distributed (Figure 1; Fulling et al. 2003; Mullin and Fulling 2004; Mullin et al. 2004; Maze-Foley and Mullin 2006; Mullin 2007; Garrison and Aichinger Dias 2020). Seasonal aerial surveys confirm that sperm whales are present in the northern Gulf of Mexico in all seasons (Mullin et al. 1994; Hansen et al. 1996; Mullin and Hoggard 2000).

Figure 1. Distribution of sperm whale on-effort sightings from SEFSC vessel surveys during spring 1996-2001, summer 2003, spring 2004, summer 2009, summer 2017, and summer/fall 2018. Isobaths are the 200-m, 1,000-m, and 2,000-m depth contours. The darker line indicates the U.S. EEZ.

All the cetacean species found in the oceanic northern Gulf of Mexico almost certainly occur in similar habitat beyond U.S. boundaries in the southern Gulf. There are fewer cetacean sighting and stranding records in the southern Gulf due to more limited effort. Nevertheless there are records for most oceanic species in the southern Gulf (e.g., Jefferson and Schiro 1997; Ortega Ortiz 2002; Ortega-Argueta et al. 2005; Jefferson et al. 2008; Vázquez Castán et
Sperm whales throughout the world exhibit a geographic social structure where females and juveniles of both sexes occur in mixed groups and inhabit tropical and subtropical waters. Males, as they mature, initially form bachelor groups but eventually become more socially isolated and more wide-ranging, inhabiting temperate and polar waters as well (Whitehead 2003). While this pattern also applies to the Gulf of Mexico, results of multi-disciplinary research conducted in the Gulf since 2000 confirms speculation by Schmidly (1981) and indicates clearly that Gulf of Mexico sperm whales constitute a stock that is distinct from other Atlantic Ocean stocks(s) (Mullin et al. 2003; Jaquet 2006; Jochens et al. 2008). The following summarizes the most significant stock structure-related findings from the Sperm Whale Seismic Study (Jochens et al. 2008) and associated projects. Measurements of the total length of Gulf of Mexico sperm whales indicate that they are 1.5-2.0 m smaller on average compared to whales measured in other areas (Jochens et al. 2008). Female/immature group size in the Gulf is about one-third to one-fourth that found in the Pacific Ocean but more similar to group sizes in the Caribbean (Richter et al. 2008; Jaquet and Gendron 2009). Tracks from 39 whales satellite tagged in the northern Gulf were monitored for up to 607 days. No discernable seasonal migrations were made, but Gulf-wide movements primarily along the northern Gulf slope did occur. The tracks showed that whales exhibit a range of movement patterns within the Gulf, including movement into the southern Gulf in a few cases, but that only one whale (a male) left the Gulf of Mexico (Jochens et al. 2008). This animal moved into the North Atlantic and then back into the Gulf after about two months. Additionally, no matches were found when 285 individual whales photo-identified from the Gulf and about 2500 from the North Atlantic and Mediterranean Sea were compared (Jochens et al. 2008).

More recently, Gero et al. (2007) also suggested that movements of sperm whales between the adjacent areas of the Caribbean Sea, Gulf of Mexico and Atlantic may not be common. No matches were made from animals photo-identified in the eastern Caribbean Sea (islands of Dominica, Guadeloupe, Grenada, St. Lucia and Martinique) with either animals from the Sargasso Sea or the Gulf of Mexico. Engelhaupt et al. (2009) conducted an analysis of matrilineally inherited mitochondrial DNA and found significant genetic differentiation between animals from the northern Gulf of Mexico and those from the western North Atlantic Ocean, North Sea and Mediterranean Sea. Analysis of biparentally inherited nuclear DNA showed no significant difference between whales sampled in the Gulf and those from the other areas of the North Atlantic, suggesting that while females show strong philopatry to the Gulf, male-mediated gene flow between the Gulf and North Atlantic Ocean may be occurring (Engelhaupt et al. 2009).

Sperm whales make vocalizations called “codas” that have distinct patterns and are apparently culturally transmitted (Watkins and Schevill 1977; Whitehead and Weilgart 1991; Rendell and Whitehead 2001), and based on degree of social affiliation, mixed groups of sperm whales (mixed-sex groups of females/immatures) worldwide can be placed in recognizable acoustic clans (Rendell and Whitehead 2003). Recordings from mixed groups in the Gulf of Mexico compared to those from other areas of the Atlantic indicated that Gulf sperm whales constitute a distinct acoustic clan that is rarely encountered outside of the Gulf. It is assumed from this that groups from other clans enter the northern Gulf only infrequently (Gordon et al. 2008). Antunes (2009) used additional data to further examine variation in sperm whale coda repertoires in the North Atlantic Ocean, and found that variation in the North Atlantic is mostly geographically structured as coda patterns were unique to certain regions and a significant negative correlation was found between coda repertoire similarities and geographic distance. His work also suggested sperm whale codas differed between the Gulf of Mexico and the North Atlantic.

Thus, there are now multiple lines of evidence supporting delimitation of separate Gulf of Mexico and western North Atlantic stocks of sperm whales. However, there are insufficient data to determine whether the northern Gulf of Mexico stock comprises multiple demographically independent populations. Additional morphological, acoustic, genetic, and/or behavioral data are needed to further delineate population structure within the Gulf of Mexico and across the broader geographic area.

**POPULATION SIZE**
The best abundance estimate (Nbest) for the northern Gulf of Mexico sperm whale is 1,180 (CV=0.22; Table 1). This estimate is from summer 2017 and summer/fall 2018 oceanic surveys covering waters from the 200-m isobath to the seaward extent of the U.S. EEZ (Garrison et al. 2020). The best abundance estimate available for northern Gulf of Mexico sperm whales is 763 (CV=0.38; Table 1). This estimate is from a summer 2009 oceanic survey covering waters from the 200-m isobath to the seaward extent of the U.S. EEZ.

Earlier abundance estimates

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

Recent surveys and abundance estimates

An abundance estimate for sperm whales was generated from vessel surveys conducted in the northern Gulf of Mexico from the continental shelf edge (~200-m isobath) to the seaward extent of the U.S. EEZ (Garrison et al. 2020). One survey was conducted from 2 July to 25 August 2017 and consisted of 7,302 km of on-effort trackline, and the second survey was conducted from 11 August to 6 October 2018 and consisted of 6,473 km of on-effort trackline within the surveyed strata. Both surveys used a double-platform data-collection procedure, which allowed estimation of the detection probability on the trackline using the independent observer approach assuming point independence (Laake and Borchers 2004). Abundance was calculated using mark-recapture distance sampling implemented in package mrds (version 2.21, Laake et al. 2020) in the R statistical programming language. This approach accounted for the effects of covariates (e.g., sea state, glare) on detection probability within the surveyed strip. The surveys were conducted in "passing mode" (e.g., Schwarz et al. 2010) while all prior surveys in the Gulf of Mexico have been conducted in "closing mode." The abundance estimate for this stock included sightings of unidentified large whales that were apportioned among identified species based on their relative density within the survey strata (Garrison et al. 2020). The inverse variance weighted mean abundance estimate for sperm whales in oceanic waters during 2017 and 2018 is 1,180 (CV=0.22; Table 1; Garrison et al. 2020). Unlike previous abundance estimates, this estimate was corrected for the probability of detection on the trackline. There may be a portion of the detection probability that is not accounted for due to long dive times. During summer 2009, a line transect survey dedicated to estimating the abundance of oceanic cetaceans was conducted in the northern Gulf of Mexico. Survey lines were stratified in relation to depth and the location of the Loop Current. The abundance estimate for sperm whales in oceanic waters during 2009 was 763 (CV=0.38; Table 1).

Table 1. Most recent abundance estimate (Nbest) and coefficient of variation (CV) of northern Gulf of Mexico sperm whales in oceanic waters (200 m to the offshore extent of the EEZ) based on an average from summer 2017 and summer/fall 2018 vessel surveys.

<table>
<thead>
<tr>
<th>Years</th>
<th>Area</th>
<th>Nbest</th>
<th>CV</th>
</tr>
</thead>
<tbody>
<tr>
<td>2017, 2018</td>
<td>Gulf of Mexico</td>
<td>1,180</td>
<td>0.22</td>
</tr>
</tbody>
</table>

Table 1: Summary of abundance estimates for northern Gulf of Mexico sperm whales. Month, year and area covered during each abundance survey, and resulting abundance estimate (N_{best}) and coefficient of variation (CV).

<table>
<thead>
<tr>
<th>Month/Year</th>
<th>Area</th>
<th>N_{best}</th>
<th>CV</th>
</tr>
</thead>
<tbody>
<tr>
<td>Apr-Jun 1991-1994</td>
<td>Oceanic waters</td>
<td>530</td>
<td>0.34</td>
</tr>
<tr>
<td>Apr-Jun 1996-2001(excluding 1998)</td>
<td>Oceanic waters</td>
<td>1,240</td>
<td>0.23</td>
</tr>
</tbody>
</table>
Minimum Population Estimate

The minimum population estimate (Nmin) is the lower limit of the two-tailed 60% confidence interval of the log-normal distributed abundance estimate. This is equivalent to the 20th percentile of the log-normal distributed abundance estimate as specified by Wade and Angliss (1997). The best estimate of abundance for sperm whales is 1,180 (CV=0.2238). The minimum population estimate for the northern Gulf of Mexico sperm whale stock is 983 (Table 2). Sperm whales.

Current Population Trend

The statistical power to detect a trend in abundance for this stock is poor due to the relatively imprecise abundance estimates and long intervals between surveys. For example, the power to detect a precipitous decline in abundance (i.e., 50% decrease in 15 years) with estimates of low precision (e.g., CV>0.30) remains below 80% (alpha=0.30) unless surveys are conducted on an annual basis (Taylor et al. 2007). In addition, because these surveys are restricted to U.S. waters, it is not possible to distinguish between changes in population size and Gulf-wide shifts in spatial distribution. Five point estimates of sperm whale abundance have been made based on data from surveys during: 2003 (June–August), 2004 (April–June), 2009 (July–August), 2017 (July–August), and 2018 (August–October). Each of these surveys had a similar design and was conducted using the same vessel or a vessel with a similar observation platform. Prior year surveys only employed a single survey team. The estimates for 2017 and 2018 were conducted in "passing" mode which resulted in increased numbers of unidentified sightings and may have affected group size estimation for distant groups of dolphins and small whales. Apportioning unidentified species among identified taxa addresses the first issue, and there were no apparent relationships between sighting distance and estimated group size. The model for detection probability on the trackline from the summer 2017 survey was applied to the abundance estimates from the 2003, 2004, and 2009 surveys. The resulting abundance estimates were: 2003, N=2,542 (CV=0.34); 2004, N=1,686 (CV=0.41); 2009, N=2,096 (CV=0.55); 2017, N=1,078 (CV=0.29); and 2018, N=1,307 (CV=0.33). Pairwise comparisons of the log-transformed means were conducted between years, and significant differences were assessed at alpha=0.10. P-values were adjusted for multiple comparisons. There were no significant differences between survey years. A trend analysis has not been conducted for this stock. Four point estimates of sperm whale abundance have been made based on data from surveys covering 1991–2009. (Table 1). The estimates vary by a maximum factor of 3.1. To determine whether changes in abundance have occurred over this period, an analysis of all the survey data needs to be conducted which incorporates covariates (e.g., survey conditions, season) that could potentially affect estimates. It should be noted that since this is a transboundary stock and the abundance estimates are for U.S. waters only, it will be difficult to interpret any detected trends.

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 lifespan (Barlow et al. 1995).

POTENTIAL BIOLOGICAL REMOVAL

Potential Biological Removal (PBR) is the product of the minimum population size, one half the maximum net productivity rate and a recovery factor (MMPA Sec. 3.16 U.S.C. 1362; Wade and Angliss 1997). The minimum population size is 983 (CV=0.329). The maximum productivity rate is 0.04, the default value for cetaceans. The recovery factor is 0.1 because the sperm whale is an endangered species. PBR for the northern Gulf of Mexico sperm whale is 2.0 (Table 2).
The estimated mean annual fishery-related mortality and serious injury for this stock during 2014–2018 was 0.2 sperm whales (CV=1.00) due to interactions with the large pelagics longline fishery (see Fisheries Information sections below; Tables 3–4). Mean annual mortality and serious injury during 2014–2018 due to other human-caused actions (the Deepwater Horizon oil spill) was predicted to be 9.4. The minimum total mean annual human-caused mortality and serious injury for this stock during 2014–2018 was, therefore, 9.6. The total human-caused mortality and serious injury for sperm whales in the northern Gulf of Mexico during 2009–2013 was 0.

Table 3. Total annual estimated fishery-related mortality and serious injury for the northern Gulf of Mexico sperm whale.

<table>
<thead>
<tr>
<th>Years</th>
<th>Source</th>
<th>Annual Avg.</th>
<th>CV</th>
</tr>
</thead>
<tbody>
<tr>
<td>2014–2018</td>
<td>U.S. fisheries using observer data</td>
<td>0.2</td>
<td>1.00</td>
</tr>
</tbody>
</table>

New Serious Injury Guidelines

NMFS updated its serious injury designation and reporting process, which uses guidance from previous serious injury workshops, expert opinion, and analysis of historic injury cases to develop new criteria for distinguishing serious from non-serious injury (Angliss and DeMaster 1998; Anderson et al. 2008; NOAA 2012). NMFS defines serious injury as an “injury that is more likely than not to result in mortality.” Injury determinations for stock assessments revised in 2013 or later incorporate the new serious injury guidelines, based on the most recent 5-year period for which data are available.

Fisheries Information

There are two commercial fisheries that interact, or that potentially could interact, with this stock in the Gulf of Mexico. These are the Category I Atlantic Highly Migratory Species (high seas) longline fishery and the Category I Atlantic Ocean, Caribbean, Gulf of Mexico large pelagics longline fishery (Appendix III). Percent observer coverage (percentage of sets observed) for these longline fisheries for each year during 2014–2018 was 18, 19, 23, 13 and 20, respectively.

There is very little effort within the Gulf of Mexico by the Atlantic Highly Migratory Species (high seas) longline fishery, and no takes of sperm whales within high seas waters of the Gulf of Mexico have been observed or reported thus far. The commercial fishery that interacts with this stock in the Gulf of Mexico is the Atlantic Ocean, Caribbean, Gulf of Mexico large pelagics longline fishery (Appendix III). Pelagic swordfish, tunas and billfish are the targets of the large pelagics longline fishery operating in the northern Gulf of Mexico. During the second quarter of 2015, one sperm whale was observed to be seriously injured (Garrison and Stokes 2017). The average annual serious injury and mortality in the Gulf of Mexico pelagic longline fishery for the five-year period from 2014 to 2018 is 0.2 (CV=1.00; Table 4; Garrison and Stokes 2016; 2017; 2019; 2020; in review). There have been no reports of mortality or serious injury to sperm whales by this fishery in recent years (2009–2013) or historically 1998–2008 (Yeung 1999; 2001; Garrison 2003; Garrison and Richards 2004; Garrison 2005; Fairfield-Walsh and Garrison 2006; Fairfield-Walsh and Garrison 2007; Fairfield and Garrison 2008; Garrison et al. 2009; Garrison and Stokes 2010; 2012a,b; 2013; 2014). However, in 2008 during quarter 2, there was an entanglement and live release without serious injury of 1 sperm whale (Garrison et al. 2009). The whale was entangled in mainline and other gear and was accompanied by a calf. The mainline broke when the whale dove and gear remained on the animal; however, since it was a large whale it was not considered seriously injured (Garrison and Stokes 2008). This was the first observed interaction between a sperm whale and this fishery.

During 15 April – 15 June 2008, and also subsequently during the first and second quarters (15 April – 15 June) of 20092014–20182013, observer coverage in the Gulf of Mexico pelagic longline fishery was greatly enhanced (approaching 55%)—to collect more robust information on the interactions between pelagic longline vessels and spawning bluefin tuna. Therefore, the high annual observer coverage rates during 2014–20182008–2013 primarily reflect high coverage rates during the first and second quarters of each year. During these second quarters, this elevated...
coverage results in an increased probability that relatively rare interactions will be detected. Species within the oceanic Gulf of Mexico are presumed to be resident year-round; however, it is unknown if the bycatch rates observed during the first and second quarter are representative of that which occurs throughout the year.

A commercial fishery for sperm whales operated in the Gulf of Mexico in deep waters between the Mississippi River delta and DeSoto Canyon during the late 1700s to the early 1900s (Mullin et al. 1994), but the exact number of whales taken is not known (Townsend 1935; Lowery 1974). A commercial fishery for sperm whales operated in the Gulf of Mexico during the late 1700s to the late 1800s (Reeves et al. 2011), but the exact number of whales taken is not known (Townsend 1935; Lowery 1974). Reeves et al. (2011) estimated the number of sperm whales removed from the Gulf during the 1780s–1870s as 1,179 (SE=224). Townsend (1935) reported many records of sperm whales from April through July in the north-central Gulf (Petersen and Hoggard 1996).

Table 4. Summary of the incidental mortality and serious injury of sperm whales by the pelagic longline commercial fishery including the years sampled (Years), the type of data used (Data Type), the annual observer coverage (Observer Coverage), the annual observed serious injury and mortality recorded by on-board observers, the annual estimated serious injury and mortality, the combined annual estimates of serious injury and mortality (Est. Combined Mortality), the estimated CV of the combined annual mortality estimates (Est. CVs), the mean of the combined annual mortality estimates, and the CV of the mean combined annual mortality estimate (CV of Mean).

<table>
<thead>
<tr>
<th>Fishery</th>
<th>Years</th>
<th>Data Type</th>
<th>Observer Coverage</th>
<th>Observed Serious Injury</th>
<th>Observed Mortality</th>
<th>Estimated Serious Injury</th>
<th>Est. Mort.</th>
<th>Est. Combined Mortality</th>
<th>Est. CVs</th>
<th>Mean Combined Annual Mortality</th>
<th>CV of Mean</th>
</tr>
</thead>
<tbody>
<tr>
<td>Pelagic Longline</td>
<td>2014</td>
<td>Obs. Data</td>
<td>0.18</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>-</td>
<td>0.2</td>
</tr>
<tr>
<td></td>
<td>2015</td>
<td>Trip Logbook</td>
<td>0.19</td>
<td>1</td>
<td>0</td>
<td>0.94</td>
<td>0</td>
<td>0.94</td>
<td>1</td>
<td>-</td>
<td>0.2</td>
</tr>
<tr>
<td></td>
<td>2016</td>
<td></td>
<td>0.23</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>-</td>
<td>0.2</td>
</tr>
<tr>
<td></td>
<td>2017</td>
<td></td>
<td>0.13</td>
<td>0</td>
<td>0</td>
<td>0.94</td>
<td>0</td>
<td>0.94</td>
<td>1</td>
<td>-</td>
<td>0.2</td>
</tr>
<tr>
<td></td>
<td>2018</td>
<td></td>
<td>0.20</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>-</td>
<td>0.2</td>
</tr>
<tr>
<td>TOTAL</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td>0.2</td>
<td>1.00</td>
</tr>
</tbody>
</table>

Other Mortality

There were seven sperm whale strandings in the northern Gulf of Mexico during 2009–2014–2018 (NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 21 May 2019). There was evidence of human interaction for one stranding (healed scarring). No evidence of human interaction was detected for one stranding, and for the remaining five strandings it could not be determined if there was evidence of human interaction. It could not be determined if there was evidence of human interaction for any of the 8 stranded animals. Stranding data probably underestimate the extent of human and fishery-related mortality and serious injury, particularly for offshore species such as sperm whales, because not all of the whales that die or are seriously injured in human interactions wash ashore, or, if they do, they are not all recovered (Peltier et al. 2012; Wells et al. 2015). In particular, oceanic stocks in the Gulf of Mexico are less likely to strand than nearshore coastal stocks or shelf stocks (Williams et al. 2011). Additionally, not all carcasses will show evidence of human interaction, entanglement or other fishery-related interaction due to decomposition, scavenger damage, etc. (Byrd et al. 2014). Finally, the level of technical expertise among stranding network personnel varies widely as does the ability to recognize signs of human interaction.

An Unusual Mortality Event (UME) was declared for cetaceans in the northern Gulf of Mexico beginning 1 February 2010 and ending 31 July 2014; and, as of September 2014, the event is still ongoing (Litz et al. 2014; https://www.fisheries.noaa.gov/national/marine-life-distress/2010-2014-cetacean-unusual-mortality-event-northern-gulf-mexico). It includes cetaceans that stranded prior to the Deepwater Horizon (DWH) oil spill (see “Habitat Issues” below), during the spill, and after. Exposure to the DWH oil spill was determined to be the primary underlying cause of the elevated stranding numbers in the northern Gulf of Mexico after the spill (e.g., Schwacke et al. 2014;
A population model was developed to estimate the injury and time to recovery for stocks affected by the DWH oil spill, taking into account long-term effects resulting from mortality, reproductive failure, reduced survival rates, and the proportion of the stock exposed to DWH oil (DWH MMIQT 2015). Overall, the model estimated that this stock experienced a 7% maximum reduction in population size due to the oil spill (DWH MMIQT 2015). The mortality projected for the years 2010–2013 due to the spill has not been reported previously. Based on the population model, it was projected that 94 sperm whales died during 2010–2013 (four year annual average of 24) due to elevated mortality associated with oil exposure (see Appendix VI). For the 2014–2018 reporting period of this SAR, the population model estimated 47 sperm whales died due to elevated mortality associated with oil exposure. The population model used to predict sperm whale mortality due to the DWH event has a number of sources of uncertainty. Model parameters (e.g., survival rates, reproductive rates, and life-history parameters) were derived from literature sources for sperm whales occupying waters outside of the Gulf of Mexico. Proxy values for the effects of DWH oil exposure on both survival rates and reproductive success were applied based upon estimated values for common bottlenose dolphins in Barataria Bay. Also, there was no estimation of uncertainty in model parameters or outputs.

Ship strikes to whales occur worldwide and are a source of injury and mortality. No vessel strikes have been documented in recent years (2009–2013) for sperm whales in the Gulf of Mexico. Historically, 1 possible sperm whale mortality due to a vessel strike has been documented for the Gulf of Mexico. The incident occurred in 1990 in the vicinity of Grande Isle, Louisiana. Deep cuts on the dorsal surface of the whale indicated the ship strike was probably pre-mortem (Jensen and Silber 2004).

**Table 5. Sperm whale strandings along the northern Gulf of Mexico coast, 2014–2018. Data are from the NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 21 May 2019.**

<table>
<thead>
<tr>
<th>State</th>
<th>2014</th>
<th>2015</th>
<th>2016</th>
<th>2017</th>
<th>2018</th>
<th>Total</th>
</tr>
</thead>
<tbody>
<tr>
<td>Alabama</td>
<td>0</td>
<td>0</td>
<td>1</td>
<td>0</td>
<td>0</td>
<td>1</td>
</tr>
<tr>
<td>Florida</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>1</td>
</tr>
<tr>
<td>Louisiana</td>
<td>0</td>
<td>0</td>
<td>2</td>
<td>1</td>
<td>0</td>
<td>3</td>
</tr>
<tr>
<td>Texas</td>
<td>1</td>
<td>0</td>
<td>1</td>
<td>0</td>
<td>0</td>
<td>2</td>
</tr>
<tr>
<td>TOTAL</td>
<td>1</td>
<td>0</td>
<td>5</td>
<td>1</td>
<td>0</td>
<td>7</td>
</tr>
</tbody>
</table>

**HABITAT ISSUES**

The *Deepwater Horizon* (DWH) MC252 drilling platform, located approximately 50 miles (80 km) southeast of the Mississippi River Delta in waters about 1,500 m deep, exploded on 20 April 2010. The rig sank, and over 87 days up to ~3.2 million barrels of oil were discharged from the wellhead until it was capped on 15 July 2010 (McNutt et al. 2012; DWH NRDAT 2016). During the response effort dispersants were applied extensively at the seafloor and at the sea surface (Lehr et al. 2010; OSAT 2010). In-situ burning, or controlled burning of oil at the surface, was also used extensively as a response tool (Lehr et al. 2010). The oil, dispersant and burn residue compounds present ecological concerns (Buist et al. 1999; NOAA 2011). The magnitude of this oil spill was unprecedented in U.S. history, causing impacts to wildlife, natural habitats and human communities along coastal areas from western Louisiana to the Florida Panhandle (NOAA 2011). It could be years before the entire scope of damage is ascertained (NOAA 2011).

Shortly after the oil spill, the Natural Resource Damage Assessment (NRDA) process was initiated under the Oil Pollution Act of 1990. A variety of NRDA research studies were conducted to determine potential impacts of the spill on marine mammals. These studies estimated that 16% (95% CI: 11–23) of sperm whales in the Gulf were exposed to oil, that 7% (95% CI: 3–10) of females suffered from reproductive failure, and 6% (95% CI: 2–9) of sperm whales suffered adverse health effects (DWH MMIQT 2015). A population model estimated the stock experienced a maximum 7% reduction in population size (see Other Mortality section above). These studies have focused on identifying the type, magnitude, severity, length and impact of oil exposure to oceanic, continental shelf, coastal and estuarine marine mammals. For continental shelf and oceanic cetaceans, the NOAA-led efforts include: aerial surveys to document the distribution, abundance, species and exposure relative to oil from the DWH spill; and ship surveys to...
evaluate exposure to oil and other chemicals and to assess changes in animal behavior and distribution relative to oil exposure through visual and acoustic surveys, deployment of passive acoustic monitoring systems, collection of tissue samples, and deployment of satellite tags on sperm and Bryde’s whales.

Vessel and aerial surveys documented sperm whales, bottlenose dolphins, Atlantic spotted dolphins, rough-toothed dolphins, spinner dolphins, pantropical spotted dolphins, Risso’s dolphins, striped dolphins, dwarf pygmy sperm whales and a Cuvier’s beaked whale swimming in oil or potentially oil-derived substances (e.g., sheen, mousse) in the offshore waters of the northern Gulf of Mexico following the DWH oil spill. The effects of oil exposure on marine mammals depend on a number of factors including the type and mixture of chemicals involved; the amount, frequency and duration of exposure; the route of exposure (inhaled, ingested, absorbed, or external); and biomedical risk factors of the particular animal (Geraci 1990). In general, direct external contact with petroleum compounds or dispersants with skin may cause skin irritation, chemical burns and infections. Inhalation of volatile petroleum compounds or dispersants may irritate or injure the respiratory tract, which could lead to pneumonia or inflammation. Ingestion of petroleum compounds may cause injury to the gastrointestinal tract, which could affect an animal’s ability to digest or absorb food. Absorption of petroleum compounds or dispersants may damage kidney, liver and brain function in addition to causing immune suppression and anemia. Long-term chronic effects such as lowered reproductive success and decreased survival may occur (Geraci 1990).

Anthropogenic sound in the world’s oceans has been shown to affect marine mammals, with vessel traffic, seismic surveys, and active naval sonars being the main anthropogenic contributors to low- and mid-frequency noise in oceanic waters (e.g., Nowacek et al. 2015; Gomez et al. 2016; NMFS 2018). Seismic vessel operations in the Gulf of Mexico (commercial and academic) now operate with marine mammal observers as part of required mitigation measures. There have been no reported seismic-related or industry ship-related mortalities or injuries to sperm whales. However, disturbance by anthropogenic noise may prove to be an important habitat issue in some areas of this population’s range, notably in areas of oil and gas activities and/or where shipping activity is high. Results from very limited studies of northern Gulf of Mexico sperm whale responses to seismic exploration indicate that sperm whales do not appear to exhibit horizontal avoidance of seismic survey activities (Miller et al. 2009; Winsor et al. 2017). Data did suggest there may be some decrease in foraging effort during exposure to full-array airgun firing, at least for some individuals. Further study is needed as sample sizes are insufficient at this time (Miller et al. 2009). Farmer et al. (2018a) developed a bio-energetics model to examine the consequences of frequent disruptions to foraging on sperm whales. The simulations suggested that frequent and severe disruptions could lead to terminal starvation. A follow-up study examined the population level effects of acoustic disturbance in combination with the impacts of the DWH oil spill and suggested that acoustic disturbance could have significant population effects, though terminal starvation and fetal abortions were unlikely (Farmer et al. 2018b). Impacts on marine mammal prey from sound are also possible (Carroll et al. 2017), but the duration and severity of any such prey effects on marine mammals are unknown.

Vessel strikes to whales occur world-wide and are a source of injury and mortality. No vessel strikes have been documented in recent years (2014–2018) for sperm whales in the Gulf of Mexico. Historically, one possible sperm whale mortality due to a vessel strike was documented for the Gulf of Mexico. The incident occurred in 1990 in the vicinity of Grande Isle, Louisiana. Deep cuts on the dorsal surface of the whale indicated the vessel strike was probably pre-mortem (Jensen and Silber 2004).

STATUS OF STOCK

The sperm whale is listed as endangered under the Endangered Species Act, and therefore the northern Gulf of Mexico stock is considered strategic under the MMPA. In addition, the mean modeled annual human-caused mortality and serious injury due to the DWH oil spill exceeds PBR for this stock. Total human-caused mortality and serious injury for this stock during 2009–2014, 2012–2013 was 0. Total fishery-related mortality and serious injury for this stock is not less than 10% of the calculated PBR and, therefore, cannot be considered to be insignificant and approaching zero mortality and serious injury rate. The total fishery-related mortality and serious injury for this stock is insignificant and approaching zero mortality and serious injury rate. The status of sperm whales in the northern Gulf of Mexico, relative to OSP, is unknown. There was no statistically significant trend in population size for this stock in the northern Gulf of Mexico. There are insufficient data to determine the population trends for this stock.

REFERENCES CITED


Mullin, K.D. 2007. Abundance of cetaceans in the oceanic northern Gulf of Mexico from 2003 and 2004 ship surveys. NOAA Southeast Fisheries Science Center, 3209 Frederic Street, Pascagoula, Mississippi 39567. PRBD Contribution #PRBD-2016-03. 27 pp.


BRYDE’S WHALE (Balaenoptera edeni): Northern Gulf of Mexico Stock

STOCK DEFINITION AND GEOGRAPHIC RANGE

Bryde's whales are distributed worldwide in tropical and sub-tropical waters, but the taxonomy and number of species and/or subspecies of Bryde’s whales in the world is currently a topic of debate (Kato and Perrin 2009; Rosel and Wilcox 2014). In the western Atlantic Ocean, Bryde's whales are reported from the Gulf of Mexico and the southern West Indies to Cabo Frio, Brazil (Leatherwood and Reeves 1983), but which subspecies the whales belong to in these different areas is unknown. Sighting records and acoustic detections of Bryde's whales in the northern Gulf of Mexico (i.e., U.S. Gulf of Mexico) occur almost exclusively in the northeastern Gulf in the De Soto Canyon area, along the continental shelf break between 100 m and 400 m depth, with a single sighting at 408 m (Figure 1; Hansen et al. 1996; Mullin and Hoggard 2000; Mullin and Fulling 2004; Maze-Foley and Mullin 2006; Rice et al. 2014; Rosel and Wilcox 2014; Širović et al. 2014; Rosel et al. 2016; Soldevilla et al. 2017). Bryde's whales have been sighted in all seasons within the De Soto Canyon area (Mullin and Hoggard 2000; Maze-Foley and Mullin 2006; Mullin 2007; DWH MMIQT, 2015). Genetic analysis suggests that Bryde’s whales from the northern Gulf of Mexico represent a unique evolutionary lineage distinct from other recognized Bryde’s whale subspecies, including those found in the southern Caribbean and southwestern Atlantic off Brazil (Rosel and Wilcox 2014). The geographic distribution of this Bryde’s whale form has not yet been fully identified. Two strandings from the southeastern U.S. Atlantic coast share the same genetic characteristics with those from the northern Gulf of Mexico (Rosel and Wilcox 2014), but it is unclear whether these are extralimital strays (Mead 1977) or whether they indicate the population extends from the northeastern Gulf of Mexico to the Atlantic coast of the southern U.S. (Rosel and Wilcox 2014). There have been no confirmed sightings of Bryde’s whales along the U.S. east coast during NMFS cetacean surveys (Rosel et al. 2016).

Historical whaling records from the 1800s suggest Bryde’s whales may have been more common in the U.S. waters of the north central Gulf of Mexico and in the southern Gulf of Mexico in the Bay of Campeche (Reeves et al. 2011). How regularly they currently use U.S. waters of the western Gulf of Mexico is unknown. There has been only one confirmed sighting of a Gulf of Mexico Bryde’s whale in this region, a whale observed during a 2017 NMFS vessel survey off Texas, despite substantial NMFS survey effort in the north central and western Gulf. There have been no confirmed sightings of Bryde's whales in U.S. waters of the western Gulf of Mexico during NMFS cetacean surveys (Rosel et al. 2016).
no confirmed sightings in the north central or western Gulf despite NMFS survey effort in the area dating back to the early 1990s (e.g., Hansen et al. 1996; Mullin and Hoggard 2000; Mullin and Fulling 2004; Maze-Foley and Mullin 2006). A compilation of available records of cetacean sightings, strandings, and captures in Mexican waters of the southern Gulf of Mexico identified no Bryde’s whales (Ortega-Ortiz 2002). There are insufficient data to determine whether it is plausible the stock contains multiple demographically independent populations that should be separate stocks.

**POPULATION SIZE**

The best abundance estimate available for northern Gulf of Mexico Bryde’s whales in the northern Gulf of Mexico is 5133 (CV=0.50; Table 1). This estimate is from summer 2017 and summer/fall 2018 oceanic surveys covering waters from the 200-m isobath to the seaward extent of the U.S. EEZ (Garrison et al. 2020). This estimate is from a summer 2009 oceanic survey covering waters from the 200 m isobath to the seaward extent of the U.S. EEZ.

**Earlier abundance estimates**

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

**Recent surveys and abundance estimates**

An abundance estimate for Bryde’s whales was generated from vessel surveys conducted in the northern Gulf of Mexico from the continental shelf edge (~200-m isobath) to the seaward extent of the U.S. EEZ (Garrison et al. 2020). One survey was conducted from 2 July to 25 August 2017 and consisted of 7,302 km of on-effort trackline, and the second survey was conducted from 11 August to 6 October 2018 and consisted of 6,473 km of on-effort trackline. The surveys were conducted in “passing mode” (e.g., Schwarz et al. 2010) while all prior surveys in the Gulf of Mexico have been conducted in "closing mode." Both surveys used a double-platform data-collection procedure to allow estimation of the detection probability on the trackline using the independent observer approach assuming point independence (Laake and Borchers 2004). Due to the restricted habitat range of Gulf of Mexico Bryde's whales, survey effort was re-stratified to include only effort within their core habitat area (Figure 1; https://www.fisheries.noaa.gov/resource/map/gulf-mexico-brydes-whale-core-distribution-area-map-gis-data) including 941 km of effort in 2017 and 848 km of effort in 2018. In addition, there was an insufficient number of Bryde's whale sightings during these surveys to develop an appropriate detection probability function. Therefore, a detection function was derived based on 91 sightings of Bryde's whale groups observed during SEFSC large vessel surveys between 2003 and 2019. The abundance estimates include unidentified large whales and baleen whales observed within the Bryde's whale habitat. However, the estimate does not include the sighting of a confirmed Bryde's whale in the western Gulf of Mexico in 2017. It is not possible to extrapolate estimated density beyond the core area since little is known about habitat use and distribution outside of this area. Estimates of abundance were derived using MCDS distance sampling methods that account for the effects of covariates (e.g., sea state, glare) on detection probability within the surveyed strip (Thomas et al. 2010) implemented in package mrds (version 2.21, Laake et al. 2020) in the R statistical programming language. The inverse variance weighted mean abundance for Bryde's whales in oceanic waters during 2017 and 2018 was 51 (CV=0.50; Table 1; Garrison et al. 2020). This estimate was not corrected for the probability of detection on the trackline because there was only one resighting and few sightings overall of Bryde's whales during the two-team surveys. During summer 2009, a line transect survey dedicated to estimating the abundance of oceanic cetaceans was conducted in the northern Gulf of Mexico (Garrison 2016). Survey lines were stratified in relation to depth and the location of the Loop Current. The abundance estimate for Bryde’s whales in oceanic waters during 2009 was 33 (CV=1.07; Table 1).

<table>
<thead>
<tr>
<th>Years</th>
<th>Area</th>
<th>Nbest</th>
<th>CV</th>
</tr>
</thead>
<tbody>
<tr>
<td>2017, 2018</td>
<td>Gulf of Mexico</td>
<td>51</td>
<td>0.50</td>
</tr>
</tbody>
</table>
Table 1. Summary of abundance estimates for northern Gulf of Mexico Bryde’s whales. Month, year, and area covered during each abundance survey, and resulting abundance estimate (Nbest) and coefficient of variation (CV).

<table>
<thead>
<tr>
<th>Month/Year</th>
<th>Area</th>
<th>Nbest</th>
<th>CV</th>
</tr>
</thead>
<tbody>
<tr>
<td>Apr-Jun 1991-1994</td>
<td>Oceanic waters</td>
<td>35</td>
<td>1.10</td>
</tr>
<tr>
<td>Apr-Jun 1996-2001 (excluding 1998)</td>
<td>Oceanic waters</td>
<td>40</td>
<td>0.61</td>
</tr>
<tr>
<td>Jun-Aug 2003, Apr-Jun 2004</td>
<td>Oceanic waters</td>
<td>15</td>
<td>1.98</td>
</tr>
<tr>
<td>Jun-Aug 2009</td>
<td>Oceanic waters</td>
<td>33</td>
<td>1.07</td>
</tr>
</tbody>
</table>

Minimum Population Estimate

The minimum population estimate (Nmin) is the lower limit of the two-tailed 60% confidence interval of the log-normally distributed best abundance estimate. This is equivalent to the 20th percentile of the log-normal distributed abundance estimate as specified by Wade and Angliss (1997). The best estimate of abundance for Bryde’s whales is 5133 (CV=0.50), and the minimum population estimate for the northern Gulf of Mexico Bryde’s whale is 34 (Table 2) Bryde’s whales.

Current Population Trend

The statistical power to detect a trend in abundance for this stock is poor due to the relatively imprecise abundance estimates and long intervals between surveys. For example, the power to detect a precipitous decline in abundance (i.e., 50% decrease in 15 years) with estimates of low precision (e.g., CV>0.30) remains below 80% (alpha = 0.30) unless surveys are conducted on an annual basis (Taylor et al. 2007). In addition, these surveys are restricted to U.S. waters, it is not possible to distinguish between changes in population size and Gulf-wide shifts in spatial distribution. Five point estimates of Bryde’s whale abundance have been made based on data from surveys during 2003 (June-August), 2004 (April-June), 2009 (July-August), 2017 (July-August), and 2018 (August-October). Each of these surveys had a similar design and was conducted using the same vessel or a vessel with a similar observation platform. Prior year surveys only employed a single survey team. The estimates for 2017 and 2018 were conducted in “passing” mode which resulted in increased numbers of unidentified sightings and may have affected group size estimation for distant groups of dolphins and small whales. Apportioning unidentified species among identified taxa addresses the first issue, and there were no apparent relationships between sighting distance and estimated group size. The model for detection probability on the trackline from the summer 2017 survey was applied to the abundance estimates from the 2003, 2004, and 2009 surveys. The resulting abundance estimates were: 2003, N=0 (CV=NA); 2004, N=64 (CV=0.88); 2009, N=100 (CV=1.03); 2017, N=84 (CV=0.92); and 2018, N=40 (CV=0.55). Pairwise comparisons of the non-zero log-transformed means were conducted between years, and significant differences were assessed at alpha=0.10. P-values were adjusted for multiple comparisons. There were no significant differences between survey years. A trend analysis has not been conducted for this stock. Four point estimates of Bryde’s whale abundance have been made based on data from line-transect surveys covering 1991-2009 (Table 1). The statistical power to detect a trend in abundance for this stock is poor due to the relatively imprecise abundance estimates and long intervals between surveys. For example, the power to detect a precipitous decline in abundance (i.e., 50% decrease in 15 years) with estimates of low precision (e.g., CV>0.30) remains below 80% (alpha = 0.30) unless surveys are conducted on an annual basis (Taylor et al. 2007). While not a trend analysis, it should be noted that research studies conducted under the Natural Resource Damage Assessment (NRDA) estimated there was up to a 22% decline in population size resulting from the Deepwater Horizon oil spill (see Habitat Issues section).

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

**CURRENT AND MAXIMUM NET PRODUCTIVITY RATES**

Current and maximum net productivity rates are unknown for this stock. For purposes of this assessment, the maximum net productivity rate was assumed to be 0.04. This value is based on theoretical modeling showing that cetacean populations likely do not grow at rates much greater than 4% given the constraints of their reproductive life history (Barlow et al. 1995). Between 1988 and 2018, there have been two documented strandings of calves (total length <700 cm) in the northern Gulf of Mexico (SEUS Historical Stranding Database unpublished data; NOAA National Marine Mammal Health and Stranding Response Database unpublished data).

**POTENTIAL BIOLOGICAL REMOVAL**

Potential Biological Removal (PBR) is the product of the minimum population size, one-half the maximum net productivity rate and a recovery factor (MMPA Sec. 3.16 U.S.C. 1362; Wade and Angliss 1997; Wade 1998). The minimum population size is 34. The maximum productivity rate is 0.04, the default value for cetaceans. The recovery factor is 0.1 because the stock is listed as endangered. PBR for the northern Gulf of Mexico Bryde’s whale is 0.1 (Table 2). Equivalent to one take every 33 years.

<table>
<thead>
<tr>
<th>Nbest</th>
<th>CV</th>
<th>Nmin</th>
<th>Fr</th>
<th>Rmax</th>
<th>PBR</th>
</tr>
</thead>
<tbody>
<tr>
<td>51</td>
<td>0.50</td>
<td>34</td>
<td>0.1</td>
<td>0.04</td>
<td>0.1</td>
</tr>
</tbody>
</table>

**ANNUAL HUMAN-CAUSED MORTALITY AND SERIOUS INJURY**

The total annual estimated human-caused fishery-related mortality and serious injury for the Gulf of Mexico Bryde’s whale stock during 2011–2015 is unknown. There was no documented fishery-caused mortality or serious injury for this stock during 2011–2015 (Table 3). Mean annual mortality and serious injury during 2011–2015 due to other human-caused actions (the Deepwater Horizon oil spill) was predicted to be 0.58. The minimum total mean annual human-caused mortality and serious injury for this stock during 2014–2015 was, therefore, 0.58. This is considered a minimum mortality estimate as some fisheries with which the stock could interact have limited observer coverage. In addition, the likelihood is low that a whale killed at sea due to a fishery interaction or vessel strike will be recovered (Williams et al. 2011).

<table>
<thead>
<tr>
<th>Years</th>
<th>Source</th>
<th>Annual Avg.</th>
<th>CV</th>
</tr>
</thead>
<tbody>
<tr>
<td>2014–2018</td>
<td>U.S. fisheries using observer data</td>
<td>Unknown</td>
<td>-</td>
</tr>
</tbody>
</table>

**Fisheries Information**

There are three commercial fisheries that overlap geographically and potentially could interact with this stock in the Gulf of Mexico. These include the Category I Atlantic Ocean, Caribbean, Gulf of Mexico large pelagics longline fishery, and two Category III fisheries, the Southeastern U.S. Atlantic, Gulf of Mexico shark bottom longline/hook-and-line fishery and the Southeastern U.S. Atlantic, Gulf of Mexico, and Caribbean snapper-grouper and other reef fish bottom longline/hook-and-line fishery. See Appendix III for detailed fishery information. All three of these fisheries have observer programs, however observer coverage is limited for the two Category III fisheries.

Pelagic swordfish, tunas, and billfish are the targets of the large pelagics longline fishery operating in the northern Gulf of Mexico. During 2014–2018 there were no observed mortalities or serious injuries to Bryde’s whales by this fishery (Garrison and Stokes 2016; 2017; 2019; 2020; in review). Percent observer coverage (percentage of sets observed) for this longline fishery for each year during 2014–2018 was 18, 19, 23, 13, and 20, respectively. There has been no reported fishing-related mortality or serious injury of a Bryde’s whale by this fishery during 1998–2015 (Yeung 1999; Yeung 2001; Garrison 2003; Garrison and Richards 2004; Garrison 2005; Fairfield Walsh and Garrison
2006, Fairfield-Walsh and Garrison 2007; Fairfield and Garrison 2008; Garrison et al. 2009; Garrison and Stokes 2010; 2012a, b; 2013; 2014; 2016; 2017). For the two category III bottom longline/hook-and-line fisheries, the target species are large and small coastal sharks and reef fishes such as snapper, grouper, and tilefish. There has been no reported fishery-related mortality or serious injury of a Bryde's whale by either of these fisheries (e.g., Scott-Denton et al. 2011; Hale et al. 2012; Gulak et al. 2013; 2014; Enzenauer et al. 2015; 2016; Mathers et al. 2017; 2018; 2020). Within the Gulf of Mexico, observer coverage for the snapper-grouper and other reef fish bottom longline fishery is ~1% or less annually, and for the shark bottom longline fishery coverage is 1–2% annually. Usually bottom longline gear is thought to pose less of a risk for cetaceans to become entangled than pelagic longline gear. However, if cetaceans forage along the seafloor, as is suspected for the Bryde’s whale (Soldevilla et al. 2017), then there is an opportunity for these whales to become entangled in the mainline as well as in the vertical buoy lines (Rosel et al. 2016).

Two other commercial fisheries that overlap to a small degree with the primary Bryde’s whale habitat in the northeastern Gulf of Mexico are the Category III Gulf of Mexico butterfish trawl fishery and Category II Southeastern U.S. Atlantic, Gulf of Mexico shrimp trawl fishery (Rosel et al. 2016). No interactions with Bryde’s whales have been documented for either of these fisheries. There is no observer coverage for the butterfish trawl fishery. The shrimp trawl fishery has ~2% observer coverage annually.

Other Mortality

There were two no reported strandings of Bryde’s whales in the Gulf of Mexico during 2014–2018 (NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 21 May 2019). During 2012, two Bryde’s whale strandings occurred in Louisiana. It could not be determined if there was evidence of human interaction for these strandings. Both whales were in a state of advanced decomposition when observed. Stranding data probably underestimate the extent of human and fishery-related mortality and serious injury, particularly for offshore stocks such as Bryde’s whales, because not all of the whales that die or are seriously injured in human interactions wash ashore, or, if they do, they are not all recovered (Peltier et al. 2012; Wells et al. 2015). In particular, oceanic stocks in the Gulf of Mexico are less likely to strand than nearshore coastal stocks or shelf stocks (Williams et al. 2011). Additionally, not all carcasses will show evidence of human interaction, entanglement or other fishery-related interaction due to decomposition, scavenger damage, etc. (Byrd et al. 2014). Finally, the level of technical expertise among stranding network personnel varies widely as does the ability to recognize signs of human interaction.

An Unusual Mortality Event (UME) was declared for cetaceans in the northern Gulf of Mexico beginning 1 March 2010 and ending 31 July 2014 (Litz et al. 2014; http://www.nmfs.noaa.gov/pr/health/mmume/cetacean_gulfofmexico.htm, accessed 1 June 2016). It includes cetaceans that stranded prior to the Deepwater Horizon (DWH) oil spill (see “Habitat Issues” below), during the spill, and after. Exposure to the DWH oil spill was determined to be the primary underlying cause of the elevated stranding numbers in the northern Gulf of Mexico after the spill (e.g., Schwacke et al. 2014; Venn-Watson et al. 2015; Colegrove et al. 2016; DWH NRDAT 2016; see Habitat Issues section). Two of Bryde’s whale strandings in 2012 were considered to be part of this UME.

A population model was developed to estimate the injury and time to recovery for stocks affected by the DWH oil spill, taking into account long-term effects resulting from mortality, reproductive failure, reduced survival rates, and the proportion of the stock exposed to DWH oil (DWH MMIQT 2015). Based on the population model, it was projected that 2.3–3.8 Bryde’s whales died during 2014–2018 (see Appendix VI) due to elevated mortality associated with oil exposure and that the stock experienced a 22% maximum reduction in population size due to the oil spill (DWH MMIQT 2015). The DWH Marine Mammal Injury Quantification Team cautioned that the capability of Bryde’s whales to recover from the DWH oil spill is unknown because the population models do not account for stochastic processes and genetic effects (DWH MMIQT 2015), to which small populations are highly susceptible (Shaffer 1981; Rosel and Reeves 2000). The population model used to predict Bryde’s whale mortality due to the DWH event has a number of sources of uncertainty. Model parameters (e.g., survival rates, reproductive rates, and life-history parameters) were derived from literature sources for Bryde's whales occupying waters outside of the Gulf of Mexico. In addition, proxy values for the effects of DWH oil exposure on both survival rates and reproductive success were applied based upon estimated values for common bottlenose dolphins in Barataria Bay. Finally, there was no estimation of uncertainty in model parameters or outputs.
Ship strikes may pose a threat to this stock. In 2009, a Bryde’s whale was found floating in the Port of Tampa, Tampa Bay, Florida. The whale had evidence of pre-mortem and post-mortem blunt trauma, and was determined to have been struck by a ship, draped across the bow, and carried into port.

**HABITAT ISSUES**

The Deepwater Horizon DWH MC252 drilling platform, located approximately 80 km southeast of the Mississippi River Delta in waters about 1,500 m deep, exploded on 20 April 2010. The rig sank, and over 87 days, up to ~3.2 million barrels of oil were discharged from the wellhead until it was capped on 15 July 2010 (DWH NRDAT 2016). Shortly after the oil spill, the NRDA process was initiated under the Oil Pollution Act of 1990. A variety of NRDA research studies were conducted to determine potential impacts of the spill on marine mammals. These studies estimated that 48% of the Bryde’s whales in the Gulf stock were exposed to oil, that 22% (95% CI: 10–31) of females suffered from reproductive failure, and 18% (95% CI: 7–28) of the population suffered adverse health effects (DWH MMIQT 2015). A population model estimated the stock experienced a maximum 22% reduction in population size (see Other Mortality section above).

Vessel strikes also pose a threat to this stock (Soldevilla et al. 2017). In 2009, a Bryde’s whale was found floating in the Port of Tampa, Tampa Bay, Florida. The whale had evidence of pre-mortem and post-mortem blunt trauma, and was determined to have been struck by a vessel, ship, draped across the bow, and carried into port.

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

**STATUS OF STOCK**

The Bryde's whale is not listed as threatened or endangered under the Endangered Species Act, but therefore the northern Gulf of Mexico stock is considered strategic under the MMPA because minimum total mean annual human-caused mortality and serious injury exceeds PBR. In addition, the stock is very small and exhibits very low genetic diversity, which places the stock at great risk of demographic stochasticity. The stock’s restricted range also places it at risk of environmental stochasticity. The DWH oil spill is estimated to have resulted in a 22% maximum decline in population size for this stock, and in addition, the mean modeled annual human-caused mortality and serious injury due to the DWH oil spill exceeds PBR for this stock. In April 2015, NMFS made a positive 90-day finding on a petition to list the Gulf of Mexico Bryde's whale as endangered under the ESA (NMFS 2015) and a proposed rule to list was published in December 2016 (NMFS 2016). The status of this stock relative to OSP is unknown. There was no statistically significant trend in population size for this stock in the northern Gulf of Mexico. There are insufficient data to determine population trends for this stock.

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CUVIER'S BEAKED WHALE (*Ziphius cavirostris*):
Northern Gulf of Mexico Stock

STOCK DEFINITION AND GEOGRAPHIC RANGE

Cuvier's beaked whales are distributed throughout the world's oceans except for the polar regions (Leatherwood and Reeves 1983; Heyning 1989). Strandings have occurred in all months along the east coast of the U.S. (Schmidly 1981) and throughout the year in the Gulf of Mexico (Würsig et al. 2000). In the northern Gulf of Mexico, Cuvier’s beaked whales are seen primarily in waters ≥1,000 m (Figure 1) and have been seen in all seasons during GulfCat aerial and vessel surveys of the northern Gulf of Mexico (i.e., U.S. Gulf of Mexico) (Hansen et al. 1996; Mullin and Hoggard 2000; Maze-Foley and Mullin 2006; Garrison and Aichinger Dias 2020). Some of the aerial survey sightings may have included Cuvier’s beaked whales, but identification of beaked whale species from aerial surveys is problematic. Beaked whale sightings made during spring and summer vessel surveys have been widely distributed in waters >500 m deep (Maze-Foley and Mullin 2006; Figure 1). Recent beaked whale sighting locations from summer 2017 and summer/fall 2018 oceanic surveys are shown in Figure 1.

![Figure 1. Distribution of beaked whale on-effort sightings from SEFSC vessel surveys during spring 1996-2001, summer 2003, spring 2004, summer 2009, summer 2017, and summer/fall 2018. Isobaths are the 200-m, 1,000-m, and 2,000-m depth contours. The darker line indicates the U.S. EEZ.](image)

All the cetacean species found in the oceanic northern Gulf of Mexico almost certainly occur in similar habitat beyond U.S. boundaries in the southern Gulf. There are fewer cetacean sighting and stranding records in the southern Gulf due to more limited effort. Nevertheless there are records for most oceanic species in the southern Gulf (e.g., Jefferson and Schiro 1997; Ortega Ortiz 2002; Ortega-Argueta et al. 2005; Jefferson et al. 2008; Vázquez Castán et al. 2009; Whitt et al. 2011). Although there are only a few records from Gulf of Mexico waters beyond U.S. boundaries (e.g., Jefferson and Schiro 1997, Ortega Ortiz 2002), Cuvier’s beaked whales almost certainly occur throughout the oceanic Gulf of Mexico (Jefferson et al. 2008), which is also composed of waters belonging to Mexico and Cuba where there is currently little information on cetacean species abundance and distribution. U.S. waters only comprise about 40% of the entire Gulf of Mexico, and 65% of oceanic waters are south of the U.S. Exclusive Economic Zone (EEZ). This is therefore likely a transboundary stock with Cuba and/or Mexico. Because U.S. waters only comprise about 40% of the entire Gulf of Mexico and 35% of the oceanic (i.e., >200 m) Gulf of Mexico (Mullin and Fulling 2004), abundance and stock boundaries of oceanic species are poorly known.
Cuvier’s beaked whales in the northern Gulf of Mexico are managed separately from those in the western North Atlantic. Although there have been no directed studies of the degree of demographic independence between the two areas, this management structure is consistent with the fact that the Gulf of Mexico and western North Atlantic belong to distinct marine ecoregions (Spalding et al. 2007; Moore and Merrick 2011). There are insufficient data to determine whether the northern Gulf of Mexico stock comprises multiple demographically independent populations. Additional morphological, acoustic, genetic, and/or behavioral data are needed to further delineate population structure within the Gulf of Mexico and across the broader geographic area.

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

Strandings of Cuvier’s beaked whales along the west coast of North America, based on skull characteristics, are thought to represent members of a panmictic population (Mitchell 1968), but there is no information on stock differentiation in the Gulf of Mexico and nearby waters. In the absence of adequate information on stock structure, a species’ range within an ocean should be divided into defensible management units, and such management units include distinct oceanographic regions (Wade and Apgar 1997). The Gulf of Mexico population is provisionally being considered a separate stock for management purposes, although there is currently no information to differentiate this stock from the Atlantic Ocean stock(s). Additional morphological, genetic and/or behavioral data are needed to provide further information on stock delineation.

POPULATION SIZE

The best abundance estimate \( \left( N_{\text{best}} \right) \) for Cuvier’s beaked whales in the northern Gulf of Mexico is \( 1874 \) (\( CV=0.75 \), Table 1). This estimate is from a summer 2009 oceanic survey covering waters from the 200-m isobath to the seaward extent of the U.S. EEZ. This estimate is from summer 2007 and summer/fall 2018 oceanic surveys covering waters from the 200-m isobath to the seaward extent of the U.S. EEZ (Garrison et al. 2020). This estimate was not corrected for the probability of detection on the trackline, and is likely a severe underestimate due to the long dive times of this species. The estimate for the same time period for unidentified Ziphiidae was 181 (\( CV=0.31 \)), which may have also included an unknown number of Cuvier’s beaked whales. However, this abundance estimate is negatively biased because only sightings of beaked whales which could be positively identified to species were used. The estimate for the same time period for unidentified Ziphiidae is 74 (\( CV=1.04 \)), which may also include an unknown number of Cuvier’s beaked whales.

Earlier abundance estimates

Please see Appendix IV for a summary of abundance estimates, including earlier estimates and survey descriptions. All estimates of abundance were derived through the application of distance sampling analysis (Buckland et al. 2001) and the computer program DISTANCE (Thomas et al. 1998) to line transect survey data collected from ships in the oceanic northern Gulf of Mexico (i.e., 200-m isobath to seaward extent of the U.S. EEZ), and are summarized in Appendix IV.

From 1991 through 1994, and from 1996 to 2001 (excluding 1998), annual surveys were conducted during spring along a fixed plankton-sampling trackline. Due to limited survey effort in any given year, the survey effort-weighted estimated average abundance of Cuvier’s beaked whales for all surveys combined was estimated. For 1991 to 1994, the estimate was 30 (\( CV=0.50 \)) (Hansen et al. 1995), and for 1996 to 2001, 95 (\( CV=0.47 \)) (Mullin and Fulling 2004; Table 1). The estimated abundance of Cuvier’s beaked whales was negatively biased because only sightings of beaked whales which could be positively identified to species were used. The estimate for the same time period for unidentified Ziphiidae was 146 (\( CV=0.46 \)), which may have also included an unknown number of Cuvier’s beaked whales.
During summer 2003 and spring 2004, surveys dedicated to estimating cetacean abundance were conducted along a grid of uniformly spaced transect lines from a random start. The abundance estimate for Cuvier’s beaked whales, pooled from 2003 to 2004, was 65 (CV=0.67) (Mullin 2007; Table 1). The estimate for the same time period for unidentified Ziphiidae was 337 (CV=0.40), which may have also included an unknown number of Cuvier’s beaked whales.

Recent surveys and abundance estimates

During summer 2009, a line-transect survey dedicated to estimating the abundance of oceanic cetaceans was conducted in the northern Gulf of Mexico. Survey lines were stratified in relation to depth and the location of the Loop Current. An abundance estimate for Cuvier’s beaked whales was generated from vessel surveys conducted in the northern Gulf of Mexico from the continental shelf edge (~200-m isobath) to the seaward extent of the U.S. EEZ (Table 1; Garrison et al. 2020). One survey was conducted from 2 July to 25 August 2017 and consisted of 7,302 km of on-effort trackline, and the second survey was conducted from 11 August to 6 October 2018 and consisted of 6,473 km of on-effort trackline within the surveyed strata. Both surveys used a double-platform data-collection procedure to allow estimation of the detection probability on the trackline using the independent observer approach assuming point independence (Laake and Borchers 2004). However, there were too few sightings and too few resightings of this species to allow estimation of detection probability on the trackline. Therefore, abundance estimates were derived using MCDSS distance sampling methods that accounted for the effects of covariates (e.g., sea state, glare) on detection probability within the surveyed strip (Thomas et al. 2010) implemented in package mrds (version 2.21, Laake et al. 2020) in the R statistical programming language. The surveys were conducted in "passing mode" (e.g., Schwarz et al. 2010) while all prior surveys in the Gulf of Mexico have been conducted in "closing mode." The inverse variance weighted mean abundance estimate for Cuvier’s beaked whales in oceanic waters during 2017 and 2018 was 18 (CV=0.75; Table 1; Garrison et al. 2020). This estimate was not corrected for the probability of detection on the trackline, and is likely a severe underestimate due to the long dive times of this species. The abundance estimate for Cuvier’s beaked whales in oceanic waters during 2009 was 74 (CV=1.04; Table 1). The estimate for the same time period for unidentified Ziphiidae was 181 also 74 (CV=0.311.04), which may have also included an unknown number of Cuvier’s beaked whales.

Table 1. Most recent abundance estimate (Nbest) and coefficient of variation (CV) of Cuvier’s beaked whales in the northern Gulf of Mexico oceanic waters (200 m to the offshore extent of the EEZ) based on an average from summer 2017 and summer/fall 2018 vessel surveys.

<table>
<thead>
<tr>
<th>Years</th>
<th>Area</th>
<th>Nbest</th>
<th>CV</th>
</tr>
</thead>
<tbody>
<tr>
<td>2017, 2018</td>
<td>Gulf of Mexico</td>
<td>18</td>
<td>0.75</td>
</tr>
</tbody>
</table>

Table 1. Summary of abundance estimates for northern Gulf of Mexico Cuvier’s beaked whales. Month, year and area covered during each abundance survey, and resulting abundance estimate (Nbest) and coefficient of variation (CV).
Minimum Population Estimate

The minimum population estimate (Nmin) is the lower limit of the two-tailed 60% confidence interval of the log-normal distributed abundance estimate. This is equivalent to the 20th percentile of the log-normal distributed abundance estimate as specified by Wade and Angliss (1997). The best estimate of abundance for Cuvier’s beaked whales is 1874 (CV=0.75, Rmax=1.04). The minimum population estimate for the northern Gulf of Mexico is 36 (Cuvier’s beaked whale is 10 (Table 2).

Current Population Trend

The statistical power to detect a trend in abundance for this stock is poor due to the relatively imprecise abundance estimates and long intervals between surveys. For example, the power to detect a precipitous decline in abundance (i.e., 50% decrease in 15 years) with estimates of low precision (e.g., CV>0.30) remains below 80% (alpha=0.30) unless surveys are conducted on an annual basis (Taylor et al. 2007). In addition, because these surveys are restricted to U.S. waters, it is not possible to distinguish between changes in population size and Gulf-wide shifts in spatial distribution. Five point estimates of Ziphiidae abundance have been made based on data from surveys during: 2003 (June–August), 2004 (April–June), 2009 (July–August), 2017 (July–August), and 2018 (August–October). Each of these surveys had a similar design and was conducted using the same vessel or a vessel with a similar observation platform. Prior year surveys only employed a single survey team. The estimates for 2017 and 2018 were conducted in “passing” mode which resulted in increased numbers of unidentified sightings and may have affected group size estimation for distant groups of dolphins and small whales. Apportioning unidentified species among identified taxa addresses the first issue, and there were no apparent relationships between sighting distance and estimated group size. The model for detection probability on the trackline from the summer 2017 survey was applied to the abundance estimates from the 2003, 2004, and 2009 surveys. The resulting abundance estimates were: 2003, N=573 (CV=0.44); 2004, N=55 (CV=0.72); 2009, N=276 (CV=0.59); 2017, N=303 (CV=0.49); and 2018, N=322 (CV=0.34). Pairwise comparisons of the log-transformed means were conducted between years, and significant differences were assessed at alpha=0.10. P-values were adjusted for multiple comparisons. There was a significant decrease between the 2003 and 2004 estimates (p.adjusted=0.012) and a significant increase between the 2004 and 2018 estimates (p.adjusted =0.067), however there is no clear pattern in the overall trend. Four point estimates of Cuvier’s beaked whale abundance have been made based on data from surveys covering 1991-2009. The estimates vary by a maximum factor of more than three. To determine whether changes in abundance have occurred over this period, an analysis of all the survey data needs to be conducted which incorporates covariates (e.g., survey conditions, season) that could potentially affect estimates. Nevertheless, differences in temporal abundance estimates will still be difficult to interpret without a Gulf of Mexico-wide understanding of Cuvier’s beaked whale abundance. The oceanography of the Gulf of Mexico is quite dynamic, and the spatial scale of the Gulf is small relative to the ability of most cetacean species to travel. Studies based on abundance and distribution surveys restricted to U.S. waters are unable to detect temporal shifts in distribution beyond U.S. waters that might account for any changes in abundance.

CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

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

POTENTIAL BIOLOGICAL REMOVAL

Potential Biological Removal (PBR) is the product of the minimum population size, one half the maximum net productivity rate and a recovery factor (MMPA Sec. 3.16 U.S.C. 1362; Wade and Angliss 1997). The minimum population size for the Cuvier’s beaked whale is 1036. The maximum productivity rate is 0.04, the default value for cetaceans. The recovery factor for this stock is 0.5 because the stock is of unknown status. PBR for the northern Gulf of Mexico Cuvier’s beaked whale is 0.104 (Table 2).

<table>
<thead>
<tr>
<th>Nbest</th>
<th>CV</th>
<th>Nmin</th>
<th>Fr</th>
<th>Rmax</th>
<th>PBR</th>
</tr>
</thead>
<tbody>
<tr>
<td>1874</td>
<td>0.75</td>
<td>36</td>
<td>0.5</td>
<td>0.04</td>
<td>0.104</td>
</tr>
</tbody>
</table>

Table 2. Best and minimum abundance estimates for Gulf of Mexico Cuvier’s beaked whales with Maximum Productivity Rate (Rmax), Recovery Factor (Fr) and PBR.
ANNUAL HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

Total annual estimated fishery-related mortality and serious injury to this stock during 2014–2018 was presumed to be zero, as there were no reports of mortalities or serious injuries to Cuvier’s beaked whales or unidentified beaked whales in the Gulf of Mexico (Table 3). Mean annual mortality and serious injury during 2014–2018 for all beaked whales due to other human-caused actions (the Deepwater Horizon oil spill) was predicted to be 5.2. The minimum total mean annual human-caused mortality and serious injury for beaked whales during 2014–2018 was, therefore, 5.2. This is a combined estimate for Blainville’s, Gervais’, and Cuvier’s beaked whales. The minimum total mean annual human-caused mortality and serious injury for Cuvier’s beaked whale is unknown. There has been no reported fishing-related mortality of a Cuvier’s beaked whale during 1998–2010 (Yeung 1999; Yeung 2001; Garrison 2003; Garrison and Richards 2004; Garrison 2005; Fairfield Walsh and Garrison 2006; Fairfield Walsh and Garrison 2007; Fairfield and Garrison 2008; Garrison et al. 2009; Garrison and Stokes 2010; 2011). However, during 2007 there was 1 unidentified beaked whale released alive with no serious injury after an entanglement interaction with the pelagic longline fishery (Fairfield and Garrison 2008).

Table 3. Total annual estimated fishery-related mortality and serious injury for northern Gulf of Mexico Cuvier’s beaked whales.

<table>
<thead>
<tr>
<th>Years</th>
<th>Source</th>
<th>Annual Avg.</th>
<th>CV</th>
</tr>
</thead>
<tbody>
<tr>
<td>2014-2018</td>
<td>U.S. fisheries using observer data</td>
<td>0</td>
<td></td>
</tr>
</tbody>
</table>

Fisheries Information

There are two commercial fisheries that interact, or that could potentially interact, with this stock in the Gulf of Mexico. These are the Category I Atlantic Highly Migratory Species (high seas) longline fishery and the Category I Atlantic Ocean, Caribbean, Gulf of Mexico large pelagics longline fishery (Appendix III). Percent observer coverage (percentage of sets observed) for these longline fisheries for each year during 2014–2018 was 18, 19, 23, 13 and 20, respectively. There is very little effort within the Gulf of Mexico by the Atlantic Highly Migratory Species (high seas) longline fishery, and no takes of beaked whales within high seas waters of the Gulf of Mexico have been observed or reported thus far. Pelagic swordfish, tunas and billfish are the targets of the pelagic longline fishery operating in the northern Gulf of Mexico. There were no reports of mortality or serious injury to Cuvier’s or other beaked whales by this fishery during 2014–2018 (Garrison and Stokes 2016; 2017; 2019; 2020; in review).

Other mortality

Cuvier’s beaked whales were taken occasionally in a small, directed fishery for cetaceans that operated out of the Lesser Antilles (Caldwell and Caldwell 1971). There were nowas one reported strandings of a Cuvier’s beaked whale in the Gulf of Mexico during 2006-2014-2018 (NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 21 May 2019; 16 November 2011). The whale stranded in 2014 in Florida, and it could not be determined if there was evidence of human interaction. Stranding data underestimate the extent of human and fishery-related mortality and serious injury because not all of the marine mammals that die or are seriously injured in human interactions wash ashore, or, if they do, they are not all recovered (Peltier et al. 2012; Wells et al. 2015). In particular, oceanic stocks in the Gulf of Mexico are less likely to strand than nearshore coastal stocks or shelf stocks (Williams et al. 2011). Additionally, not all carcasses will show evidence of human interaction, entanglement or other fishery-related interaction due to decomposition, scavenger damage, etc. (Byrd et al. 2014). Finally, the level of
An Unusual Mortality Event (UME) was declared for cetaceans in the northern Gulf of Mexico beginning 1 February, March 2010 and ending 31 July 2014 (Litz et al. 2014; https://www.fisheries.noaa.gov/national/marine-life-distress/2010-2014-cetacean-unusual-mortality-event-northern-gulf-mexico) and, as of early 2012, the event is still ongoing. It included cetaceans that stranded prior to the Deepwater Horizon (DWH) oil spill (see “Habitat Issues” below), during the spill, and after. Exposure to the DWH oil spill was determined to be the primary underlying cause of the elevated stranding numbers in the northern Gulf of Mexico after the spill (e.g., Schwacke et al. 2014; Venn-Watson et al. 2015; Colegrove et al. 2016; DWH NRDAT 2016; see Habitat Issues section). However, there were no Cuvier’s beaked whale strandings recovered within the spatial and temporal boundaries of this UME. During 2010, no animals from this stock were considered to be part of the UME.

A population model was developed to estimate the injury and time to recovery for stocks affected by the DWH oil spill, taking into account long-term effects resulting from mortality, reproductive failure, reduced survival rates, and the proportion of the stock exposed to DWH oil (DWH MMIQT 2015). Overall, this model estimated that the stocks experienced a 6% maximum reduction in population size due to the oil spill (DWH MMIQT 2015). The mortality projected for the years 2010–2013 due to the spill has not been reported previously. Based on the population model, it was projected that 51 beaked whales died during 2010–2013 (four year annual average of 13) due to elevated mortality associated with oil exposure (see Appendix VI). For the 2014–2018 reporting period of this SAR, the population model estimated 26 beaked whales died due to elevated mortality associated with oil exposure. However, this mortality estimate is not comparable to the current abundance estimate derived from visual surveys because the population model included a correction factor for detection probability derived from acoustic density estimates (DWH MMIQT 2015). The population model used to predict beaked whale mortality due to the DWH event has a number of sources of uncertainty. Model parameters (e.g., survival rates, reproductive rates, and life-history parameters) were derived from literature sources for beaked whales occupying waters outside of the Gulf of Mexico. In addition, proxy values for the effects of DWH oil exposure on both survival rates and reproductive success were applied based upon estimated values for common bottlenose dolphins in Barataria Bay. Finally, there was no estimation of uncertainty in model parameters or outputs.

Several unusual mass strandings of beaked whales in North Atlantic marine environments have been associated with military naval activities. 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 whales 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 active sonar tests conducted by the North Atlantic Treaty Organization (Frantzis 1998). In March 2000, 14 beaked whales live stranded in the Bahamas; 6 beaked whales (5 Cuvier’s and 1 Blainville’s) died (Evans and England 2001; Balcomb and Claridge 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. Necropsies were performed on 5 of the dead beaked whales and 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) (Evans and England 2001; Cox et al. 2006).

HABITAT ISSUES

The Deepwater Horizon (DWH) MC252 drilling platform, located approximately 50 miles 80 km southeast of the Mississippi River Delta in waters about 1,500 m deep, exploded on 20 April 2010. The rig sank, and forever 87 days ~3.2 million barrels of oil and gas were discharged from the wellhead until it was capped on 15 July 2010 (DWH NRDAT 2016). During the response effort dispersants were applied extensively at the seafloor and at the sea surface (Lehr et al. 2010; OSAT 2010). In situ burning, or controlled burning of oil at the surface, was also used extensively as a response tool (Lehr et al. 2010). The oil, dispersant and burn residue compounds present ecological concerns. The magnitude of this oil spill was unprecedented in U.S. history, causing impacts to wildlife, natural habitats and human communities along coastal areas from western Louisiana to the Florida Panhandle (NOAA 2011). It could be
anthropogenic sound in the world’s oceans has been shown to affect marine mammals, with vessel traffic, seismic surveys, and active naval sonars being the main anthropogenic contributors to low- and mid-frequency noise in oceanic waters (e.g., Nowacek et al. 2015; Gomez et al. 2016; NMFS 2018). Anthropogenic noise, particularly from military sonar, shipping, and seismic testing, is an increasing habitat concern for beaked whales (Aguilar de Soto et al. 2006; Cox et al. 2006; McCarthy et al. 2011; Tyack et al. 2011; Joyce et al. 2020). Several mass strandings of beaked whales throughout their worldwide range have been associated with naval activities (D’Amico et al. 2009; Filadello et al. 2009). In March 2000, 14 beaked whales live stranded in the Bahamas. Six of the whales (5 Cuvier’s and 1 Blainville’s) died and necropsy revealed evidence of tissue trauma associated with an acoustic or impulse injury that caused the animals to strand (Balcomb and Claridge 2001; NMFS 2001; Cox et al. 2006). Fourteen beaked whales (mostly Cuvier’s and Blainville’s) stranded in the Canary Islands in 2002 (Cox et al. 2006; Fernandez et al. 2005; Martin et al. 2004). Gas bubble-associ ated lesions and fat embolism were found in necropsied animals from this event, leading researchers to link nitrogen supersaturation with sonar exposure (Fernandez et al. 2005). The long-term and population consequences of these impacts are less well-documented and likely vary by species and other factors. Impacts on marine mammal prey from sound are also possible (Carroll et al. 2017), but the duration and severity of any such prey effects on marine mammals are unknown. Finally, ingestion of marine debris, particularly plastics, is a concern; plastic is occasionally found in the stomach contents of stranded beaked whales. These studies have focused on identifying the type, magnitude, severity, length and impact of oil exposure to oceanic, coastal and estuarine marine mammals. The research is ongoing and likely will continue for some time. For continental shelf and oceanic cetaceans, the NOAA-led efforts include: aerial surveys to document the distribution, abundance, species and exposure of marine mammals and turtles relative to oil from DWH spill; and ship surveys to evaluate exposure to oil and other chemicals and to assess changes in animal behavior and distribution relative to oil exposure through visual and acoustic surveys, deployment of passive acoustic monitoring systems, collection of tissue samples, and deployment of satellite tags on sperm and Bryde’s whales.

Aerial surveys have observed Risso’s dolphins, spinner dolphins, pantropical spotted dolphins, striped dolphins, bottlenose dolphins and sperm whales swimming in oil in offshore waters (NOAA 2010a). The effects of oil exposure on marine mammals depend on a number of factors including the type and mixture of chemicals involved, the amount, frequency and duration of exposure, the route of exposure (inhaled, ingested, absorbed, or external) and biomedical risk factors of the particular animal (Geraci 1990; NOAA 2010b). In general, direct external contact with petroleum compounds or dispersants with skin may cause skin irritation, chemical burns and infections. Inhalation of volatile petroleum compounds or dispersants may irritate or injure the respiratory tract, which could lead to pneumonia or inflammation. Ingestion of petroleum compounds may cause injury to the gastrointestinal tract, which could affect an animal’s ability to digest or absorb food. Absorption of petroleum compounds or dispersants may damage kidney, liver and brain function in addition to causing immune suppression and anemia. Long-term chronic effects such as lowered reproductive success and decreased survival may occur (Geraci 1990; NOAA 2010b).

STATUS OF STOCK

Cuvier’s beaked whales are not listed as threatened or endangered under the Endangered Species Act, and the northern Gulf of Mexico stock is not considered strategic under the MMPA because PBR is likely a severe underestimate due to the long dive times of this species and because the mean modeled annual human-caused mortality and serious injury due to the DWH oil spill is based on all beaked whale species combined and cannot be apportioned to individual species. No fishery-related mortality or serious injury has been observed; therefore, total fishery-related mortality and serious injury can be considered insignificant and approaching the zero mortality and serious injury rate. The status of Cuvier’s beaked whales in the U.S. EEZ relative to OSP is unknown. There are insufficient data to determine the population trends for this stock. The status of Cuvier’s beaked whales and other beaked whales in the northern Gulf of Mexico, relative to OSP, is unknown. The species is not listed as threatened or endangered under the
Endangered Species Act. There are insufficient data to determine the population trends for this species. Total human-caused mortality and serious injury for this stock is not known but none has been documented. There is insufficient information available to determine whether the total fishery-related mortality and serious injury for this stock is insignificant and approaching zero mortality and serious injury rate. This is not a strategic stock because it is assumed that the average annual human-related mortality and serious injury does not exceed PBR.

Disturbance by anthropogenic noise may prove to be an important habitat issue in some areas of this population’s range, notably in areas of oil and gas activities or where shipping or naval activities are high. Limited studies are currently being conducted to address this issue and its impact, if any, on this and other marine species.

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BLAINVILLE’S BEAKED WHALE (*Mesoplodon densirostris*): Northern Gulf of Mexico Stock

**STOCK DEFINITION AND GEOGRAPHIC RANGE**

Three species of *Mesoplodon* are known to occur in the Gulf of Mexico, based on stranding or sighting data (Hansen et al. 1995; Würsig et al. 2000). These are Blainville's beaked whale (*M. densirostris*), Gervais’ beaked whale (*M. europaeus*) and Sowerby's beaked whale (*M. bidens*). Sowerby’s beaked whale in the Gulf of Mexico is considered extralimital because there is only one known stranding of this species (Bonde and O'Shea 1989) and because it normally occurs in northern temperate waters of the North Atlantic (Mead 1989). The possibility of another unknown species of *Mesoplodon* inhabiting the Gulf has been suggested based on passive acoustic recordings (Hildebrand et al. 2015).

Identification of *Mesoplodon* to species in the Gulf of Mexico is very difficult, and in many cases, *Mesoplodon* and Cuvier’s beaked whales (*Ziphius cavirostris*) cannot be distinguished; therefore, sightings of beaked whales (Family Ziphiidae) may be identified as *Mesoplodon* sp., Cuvier’s beaked whale, or unidentified Ziphiidae.

Blainville’s beaked whales appear to be widely but sparsely distributed in temperate and tropical waters of the world’s oceans (Leatherwood et al. 1976; Leatherwood and Reeves 1983). Strandings have occurred along the northwestern Atlantic coast from Florida to Nova Scotia (Schmidly 1981), and there have been 4 documented strandings and 2 sightings of this species in the northern Gulf of Mexico (i.e., U.S. Gulf of Mexico) (e.g., Hansen et al. 1995; Würsig et al. 2000). Because at-sea identification of *Mesoplodon* to species in the Gulf of Mexico is very difficult, sightings of beaked whales (Family Ziphiidae) made during visual surveys are often identified only as *Mesoplodon* sp. or unidentified ziphiidae, and are referred to more generically as ‘beaked whales.’ In the northern Gulf of Mexico, beaked whales are sighted most commonly in waters ≥1,000 m and they have been seen in all seasons during aerial and vessel surveys of the northern Gulf of Mexico (i.e., U.S. Gulf of Mexico) during GulfCat aerial surveys of the northern Gulf of Mexico from 1992 to 1998 (Hansen et al. 1996; Mullin and Hoggard 2000; Maze-Foley and Mullin 2006; Garrison and Aichinger Dias 2020; Figure 1). There are several confirmed sightings of Blainville’s beaked whales, two in the western Gulf and one off the Florida shelf (Garrison and Aichinger Dias 2020).

Beaked whale sightings made during spring and summer vessel surveys have been widely distributed in waters >500 m deep.
Recent beaked whale sighting locations from summer 2017 and summer/fall 2018 oceanic surveys are shown in Figure 1. All the cetacean species found in the oceanic northern Gulf of Mexico almost certainly occur in similar habitat beyond U.S. boundaries in the southern Gulf. There are fewer cetacean sighting and stranding records in the southern Gulf due to more limited effort. Nevertheless, there are records for most oceanic species in the southern Gulf (e.g., Ortega Ortiz 2002; Ortega-Argueta et al. 2005; Jefferson et al. 2008; Vázquez Castán et al. 2009; Whitt et al. 2011). This is therefore likely a transboundary stock with Cuba and/or Mexico. Because U.S. waters only comprise about 40% of the entire Gulf of Mexico and 35% of the oceanic (i.e., ≥200 m) Gulf of Mexico (Mullin and Fulling 2004), abundance and stock boundaries of oceanic species are poorly known. Although there are only a few records from Gulf of Mexico waters beyond U.S. boundaries (e.g., Ortega Ortiz 2002), Blainville’s beaked whales almost certainly occur throughout the oceanic Gulf of Mexico (Jefferson et al. 2008), which is also composed of waters belonging to Mexico and Cuba where there is currently little information on cetacean species abundance and distribution. U.S. waters only comprise about 40% of the entire Gulf of Mexico, and 65% of oceanic waters are south of the U.S. Exclusive Economic Zone (EEZ).

Blainville’s beaked whales in the northern Gulf of Mexico are managed separately from those in the western North Atlantic. Although there have been no directed studies of the degree of demographic independence between the two areas, this management structure is consistent with the fact that the Gulf of Mexico and western North Atlantic belong to distinct marine ecoregions (Spalding et al. 2007; Moore and Merrick 2011). There are insufficient data to determine whether the northern Gulf of Mexico stock comprises multiple demographically independent populations. Additional morphological, acoustic, genetic, and/or behavioral data are needed to further delineate population structure within the Gulf of Mexico and across the broader geographic area. The Gulf of Mexico population is provisionally being considered a separate stock for management purposes, although there is currently no information to differentiate this stock from the Atlantic Ocean stock(s). Additional morphological, acoustic, genetic, and/or behavioral data are needed to provide further information on stock delineation.

Figure 1. Distribution of beaked whale sightings from SEFSC vessel surveys from 2003 through 2018, during spring 1996-2001, summer 2003 and spring 2004, and summer 2009. All the on-effort sightings are shown, though not all were used to estimate abundance. Solid lines indicate the 100-m and 1,000-m isobaths and the offshore extent of the U.S. EEZ.

**POPULATION SIZE**

The total number of Blainville’s beaked whales in the northern Gulf of Mexico is unknown. The best available abundance estimate (Nbest) is for *Mesoplodon* spp., and is a combined estimate for Blainville’s beaked whale and Gervais’ beaked whale. The estimate of abundance for *Mesoplodon* spp. in oceanic waters, using data from a summer 2009 oceanic survey, is 149 (CV=0.91; Table 1). From summer 2017 and summer/fall 2018 oceanic surveys covering waters from the 200-m isobath to the seaward extent of the U.S. EEZ is 98 (CV=0.46; Table 1; Garrison et al. 2020). This estimate was not corrected for the probability of detection on the trackline, and is likely a severe underestimate due to the long dive times of this species. The estimate for the same time period for unidentified Ziphiidae was 181 (CV=0.31), which may have also included an unknown number of Blainville’s beaked whales.

**Earlier abundance estimates**

Please see Appendix IV for a summary of abundance estimates, including earlier estimates and survey descriptions. All estimates of abundance were derived through the application of distance sampling analysis (Buckland et al. 2001) and the computer program DISTANCE (Thomas et al. 1998) to line transect survey data collected from ships in the oceanic northern Gulf of Mexico (i.e., 200-m isobath to seaward extent of the U.S. EEZ) and are summarized in Appendix IV.

From 1991 through 1994, and from 1996 through 2001 (excluding 1998), annual surveys were conducted during spring along a fixed plankton-sampling trackline. Due to limited survey effort in any given year during 1991-1994, the survey effort-weighted estimated average abundance of undifferentiated beaked whales (*Mesoplodon* spp. and unidentified Ziphiidae) for all surveys combined was 117 (CV=0.38) (Hansen et al. 1995). Hansen et al. (1995) did not estimate the abundance of *Mesoplodon* spp. For 1996 to 2001, the survey effort weighted estimated average
abundance for *Mesoplodon* spp. was 106 (CV=0.41) (Mullin and Fulling 2004; Table 1). This was a combined estimate for Blainville’s and Gervais’ beaked whales. The estimate for the same time period for unidentified Ziphiidae was 146 (CV=0.46) which may have also included an unknown number of Blainville’s beaked whales.

During summer 2003 and spring 2004, surveys dedicated to estimating cetacean abundance were conducted along a grid of uniformly-spaced transect lines from a random start. The estimate of abundance for *Mesoplodon* spp., pooled from 2003 to 2004, was 57 (CV=1.40) (Mullin 2007; Table 1). This was a combined estimate for Blainville’s and Gervais’ beaked whales. The estimate for the same time period for unidentified Ziphiidae was 337 (CV=0.40), which may have also included an unknown number of Blainville’s beaked whales.

**Recent surveys and abundance estimates**

- During summer 2009, a line-transect survey dedicated to estimating the abundance of oceanic cetaceans was conducted in the northern Gulf of Mexico. Survey lines were stratified in relation to depth and the location of the Loop Current. The estimate of abundance for *Mesoplodon* spp. in oceanic waters during 2009 was 149 (CV=0.91; Table 1). An abundance estimate for *Mesoplodon* spp. was generated from vessel surveys conducted in the northern Gulf of Mexico from the continental shelf edge (~200 -m isobath) to the seaward extent of the U.S. EEZ (Table 1; Garrison et al. 2020). One survey was conducted from 2 July to 25 August 2017 and consisted of 7,302 km of on-effort trackline, and the second survey was conducted from 11 August to 6 October 2018 and consisted of 6,473 km of on-effort trackline within the surveyed strata. Both surveys used a double-platform data-collection procedure to allow estimation of the detection probability on the trackline using the independent observer approach assuming point independence (Laake and Borchers 2004). However, there were too few sightings and too few resightings of these species to allow estimation of detection probability on the trackline. Therefore, abundance estimates were derived using MCDS distance sampling methods that accounted for the effects of covariates (e.g., sea state, glare) on detection probability within the surveyed strip (Thomas et al. 2010) implemented in package mrcs (version 2.21, Laake et al. 2020) in the R statistical programming language. The surveys were conducted in “passing mode” (e.g., Schwarz et al. 2010) while all prior surveys in the Gulf of Mexico have been conducted in “closing mode.”. The inverse variance weighted mean abundance estimate for *Mesoplodon* spp. in oceanic waters during 2017 and 2018 was 98 (CV=0.46; Table 1; Garrison et al. 2020). This was a combined estimate for Blainville’s and Gervais’ beaked whales. This estimate was not corrected for the probability of detection on the trackline, and is likely a severe underestimate due to the long dive times of these species. The estimate for the same time period for unidentified Ziphiidae was 181 (CV=0.31), which may have also included an unknown number of Blainville’s beaked whales.

**Table 1. Most recent abundance estimate (N(best)) and coefficient of variation (CV) of *Mesoplodon* spp. in the northern Gulf of Mexico oceanic waters (200 m to the offshore extent of the EEZ) based on an average from summer 2017 and summer/fall 2018 vessel surveys.**

<table>
<thead>
<tr>
<th>Years</th>
<th>Area</th>
<th>N(best)</th>
<th>CV</th>
</tr>
</thead>
<tbody>
<tr>
<td>2017, 2018</td>
<td>U.S. Gulf of Mexico</td>
<td></td>
<td>98</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td>0.46</td>
</tr>
</tbody>
</table>

Table 1. Summary of abundance estimates for northern Gulf of Mexico *Mesoplodon* spp., which is a combined estimate for Blainville’s beaked whale and Gervais’ beaked whale. Month, year and area covered during each abundance survey, and resulting abundance estimate (N(best)) and coefficient of variation (CV).
Minimum Population Estimate

The minimum population estimate ($N_{min}$) is the lower limit of the two-tailed 60% confidence interval of the log-normal distributed abundance estimate. This is equivalent to the 20th percentile of the log-normal distributed abundance estimate as specified by Wade and Angliss (1997). The best estimate of abundance for *Mesoplodon* spp. *in the northern Gulf of Mexico* is 98,449 (CV=0.4691). The minimum population estimate for *Mesoplodon* spp. in the northern Gulf of Mexico is 68,777 (Table 2).

Current Population Trend

The statistical power to detect a trend in abundance for this stock is poor due to the relatively imprecise abundance estimates and long intervals between surveys. For example, the power to detect a precipitous decline in abundance (i.e., 50% decrease in 15 years) with estimates of low precision (e.g., CV>0.30) remains below 80% (alpha=0.30) unless surveys are conducted on an annual basis (Taylor et al. 2007). In addition, because these surveys are restricted to U.S. waters, it is not possible to distinguish between changes in population size and Gulf-wide shifts in spatial distribution. Five point estimates of Ziphiidae abundance have been made based on data from surveys during: 2003 (June–August), 2004 (April–June), 2009 (July–August), 2017 (July–August), and 2018 (August–October). Each of these surveys had a similar design and was conducted using the same vessel or a vessel with a similar observation platform. Prior year surveys only employed a single survey team. The estimates for 2017 and 2018 were conducted in "passing" mode which resulted in increased numbers of unidentified sightings and may have affected group size estimation for distant groups of dolphins and small whales. Apportioning unidentified species among identified taxa addresses the first issue, and there were no apparent relationships between sighting distance and estimated group size.

The model for detection probability on the trackline from the summer 2017 survey was applied to the abundance estimates from the 2003, 2004, and 2009 surveys. The resulting abundance estimates were: 2003, $N=573$ (CV=0.44); 2004, $N=55$ (CV=0.72); 2009, $N=276$ (CV=0.59); 2017, $N=303$ (CV=0.49); and 2018, $N=322$ (CV=0.34). Pairwise comparisons of the log-transformed means were conducted between years, and significant differences were assessed at alpha=0.10. P-values were adjusted for multiple comparisons. There was a significant decrease between the 2003 and 2004 estimates (p-adjusted=0.012) and a significant increase between the 2004 and 2018 estimates (p-adjusted =0.067), however there is no clear pattern in the overall trend. There are insufficient data to determine the population trends for this species due to uncertainty in species identification at sea. Three point estimates of *Mesoplodon* spp. abundance have been made based on data from surveys covering 1996–2009. The estimates vary by a maximum factor of nearly three. To determine whether changes in abundance have occurred over this period, an analysis of all the survey data needs to be conducted which incorporates covariates (e.g., survey conditions, season) that could potentially affect estimates. Nevertheless, differences in temporal abundance estimates will still be difficult to interpret without a Gulf of Mexico-wide understanding of *Mesoplodon* abundance. The oceanography of the Gulf of Mexico is quite dynamic, and the spatial scale of the Gulf is small relative to the ability of most cetacean species to travel. Studies based on abundance and distribution surveys restricted to U.S. waters are unable to detect temporal shifts in distribution beyond U.S. waters that might account for any changes in abundance.

CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

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

POTENTIAL BIOLOGICAL REMOVAL

Potential Biological Removal (PBR) is the product of the minimum population size, one half the maximum net productivity rate and a recovery factor (MMPA Sec. 3.16 U.S.C. 1362; Wade and Angliss 1997). The minimum population size for *Mesoplodon* spp. is 68,777. The maximum productivity rate is 0.04, the default value for cetaceans. The recovery factor, which accounts for endangered, depleted, threatened stocks, or stocks of unknown status relative to optimum sustainable population (OSP), is assumed to be 0.5 because the stock is of unknown status. PBR for the northern Gulf of Mexico *Mesoplodon* spp. is 0.708 (Table 2). It is not possible to determine the PBR for only
Blainville’s beaked whales.

Table 2. Best and minimum abundance estimates for Gulf of Mexico Mesoplodon spp. with Maximum Productivity Rate (Rmax), Recovery Factor (Fr) and PBR.

<table>
<thead>
<tr>
<th>Nbest</th>
<th>CV</th>
<th>Nmin</th>
<th>Fr</th>
<th>Rmax</th>
<th>PBR</th>
</tr>
</thead>
<tbody>
<tr>
<td>98</td>
<td>0.46</td>
<td>68</td>
<td>0.5</td>
<td>0.04</td>
<td>0.7</td>
</tr>
</tbody>
</table>

ANNUAL HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

Total annual estimated fishery-related mortality and serious injury to this stock during 2014–2018 was presumed to be zero, as there were no reports of mortalities or serious injuries to Blainville’s beaked whales or unidentified beaked whales from U.S. fisheries in the Gulf of Mexico (Table 3). Mean annual mortality and serious injury during 2014–2018 for all beaked whales due to other human-caused actions (the Deepwater Horizon oil spill) was predicted to be 5.2. The minimum total mean annual human-caused mortality and serious injury for beaked whales during 2014–2018 was, therefore, 5.2. This is a combined estimate for Blainville’s, Gervais’, and Cuvier’s beaked whales. The minimum total mean annual human-caused mortality and serious injury for Blainville’s beaked whale is unknown.

Table 3. Total annual estimated fishery-related mortality and serious injury for northern Gulf of Mexico Blainville’s beaked whales.

<table>
<thead>
<tr>
<th>Years</th>
<th>Source</th>
<th>Annual Avg.</th>
<th>CV</th>
</tr>
</thead>
<tbody>
<tr>
<td>2014-2018</td>
<td>U.S. fisheries using observer data</td>
<td>0</td>
<td>-</td>
</tr>
</tbody>
</table>

There has been no reported fishing-related mortality of a beaked whale during 1998–2010 (Yeung 1999; Yeung 2001; Garrison 2003; Garrison and Richards 2004; Garrison 2005; Fairfield Walsh and Garrison 2006; Fairfield Walsh and Garrison 2007; Fairfield and Garrison 2008; Garrison et al. 2009; Garrison and Stokes 2010; 2011). However, during 2007 there was 1 unidentified beaked whale released alive with no serious injury after an entanglement interaction with the pelagic longline fishery (Fairfield and Garrison 2008).

Fisheries Information

There are two commercial fisheries that interact, or that could potentially interact, with this stock in the Gulf of Mexico. These are the Category I Atlantic Highly Migratory Species (high seas) longline fishery and the Atlantic Ocean, Caribbean, Gulf of Mexico large pelagics longline fishery (Appendix III). Percent observer coverage (percentage of sets observed) for these longline fisheries for each year during 2014–2018 was 18, 19, 23, 13 and 20, respectively. The commercial fishery which potentially could interact with this stock in the Gulf of Mexico is the Atlantic Ocean, Caribbean, Gulf of Mexico large pelagic longline fishery (Appendix III). There is very little effort within the Gulf of Mexico by the Atlantic Highly Migratory Species (high seas) longline fishery, and no takes of beaked whales within high seas waters of the Gulf of Mexico have been observed or reported thus far. Pelagic swordfish, tunas and billfish are the targets of the large pelagics longline fishing operating in the northern Gulf of Mexico. There were no reports of mortality or serious injury to Blainville’s or other beaked whales by this fishery during 2014–2018 (Garrison and Stokes 2016; 2017; 2019; 2020; in review). 1998–2010 (Yeung 1999; Yeung 2001; Garrison 2003; Garrison and Richards 2004; Garrison 2005; Fairfield Walsh and Garrison 2006; Fairfield Walsh and Garrison 2007; Fairfield and Garrison 2008; Garrison et al. 2009; Garrison and Stokes 2010; 2011). However, during 2007, 1 unidentified beaked whale was observed entangled and released alive in the northern Gulf of Mexico. All gear was removed and the animal was presumed to have no serious injuries (Fairfield and Garrison 2008).

Other Mortality

There were no strandings of a Blainville’s beaked whale of Mesoplodon spp. or unidentified beaked whales during 2006–2018 (NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 16 November 2014; 21 May 2019). This stranding occurred in Florida in 2014, and it could not be determined if there was evidence of human interaction. Stranding data probably underestimate the extent of fishery-related mortality and serious injury because not all of the marine mammals which die or are seriously injured in fishery interactions wash ashore, not all that wash ashore are discovered, reported or investigated, 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 interactions. Stranding data underestimate the extent of human and fishery-related mortality and serious injury because not all of the marine mammals that die or are seriously injured in human interactions wash ashore, or, if they do, they are not all recovered (Peltier et al. 2012; Wells et al. 2015). In particular, oceanic stocks in the Gulf of Mexico are less likely to strand than nearshore coastal stocks or shelf stocks (Williams et al. 2011). Additionally, not all carcasses will show evidence of human interaction, entanglement or other fishery-related interaction due to decomposition, scavenger damage, etc. (Byrd et al. 2014). Finally, the level of technical expertise among stranding network personnel varies widely as does the ability to recognize signs of human interaction.

Since 1990, there have been 4213 bottlenose dolphin die-offs or Unusual Mortality Events (UMEs) in the northern Gulf of Mexico, and 1 of these included a few Blainville’s beaked whales. Between August 1999 and May 2000, 150 common bottlenose dolphins died coincident with K. brevis blooms and fish kills in the Florida Panhandle (additional strandings included three Atlantic spotted dolphins, Stenella frontalis, one Risso’s dolphin, Grampus griseus, two Blainville’s beaked whales, and four unidentified dolphins). Brevetoxin was determined to be the cause of this event (Twiner et al. 2012; Litz et al. 2014). Between August 1999 and May 2000, 152 bottlenose dolphins died coincident with K. brevis blooms and fish kills in the Florida Panhandle. Additional strandings included 3 Atlantic spotted dolphins, Stenella frontalis, 1 Risso’s dolphin, Grampus griseus, 2 Blainville’s beaked whales, and 4 unidentified dolphins. An UME was declared for cetaceans in the northern Gulf of Mexico beginning 1 February 2010; and ending 31 July 2014 and, as of early 2012, the event is still ongoing (Litz et al. 2014; https://www.fisheries.noaa.gov/national/marine-life-distress/2010-2014-cetacean-unusual-mortality-event-northern-gulf-mexico). It includes cetaceans that stranded prior to the Deepwater Horizon (DWH) oil spill (see “Habitat Issues” below), during the spill, and after. Exposure to the DWH oil spill was determined to be the primary underlying cause of the elevated stranding numbers in the northern Gulf of Mexico after the spill (e.g., Schwacke et al. 2014; Venn-Watson et al. 2015; Colegrove et al. 2016; DWH NRDAT 2016; see Habitat Issues section). During 2010, The 2014 stranding of a Blainville’s beaked whale was no animals from this stock were considered to be part of this UME.

A population model was developed to estimate the injury and time to recovery for stocks affected by the DWH oil spill, taking into account long-term effects resulting from mortality, reproductive failure, reduced survival rates, and the proportion of the stock exposed to DWH oil (DWH MMIQT 2015). Overall, this model estimated that the stocks experienced a 6% maximum reduction in population size due to the oil spill (DWH MMIQT 2015). The mortality projected for the years 2010–2013 due to the spill has not been reported previously. Based on the population model, it was projected that 51 beaked whales died during 2010–2013 (four year annual average of 13) due to elevated mortality associated with oil exposure (see Appendix VI). For the 2014–2018 reporting period of this SAR, the population model estimated 26 beaked whales died due to elevated mortality associated with oil exposure. However, this mortality estimate is not comparable to the current abundance estimate derived from visual surveys because the population model included a correction factor for detection probability derived from acoustic density estimates (DWH MMIQT 2015). The population model used to predict beaked whale mortality due to the DWH event has a number of sources of uncertainty. Model parameters (e.g., survival rates, reproductive rates, and life-history parameters) were derived from literature sources for beaked whales occupying waters outside of the Gulf of Mexico. In addition, proxy values for the effects of DWH oil exposure on both survival rates and reproductive success were applied based upon estimated values for common bottlenose dolphins in Barataria Bay. Finally, there was no estimation of uncertainty in model parameters or outputs.

- Several unusual mass strandings of beaked whales in North Atlantic marine environments have been associated with military naval activities. 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 whales 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 active sonar tests conducted by the North Atlantic Treaty Organization (Franzis 1998). In March 2000, 14 beaked whales live stranded in the Bahamas; 6 beaked whales (5 Cuvier’s and 1 Blainville’s) died (Evans and England 2001; Balcomb and Claridge 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. Necropsies were performed on 5 of the dead beaked whales and 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) (NMFS 2001; Cox et al. 2006).
petroleum compounds or dispersants may irritate or injure the respiratory tract, which could lead to pneumonia or
compounds or dispersants with skin may cause skin irritation, chemical burns and infections. Inhalation of volatile
risk factors of the particular animal (Geraci 1990; NOAA 2010b). In general, direct external contact with petroleum
frequency and duration of exposure, the route of exposure (inhaled, ingested, absorbed, or external) and biomedical
on marine mammals depend on a number of factors including the type and mixture of chemicals involved, the amount,
bottlenose dolphins and sperm whales swimming in oil in offshore waters (NOAA 2010a). The effects of oil exposure
Aerial surveys have observed Risso’s dolphins, spinner dolphins, pantropical spotted dolphins, striped dolphins,
habitat issues
The Deepwater Horizon (DWH) MC252 drilling platform, located approximately 50 miles 80 km southeast of the
Mississippi River Delta in waters about 1,500 m deep, exploded on 20 April 2010. The rig sank, and ever 87 days
~3.2 million barrels of oil and gas were discharged from the wellhead until it was capped on 15 July 2010 (DWH
NRDAT 2016). During the response effort dispersants were applied extensively at the seafloor and at the sea surface
(Lehr et al. 2010; OSAT 2010). In situ burning, or controlled burning of oil at the surface, was also used extensively
as a response tool (Lehr et al. 2010). The oil, dispersant and burn residue compounds present ecological concerns.
The magnitude of this oil spill was unprecedented in U.S. history, causing impacts to wildlife, natural habitats and
human communities along coastal areas from western Louisiana to the Florida Panhandle (NOAA 2011). It could be
years before the entire scope of damage is ascertained (NOAA 2011).

Shortly after the oil spill, the Natural Resource Damage Assessment (NRDA) process was initiated under the Oil
Pollution Act of 1990. A variety of NRDA research studies are being conducted to determine potential impacts of
the spill on marine mammals. These studies estimated that 12% (95% CI: 2–22) of beaked whales in the Gulf, which included Blainville’s, Cuvier’s and Gervais’ beaked whales, were exposed to oil, that 5% (95% CI: 3–8) of
females suffered from reproductive failure, and 4% (95% CI: 2–7) of the beaked whale populations suffered adverse
health effects (DWH MMIQT 2015). A population model estimated the stocks experienced a maximum 6% reduction in
population size (see Other Mortality section above).

Anthropogenic sound in the world’s oceans has been shown to affect marine mammals, with vessel traffic, seismic
surveys, and active naval sonars being the main anthropogenic contributors to low- and mid-frequency noise in oceanic
waters (e.g., Nowacek et al. 2015; Gomez et al. 2016; NMFS 2018). Anthropogenic noise, particularly from military
sonar, shipping, and seismic testing, is an increasing habitat concern for beaked whales (Aguilar de Soto et al. 2006; Cox et al. 2006; McCarthy et al. 2011; Tyack et al. 2011; Joyce et al. 2020). Seven Blainville’s beaked whales were
satellite tagged and tracked in the Bahamas prior to and during naval sonar exercises (Joyce et al. 2020). Following
exposure to mid-frequency active sonar, five of the whales were displaced 28–68 km from the source, and did not
return for two to four days after military exercises ceased. Data also suggested the whales spent less time in deep dives
during the early periods of sonar exposure (Joyce et al. 2020). In addition, several mass strandings of beaked whales
throughout their worldwide range have been associated with naval activities (D’Amico et al. 2009; Filadelfo et al.
2009). In March 2000, 14 beaked whales live stranded in the Bahamas. Six of the whales (5 Cuvier’s and 1
Blainville’s) died and necropsy revealed evidence of tissue trauma associated with an acoustic or impulse injury that
caused the animals to strand (Balcomb and Claridge 2001; NMFS 2001; Cox et al. 2006). Fourteen beaked whales
(mostly Cuvier’s beaked whales but also including Gervais’ and Blainville’s beaked whales) stranded in the Canary
embolism were found in necropsied animals from this event, leading researchers to link nitrogen supersaturation with
sonar exposure (Fernandez et al. 2005). The long-term and population consequences of these impacts are less well-
documented and likely vary by species and other factors. Impacts on marine mammal prey from sound are also possible
(Carroll et al. 2017), but the duration and severity of any such prey effects on marine mammals are unknown.

Finally, ingestion of marine debris, particularly plastics, is a concern; plastic is occasionally found in the stomach
contents of stranded beaked whales. These studies have focused on identifying the type, magnitude, severity, length and impact of oil exposure to oceanic, coastal and estuarine marine mammals. The research is ongoing and likely will continue for some time. For continental
shelf and oceanic cetaceans, the NOAA-led efforts include: aerial surveys to document the distribution, abundance,
species and exposure of marine mammals and turtles relative to oil from DWH spill; and ship surveys to evaluate
exposure to oil and other chemicals and to assess changes in animal behavior and distribution relative to oil exposure
through visual and acoustic surveys, deployment of passive acoustic monitoring systems, collection of tissue samples,
and deployment of satellite tags on sperm and Bryde’s whales.

Aerial surveys have observed Risso’s dolphins, spinner dolphins, pantropical spotted dolphins, striped dolphins,
bottlenose dolphins and sperm whales swimming in oil in offshore waters (NOAA 2010a). The effects of oil exposure
on marine mammals depend on a number of factors including the type and mixture of chemicals involved, the amount,
frequency and duration of exposure, the route of exposure (inhaled, ingested, absorbed, or external) and biomedical
risk factors of the particular animal (Geraci 1990; NOAA 2010b). In general, direct external contact with petroleum
compounds or dispersants with skin may cause skin irritation, chemical burns and infections. Inhalation of volatile
petroleum compounds or dispersants may irritate or injure the respiratory tract, which could lead to pneumonia or
inflammation. Ingestion of petroleum compounds may cause injury to the gastrointestinal tract, which could affect an animal’s ability to digest or absorb food. Absorption of petroleum compounds or dispersants may damage kidney, liver and brain function in addition to causing immune suppression and anemia. Long-term chronic effects such as lowered reproductive success and decreased survival may occur (Geraci 1990; NOAA 2010b).

STATUS OF STOCK

Blainville’s beaked whales are not listed as threatened or endangered under the Endangered Species Act, and the northern Gulf of Mexico stock is not considered strategic under the MMPA because PBR is likely a severe underestimate due to the long dive times of this species and because the mean modeled annual human-caused mortality and serious injury due to the DWH oil spill is based on all beaked whale species combined and cannot be apportioned to individual species. No fishery-related mortality or serious injury has been observed; therefore, total fishery-related mortality and serious injury can be considered insignificant and approaching the zero mortality and serious injury rate. The status of Blainville’s beaked whales in the U.S. EEZ relative to OSP is unknown. There are insufficient data to determine the population trends for this stock. The status of Blainville’s beaked whales or other beaked whales in the northern Gulf of Mexico, relative to OSP, is unknown. The species is not listed as threatened or endangered under the Endangered Species Act. There are insufficient data to determine the population trends for this species. Total human-caused mortality and serious injury for this stock is insignificant and approaching zero mortality and serious injury rate. This is not a strategic stock because it is assumed that the average annual human-related mortality and serious injury does not exceed PBR.

Disturbance by anthropogenic noise may prove to be an important habitat issue in some areas of this population’s range, notably in areas of oil and gas activities or where shipping or naval activities are high. Limited studies are currently being conducted to address this issue and its impact, if any, on this and other marine species.

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GERVAIS' BEAKED WHALE (Mesoplodon europaeus):
Northern Gulf of Mexico Stock

STOCK DEFINITION AND GEOGRAPHIC RANGE

Three species of Mesoplodon are known to occur in the Gulf of Mexico, based on stranding or sighting data (Hansen et al. 1995; Würsig et al. 2000). These are Gervais' beaked whale (M. europaeus), Blainville's beaked whale (M. densirostris), and Sowerby's beaked whale (M. bidens). Sowerby’s beaked whale in the Gulf of Mexico is considered extralimital because there is only one known stranding of this species (Bonde and O'Shea 1989) and because it normally occurs in northern temperate waters of the North Atlantic (Mead 1989). The possibility of another unknown species of Mesoplodon inhabiting the Gulf has been suggested based on passive acoustic recordings (Hildebrand et al. 2015). Identification of Mesoplodon to species in the Gulf of Mexico is very difficult, and in many cases, Mesoplodon and Cuvier's beaked whale (Ziphius cavirostris) cannot be distinguished; therefore, sightings of beaked whales (Family Ziphiidae) are identified as Mesoplodon sp., Cuvier's beaked whale, or unidentified Ziphiidae.

Gervais' beaked whales appear to be widely but sparsely distributed in temperate and tropical waters of the world's oceans (Leatherwood et al. 1976; Leatherwood and Reeves 1983). Strandings have occurred along the northwestern Atlantic coast from Florida to Nova Scotia (Schmidly 1981) and there have been 16 documented strandings in the northern Gulf of Mexico (Würsig et al. 2000). Because at-sea identification of Mesoplodon to species in the Gulf of Mexico is very difficult, sightings of beaked whales (Family Ziphiidae) made during visual surveys are often identified only as Mesoplodon sp. or unidentified Ziphiid, and are referred to more generically as "beaked whales." In the northern Gulf of Mexico, beaked whales are sighted most commonly in waters ≥500 m and they have been seen in all seasons during aerial and vessel surveys of the northern Gulf of Mexico (i.e., U.S. Gulf of Mexico) (Hansen et al. 1996; Mullin and Hoggard 2000; Maze-Foley and Mullin 2006; Garrison and Aichinger Dias 2020;
Figure 1. Beaked whales were seen in all seasons during GulfCet aerial surveys of the northern Gulf of Mexico (i.e., U.S. Gulf of Mexico) from 1992 to 1998 (Hansen et al. 1996; Mullin and Hoggard 2000). Beaked whale sightings made during spring and summer vessel surveys have been widely distributed in waters >500 m deep (Maze-Foley and Mullin 2006; Figure 1).

All the cetacean species found in the oceanic northern Gulf of Mexico almost certainly occur in similar habitat beyond U.S. boundaries in the southern Gulf. There are fewer cetacean sighting and stranding records in the southern Gulf due to more limited effort. Nevertheless, there are records for most oceanic species in the southern Gulf (e.g., Jefferson and Schiro 1997; Ortega Ortiz 2002; Ortega-Argueta et al. 2005; Jefferson et al. 2008; Vázquez Castán et al. 2009; Whitt et al. 2011). This is therefore likely a transboundary stock with Cuba and/or Mexico. Because U.S. waters only comprise about 40% of the entire Gulf of Mexico and 35% of the oceanic (i.e., >200 m) Gulf of Mexico (Mullin and Fulling 2004), abundance and stock boundaries of oceanic species are poorly known.

Gervais’ beaked whales in the northern Gulf of Mexico are managed separately from those in the western North Atlantic. Although there have been no directed studies of the degree of demographic independence between the two areas, this management structure is consistent with the fact that the Gulf of Mexico and western North Atlantic belong to distinct marine ecoregions (Spalding et al. 2007; Moore and Merrick 2011). There are insufficient data to determine whether the northern Gulf of Mexico stock comprises multiple demographically independent populations. Additional morphological, acoustic, genetic, and/or behavioral data are needed to further delineate population structure within the Gulf of Mexico and across the broader geographic area.

Although there are only a few records from Gulf of Mexico waters beyond U.S. boundaries (e.g., Jefferson and Schiro 1997, Ortega Ortiz 2002), Gervais’ beaked whales almost certainly occur throughout the oceanic Gulf of Mexico (Jefferson et al. 2008), which is also composed of waters belonging to Mexico and Cuba where there is currently little information on cetacean species abundance and distribution. U.S. waters only comprise about 40% of the entire Gulf of Mexico, and 65% of oceanic waters are south of the U.S. Exclusive Economic Zone (EEZ).

The Gulf of Mexico population is provisionally being considered a separate stock for management purposes, although there is currently no information to differentiate this stock from the Atlantic Ocean stock(s). Additional morphological, genetic and/or behavioral data are needed to provide further information on stock delineation.

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

POPULATION SIZE

The total number of Gervais’ beaked whales in the northern Gulf of Mexico is unknown. The best available abundance estimate is for *Mesoplodon* spp., and is a combined estimate for Gervais’ beaked whale and Blainville’s beaked whale. The estimate of abundance for *Mesoplodon* spp in oceanic waters, using data from a summer 2009 oceanic survey, is 149 (CV=0.91; Table 1). The best abundance estimate (Nbest) for Gervais’ beaked whales in the northern Gulf of Mexico is 20 (CV=0.98; Table 1). This estimate is from summer 2017 and summer/fall 2018 oceanic surveys covering waters from the 200-m isobath to the seaward extent of the U.S. EEZ (Garrison et al. 2020). This estimate was not corrected for the probability of detection on the trackline, and is likely a severe underestimate due to the long dive times of this species. The estimate for the same time period for *Mesoplodon* spp. (Blainville’s and Gervais’ beaked whales) was 98 (CV=0.46), and that for unidentified Ziphiidae was 181 (CV=0.31). The *Mesoplodon* spp. and unidentified Ziphiidae may have also included an unknown number of Gervais’ beaked whales.

Earlier abundance estimates

Please see Appendix IV for a summary of abundance estimates, including earlier estimates and survey descriptions. All estimates of abundance were derived through the application of distance sampling analysis (Buckland
et al. 2001) and the computer program DISTANCE (Thomas et al. 1998) to line-transect survey data collected from ships in the oceanic northern Gulf of Mexico (i.e., 200m isobath to seaward extent of the U.S. EEZ) and are summarized in Appendix IV.

From 1991 through 1994, and from 1996 through 2001 (excluding 1998), annual surveys were conducted during spring along a fixed plankton-sampling trackline. Due to limited survey effort in any given year during 1991-1994, the survey effort-weighted estimated average abundance of undifferentiated beaked whales (Mesoplodon spp. and unidentified Ziphiidae) for all surveys combined was 117 (CV=0.38) (Hansen et al. 1995). Hansen et al. (1995) did not estimate the abundance of Mesoplodon spp. For 1996 to 2001, the survey effort-weighted estimated average abundance for Mesoplodon spp. was 106 (CV=0.41) (Mullin and Fulling 2004; Table 1). This was a combined estimate for Blainville’s and Gervais’ beaked whales. The estimate for the same time period for unidentified Ziphiidae was 146 (CV=0.46) which may have also included an unknown number of Gervais’ beaked whales.

During summer 2003 and spring 2004, surveys dedicated to estimating cetacean abundance were conducted along a grid of uniformly-spaced transect lines from a random start. The estimate of abundance for Mesoplodon spp., pooled from 2003 to 2004, was 57 (CV=1.40) (Mullin 2007; Table 1). This was a combined estimate for Blainville’s and Gervais’ beaked whales. The estimate for the same time period for unidentified Ziphiidae was 337 (CV=0.40), which may have also included an unknown number of Gervais’ beaked whales.

Recent surveys and abundance estimates

An abundance estimate for Gervais’ beaked whales was generated from vessel surveys conducted in the northern Gulf of Mexico from the continental shelf edge (~200-m isobath) to the seaward extent of the U.S. EEZ (Table 1; Garrison et al. 2020). One survey was conducted from 2 July to 25 August 2017 and consisted of 7,302 km of on-effort trackline, and the second survey was conducted from 11 August to 6 October 2018 and consisted of 6,473 km of on-effort trackline within the surveyed strata. Both surveys used a double-platform data-collection procedure to allow estimation of the detection probability on the trackline using the independent observer approach assuming point independence (Laake and Borchers 2004). However, there were too few sightings and too few resightings of this species to allow estimation of detection probability on the trackline. Therefore, abundance estimates were derived using MCDS distance sampling methods that accounted for the effects of covariates (e.g., sea state, glare) on detection probability within the surveyed strip (Thomas et al. 2010) implemented in package mrdss (version 2.21, Laake et al. 2020) in the R statistical programming language. The surveys were conducted in "passing mode" (e.g., Schwarz et al. 2010) while all prior surveys in the Gulf of Mexico have been conducted in "closing mode." The inverse variance weighted mean abundance estimate for Gervais’ beaked whales in oceanic waters during 2017 and 2018 was 20 (CV=0.98; Table 1; Garrison et al. 2020). This estimate was not corrected for the probability of detection on the trackline, and is likely a severe underestimate due to the long dive times of this species. The estimate for the same time period for Mesoplodon spp. (Blainville’s and Gervais’ beaked whales) was 98 (CV=0.46), and that for unidentified Ziphiidae was 181 (CV=0.31). The Mesoplodon spp. and unidentified Ziphiidae may have also included an unknown number of Gervais’ beaked whales. During summer 2009, a line-transect survey dedicated to estimating the abundance of oceanic cetaceans was conducted in the northern Gulf of Mexico. Survey lines were stratified in relation to depth and the location of the Loop Current. The estimate of abundance for Mesoplodon spp. in oceanic waters during 2009 was 149 (CV=0.91; Table 1). This was a combined estimate for Blainville’s and Gervais’ beaked whales. The estimate for the same time period for unidentified Ziphiidae was 20574 (CV=0.341.04), which may have also included an unknown number of Gervais’ beaked whales.

Table 1. Most recent abundance estimate (Nbest) and coefficient of variation (CV) of Gervais’ beaked whales in the northern Gulf of Mexico oceanic waters (200 m to the offshore extent of the EEZ) based on an average from summer 2017 and summer/fall 2018 vessel surveys.

<table>
<thead>
<tr>
<th>Years</th>
<th>Area</th>
<th>Nbest</th>
<th>CV</th>
</tr>
</thead>
<tbody>
<tr>
<td>2017, 2018</td>
<td>U.S. Gulf of Mexico</td>
<td>20</td>
<td>0.98</td>
</tr>
</tbody>
</table>
Table 1. Summary of abundance estimates for northern Gulf of Mexico *Mesoplodon* spp., which is a combined estimate for Gervais' beaked whale and Blainville’s beaked whale. Month, year and area covered during each abundance survey, and resulting abundance estimate ($N_{\text{best}}$) and coefficient of variation (CV).

<table>
<thead>
<tr>
<th>Month/Year</th>
<th>Area</th>
<th>$N_{\text{best}}$</th>
<th>CV</th>
</tr>
</thead>
<tbody>
<tr>
<td>Apr-Jun 1996-2001 (excluding 1998) Oceanic waters</td>
<td>106</td>
<td>0.41</td>
<td></td>
</tr>
<tr>
<td>Jun-Aug 2003, Apr-Jun 2004 Oceanic waters</td>
<td>57</td>
<td>1.40</td>
<td></td>
</tr>
<tr>
<td>Jun-Aug 2009 Oceanic waters</td>
<td>149</td>
<td>0.91</td>
<td></td>
</tr>
</tbody>
</table>

Minimum Population Estimate

The minimum population estimate ($N_{\text{min}}$) is the lower limit of the two-tailed 60% confidence interval of the log-normal distributed abundance estimate. This is equivalent to the 20th percentile of the log-normal distributed abundance estimate as specified by Wade and Angliss (1997). The best estimate of abundance for *Mesoplodon* spp. Gervais’ beaked whale is 20,149 (CV=0.9891). The minimum population estimate for the northern Gulf of Mexico Gervais’ beaked whale *Mesoplodon* spp. in the northern Gulf of Mexico is 1077 (Table 2).

Current Population Trend

The statistical power to detect a trend in abundance for this stock is poor due to the relatively imprecise abundance estimates and long intervals between surveys. For example, the power to detect a precipitous decline in abundance (i.e., 50% decrease in 15 years) with estimates of low precision (e.g., CV>0.30) remains below 80% (alpha=0.30) unless surveys are conducted on an annual basis (Taylor et al. 2007). In addition, because these surveys are restricted to U.S. waters, it is not possible to distinguish between changes in population size and Gulf-wide shifts in spatial distribution. Five point estimates of Ziphiidae abundance have been made based on data from surveys during: 2003 (June-August), 2004 (April-June), 2009 (July-August), 2017 (July-August), and 2018 (August-October). Each of these surveys had a similar design and was conducted using the same vessel or a vessel with a similar observation platform. Prior year surveys only employed a single survey team. The estimates for 2017 and 2018 were conducted in "passing" mode which resulted in increased numbers of unidentified sightings and may have affected group size estimation for distant groups of dolphins and small whales. Apportioning unidentified species among identified taxa addresses the first issue, and there were no apparent relationships between sighting distance and estimated group size. The model for detection probability on the trackline from the summer 2017 survey was applied to the abundance estimates from the 2003, 2004, and 2009 surveys. The resulting abundance estimates were: 2003, N=573 (CV=0.44); 2004, N=55 (CV=0.72); 2009, N=276 (CV=0.59); 2017, N=303 (CV=0.49); and 2018, N=322 (CV=0.34). Pairwise comparisons of the log-transformed means were conducted between years, and significant differences were assessed at alpha=0.10. P-values were adjusted for multiple comparisons. There was a significant decrease between the 2003 and 2004 estimates (p.adjusted=0.012) and a significant increase between the 2004 and 2018 estimates (p.adjusted =0.067), however there is no clear pattern in the overall trend. There are insufficient data to determine the population trends for this species due to uncertainty in species identification at sea. Three point estimates of *Mesoplodon* spp. abundance have been made based on data from surveys covering 1996-2009. The estimates vary by a maximum factor of nearly three. To determine whether changes in abundance have occurred over this period, an analysis of all the survey data needs to be conducted which incorporates covariates (e.g., survey conditions, season) that could potentially affect estimates. Nevertheless, differences in temporal abundance estimates will still be difficult to interpret without...
a Gulf of Mexico-wide understanding of *Mesoplodon* abundance. The oceanography of the Gulf of Mexico is quite dynamic, and the spatial scale of the Gulf is small relative to the ability of most cetacean species to travel. Studies based on abundance and distribution surveys restricted to U.S. waters are unable to detect temporal shifts in distribution beyond U.S. waters that might account for any changes in abundance.

**CURRENT AND MAXIMUM NET PRODUCTIVITY RATES**

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

**POTENTIAL BIOLOGICAL REMOVAL**

Potential Biological Removal (PBR) is the product of the minimum population size, one half the maximum net productivity rate and a recovery factor (MMPA Sec. 3.16 U.S.C. 1362; Wade and Angliss 1997). The minimum population size for *Gervais’ beaked whale* *Mesoplodon spp.* is 1077. The maximum productivity rate is 0.04, the default value for cetaceans. The recovery factor, which accounts for endangered, depleted, threatened stocks, or stocks of unknown status relative to optimum sustainable population (OSP), is assumed to be 0.5 because the stock is of unknown status. PBR for the northern Gulf of Mexico *Gervais’ beaked whale* *Mesoplodon spp.* is 0.108 (Table 2). It is not possible to determine the PBR for only Gervais’ beaked whales.

<table>
<thead>
<tr>
<th>Table 2. Best and minimum abundance estimates for Gulf of Mexico Gervais’ beaked whales with Maximum Productivity Rate (Rmax), Recovery Factor (Fr) and PBR.</th>
</tr>
</thead>
<tbody>
<tr>
<td>Nbest</td>
</tr>
<tr>
<td>20</td>
</tr>
</tbody>
</table>

**ANNUAL HUMAN-CAUSED MORTALITY AND SERIOUS INJURY**

Total annual estimated fishery-related mortality and serious injury to this stock during 2014–2018 was presumed to be zero, as there were no reports of mortalities or serious injuries to Gervais’ beaked whales or unidentified beaked whales in the Gulf of Mexico (Table 3). Mean annual mortality and serious injury during 2014–2018 for all beaked whales due to other human-caused actions (the Deepwater Horizon oil spill) was predicted to be 5.2. The minimum total mean annual human-caused mortality and serious injury for beaked whales during 2014–2018 was, therefore, 5.2. This is a combined estimate for Blainville’s, Gervais’, and Cuvier’s beaked whales. The minimum total mean annual human-caused mortality and serious injury for Gervais’ beaked whale is unknown.

**Table 3. Total annual estimated fishery-related mortality and serious injury for northern Gulf of Mexico Gervais’ beaked whales.**

<table>
<thead>
<tr>
<th>Years</th>
<th>Source</th>
<th>Annual Avg.</th>
<th>CV</th>
</tr>
</thead>
<tbody>
<tr>
<td>2014–2018</td>
<td>U.S. fisheries using observer data</td>
<td>0</td>
<td>z</td>
</tr>
</tbody>
</table>

There has been no reported fishing-related mortality of a beaked whale during 1998-2010 (Yeung 1999; Yeung 2001; Garrison 2003; Garrison and Richards 2004; Garrison 2005; Fairfield-Walsh and Garrison 2006; Fairfield-Walsh and Garrison 2007; Fairfield and Garrison 2008; Garrison *et al.* 2009; Garrison and Stokes 2010; 2011). However, during 2007 there was 1 unidentified beaked whale released alive with no serious injury after an entanglement interaction with the pelagic longline fishery (Fairfield and Garrison 2008).

Fisheries Information

There are two commercial fisheries that interact, or that could potentially interact, with this stock in the Gulf of Mexico. These are the Category I Atlantic Highly Migratory Species (high seas) longline fishery and the Category I Atlantic Ocean, Caribbean, Gulf of Mexico large pelagics longline fishery (Appendix III). Percent observer coverage (percentage of sets observed) for these longline fisheries for each year during 2014–2018 was 18, 19, 23, 13 and 20, respectively. There is very little effort within the Gulf of Mexico by the Atlantic Highly Migratory Species (high seas) longline fishery, and no takes of beaked whales within high seas waters of the Gulf of Mexico have been observed or reported thus far. Pelagic swordfish, tunas and billfish are the targets of the pelagic longline fishery operating in the northern Gulf of Mexico. There were no reports of mortality or serious injury to Gervais’ or other beaked whales by this fishery during 2014–2018 (Garrison and Stokes 2016; 2017; 2019; 2020; in review).
The commercial fishery which potentially could interact with this stock in the Gulf of Mexico is the Atlantic Ocean, Caribbean, Gulf of Mexico large pelagic longline fishery (Appendix III). Pelagic swordfish, tunas and billfish are the targets of the longline fishery operating in the northern Gulf of Mexico. There were no reports of mortality or serious injury to Gervais’ or other beaked whales by this fishery during 1998–2010 (Yeung 1999; Yeung 2001; Garrison 2003; Garrison and Richards 2004; Garrison 2005; Fairfield Walsh and Garrison 2006; Fairfield-Walsh and Garrison 2007; Fairfield and Garrison 2008; Garrison et al. 2009; Garrison and Stokes 2010; 2011). However, during 2007, 1 unidentified beaked whale was observed entangled and released alive in the northern Gulf of Mexico. All gear was removed and the animal was presumed to have no serious injuries (Fairfield and Garrison 2008).

Other Mortality

There were no four strandings of Gervais’ beaked whales Mesoplodon spp. or unidentified beaked whales during 2006–2014–2018 (NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 16 November 2011 21 May 2019). All four strandings occurred in Florida. For one stranding, there was evidence of human interaction (the interaction being the animal was pushed out to sea by the public). For the remaining three strandings, it could not be determined whether there was evidence of human interaction. Stranding data underestimate the extent of human and fishery-related mortality and serious injury because not all of the marine mammals that die or are seriously injured in human interactions wash ashore, or, if they do, they are not all recovered (Peltier et al. 2012; Wells et al. 2015). In particular, oceanic stocks in the Gulf of Mexico are less likely to strand than nearshore coastal stocks or shelf stocks (Williams et al. 2011). Additionally, not all carcasses will show evidence of human interaction, entanglement or other fishery-related interaction due to decomposition, scavenger damage, etc. (Byrd et al. 2014). Finally, the level of technical expertise among stranding network personnel varies widely as does the ability to recognize signs of human interaction.

Stranding data probably underestimate the extent of fishery-related mortality and serious injury because not all of the marine mammals which die or are seriously injured in fishery interactions wash ashore, not all that wash ashore are discovered, reported or investigated, 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 interactions.

An Unusual Mortality Event (UME) was declared for cetaceans in the northern Gulf of Mexico beginning 1 FebruaryMarch 2010 and ending 31 July 2014 (Litz et al. 2014; https://www.fisheries.noaa.gov/national/marine-life-distress/2010-2014-cetacean-unusual-mortality-event-northern-gulf-mexico) and, as of early 2012, the event is still ongoing. It includes cetaceans that stranded prior to the Deepwater Horizon (DWH) oil spill (see “Habitat Issues” below), during the spill, and after. Exposure to the DWH oil spill was determined to be the primary underlying cause of the elevated stranding numbers in the northern Gulf of Mexico after the spill (e.g., Schwacke et al. 2014; Venn-Watson et al. 2015; Colegrove et al. 2016; DWH NRDAT 2016; see Habitat Issues section). However, there were no Gervais’ beaked whale strandings recovered within the spatial and temporal boundaries of this UME During 2010, no animals from this stock were considered to be part of the UME.

A population model was developed to estimate the injury and time to recovery for stocks affected by the DWH oil spill, taking into account long-term effects resulting from mortality, reproductive failure, reduced survival rates, and the proportion of the stock exposed to DWH oil (DWH MMIQT 2015). Overall, this model estimated that the stocks experienced a 6% maximum reduction in population size due to the oil spill (DWH MMIQT 2015). The mortality projected for the years 2010–2013 due to the spill has not been reported previously. Based on the population model, it was projected that 51 beaked whales died during 2010–2013 (four year annual average of 13) due to elevated mortality associated with oil exposure (see Appendix VI). For the 2014–2018 reporting period of this SAR, the population model estimated 26 beaked whales died due to elevated mortality associated with oil exposure. However, this mortality estimate is not comparable to the current abundance estimate derived from visual surveys because the population model included a correction factor for detection probability derived from acoustic density estimates (DWH MMIQT 2015). The population model used to predict beaked whale mortality due to the DWH event has a number of sources of uncertainty. Model parameters (e.g., survival rates, reproductive rates, and life-history parameters) were derived from literature sources for beaked whales occupying waters outside of the Gulf of Mexico. In addition, proxy values for the effects of DWH oil exposure on both survival rates and reproductive success were applied based upon estimated values for common bottlenose dolphins in Barataria Bay. Finally, there was no estimation of uncertainty in model parameters or outputs.
Several unusual mass strandings of beaked whales in North Atlantic marine environments have been associated
with military naval activities. 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 whales 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 active sonar tests
conducted by the North Atlantic Treaty Organization (Frantzis 1998). In March 2000, 14 beaked whales live stranded
in the Bahamas; 6 beaked whales (5 Cuvier’s and 1 Blainville’s) died (Evans and England 2001; Balcomb and Claridge
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. Necropsies were performed on 5 of the dead beaked whales and 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) (Evans and England 2001; Cox et al. 2006).

HABITAT ISSUES

The Deepwater Horizon (DWH) MC252 drilling platform, located approximately 50 miles80 km southeast of the
Mississippi River Delta in waters about 1,500 m deep, exploded on 20 April 2010. The rig sank, and for over 87 days
~3.2 million barrels of oil and gas were discharged from the wellhead until it was capped on 15 July 2010 (DWH
NRDAT 2016). During the response effort dispersants were applied extensively at the seafloor and at the sea surface
(Lehr et al. 2010; OSAT 2010). In situ burning, or controlled burning of oil at the surface, was also used extensively
as a response tool (Lehr et al. 2010). The oil, dispersant and burn residue compounds present ecological concerns.
The magnitude of this oil spill was unprecedented in U.S. history, causing impacts to wildlife, natural habitats and
human communities along coastal areas from western Louisiana to the Florida Panhandle (NOAA 2011). It could be
years before the entire scope of damage is ascertained (NOAA 2011).

Shortly after the oil spill, the Natural Resource Damage Assessment (NRDA) process was initiated under the Oil
Pollution Act of 1990. A variety of NRDA research studies were conducted to determine potential impacts
of the spill on marine mammals. These studies estimated that 12% (95% CI: 2–22) of beaked whales in the Gulf,
which included Blainville’s, Cuvier’s and Gervais’ beaked whales, were exposed to oil, that 5% (95% CI: 3–8) of
females suffered from reproductive failure, and 4% (95% CI: 2–7) of the beaked whale populations suffered adverse
health effects (DWH MMIQT 2015). A population model estimated the stocks experienced a maximum 6% reduction
in population size (see Other Mortality section above).

Anthropogenic sound in the world’s oceans has been shown to affect marine mammals, with vessel traffic, seismic
surveys, and active naval sonars being the main anthropogenic contributors to low- and mid-frequency noise in oceanic
waters (e.g., Nowacek et al. 2015; Gomez et al. 2016; NMFS 2018). Anthropogenic noise, particularly from military
sonar, shipping, and seismic testing, is an increasing habitat concern for beaked whales (Aguilar de Soto et al. 2006;
Cox et al. 2006; McCarthy et al. 2011; Tyack et al. 2011; Joyce et al. 2020). Several mass strandings of beaked whales
throughout their worldwide range have been associated with naval activities (D’Amico et al. 2009; Filadelfo et al.
2009). In March 2000, 14 beaked whales live stranded in the Bahamas. Six of the whales (5 Cuvier’s and 1
Blainville’s) died and necropsy revealed evidence of tissue trauma associated with an acoustic or impulse injury that
caused the animals to strand (Balcomb and Claridge 2001; NMFS 2001; Cox et al. 2006). Fourteen beaked whales
(mostly Cuvier’s beaked whales but also including Gervais’ and Blainville’s beaked whales) stranded in the Canary
Islands in 2002 (Cox et al. 2006; Fernandez et al. 2005; Martin et al. 2004). Gas bubble-associated lesions and fat
embolism were found in necropsied animals from this event, leading researchers to link nitrogen supersaturation with
sonar exposure (Fernandez et al. 2005). The long-term and population consequences of these impacts are less well-
documented and likely vary by species and other factors. Impacts on marine mammal prey from sound are also possible
(Carroll et al. 2017), but the duration and severity of any such prey effects on marine mammals are unknown. Finally,
ingestion of marine debris, particularly plastics, is a concern; plastic is occasionally found in the stomach contents of
stranded beaked whales.

These studies have focused on identifying the type, magnitude, severity, length and impact of oil exposure to oceanic,
coastal and estuarine marine mammals. The research is ongoing and likely will continue for some time. For continental
shelf and oceanic cetaceans, the NOAA-led efforts include: aerial surveys to document the distribution, abundance,
species and exposure of marine mammals and turtles relative to oil from DWH spill; and ship surveys to evaluate
exposure to oil and other chemicals and to assess changes in animal behavior and distribution relative to oil exposure.
through visual and acoustic surveys, deployment of passive acoustic monitoring systems, collection of tissue samples, and deployment of satellite tags on sperm and Bryde’s whales. Aerial surveys have observed Risso’s dolphins, spinner dolphins, pantropical spotted dolphins, striped dolphins, bottlenose dolphins and sperm whales swimming in oil in offshore waters (NOAA 2010a). The effects of oil exposure on marine mammals depend on a number of factors including the type and mixture of chemicals involved, the amount, frequency and duration of exposure, the route of exposure (inhaled, ingested, absorbed, or external) and biomedical risk factors of the particular animal (Geraci 1990; NOAA 2010b). In general, direct external contact with petroleum compounds or dispersants with skin may cause skin irritation, chemical burns and infections. Inhalation of volatile petroleum compounds or dispersants may irritate or injure the respiratory tract, which could lead to pneumonia or inflammation. Ingestion of petroleum compounds may cause injury to the gastrointestinal tract, which could affect an animal’s ability to digest or absorb food. Absorption of petroleum compounds or dispersants may damage kidney, liver and brain function in addition to causing immune suppression and anemia. Long-term chronic effects such as lowered reproductive success and decreased survival may occur (Geraci 1990; NOAA 2010b).

STATUS OF STOCK

Gervais’ beaked whales are not listed as threatened or endangered under the Endangered Species Act, and the northern Gulf of Mexico stock is not considered strategic under the MMPA because PBR is likely a severe underestimate due to the long dive times of this species and because the mean modeled annual human-caused mortality and serious injury due to the DWH oil spill is based on all beaked whale species combined and cannot be apportioned to individual species. No fishery-related mortality or serious injury has been observed; therefore, total fishery-related mortality and serious injury can be considered insignificant and approaching the zero mortality and serious injury rate. The status of Gervais’ beaked whales in the U.S. EEZ relative to OSP is unknown. There are insufficient data to determine the population trends for this stock. The status of Gervais’ beaked whales or other beaked whales in the northern Gulf of Mexico, relative to OSP, is unknown. The species is not listed as threatened or endangered under the Endangered Species Act. There are insufficient data to determine the population trends for this species. Total human-caused mortality and serious injury for this stock is not known but none has been documented. There is insufficient information available to determine whether the total fishery-related mortality and serious injury for this stock is insignificant and approaching zero mortality and serious injury rate. This is not a strategic stock because it is assumed that the average annual human-related mortality and serious injury does not exceed PBR.- Disturbance by anthropogenic noise may prove to be an important habitat issue in some areas of this population’s range, notably in areas of oil and gas activities or where shipping or naval activities are high. Limited studies are currently being conducted to address this issue and its impact, if any, on this and other marine species.

REFERENCES


Garrison, L.P. and L. Stokes. in review. Estimated bycatch of marine mammals and sea turtles in the U.S. Atlantic pelagic longline fleet during 2018. Southeast Fisheries Science Center, Protected Resources and Biodiversity Division, 75 Virginia Beach Dr., Miami, Florida 33140. PRD Contribution # PRD-2020-08. 55 pp.


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

STOCK DEFINITION AND GEOGRAPHIC RANGE

Thirty-six common bottlenose dolphin stocks have been designated in the northern Gulf of Mexico (i.e., U.S. Gulf of Mexico) (Waring et al. 2016, 2001). Northern Gulf of Mexico inshore habitats have been separated into 31 bay, sound and estuary stocks. Three northern Gulf of Mexico coastal stocks inhabit coastal waters from the shore to the 20-m isobath. The northern Gulf of Mexico Continental Shelf Stock inhabits waters from 20 to 200 m deep. The northern Gulf of Mexico Oceanic Stock inhabits the waters from the 200-m isobath to the seaward extent of the U.S. Exclusive Economic Zone (EEZ; Figure 1).

Both “coastal” and “offshore” ecotypes of common bottlenose dolphins (Mead and Potter 1995) occur in the Gulf of Mexico (Vollmer 2011; Vollmer and Rosel 2013), but the distribution of each is not well defined. The offshore and coastal ecotypes are genetically distinct based on both mitochondrial and nuclear markers (Hoelzel et al. 1998; Vollmer 2011). In the northwestern Atlantic Ocean, Torres et al. (2003) found a statistically significant break in the distribution of the ecotypes at 34 km from shore. The offshore ecotype was found exclusively seaward of 34 km and in waters deeper than 34 m. The continental shelf is much wider in the Gulf of Mexico and these results may not apply. Ongoing research is aimed at better defining stock boundaries in coastal, continental shelf and oceanic waters of the Gulf of Mexico. Although the boundaries are not certain, all 141 *Tursiops* samples collected during 1994–2008 in waters greater than 200 m were of the offshore ecotype (Vollmer 2011), and so the Oceanic Stock as currently defined is thought to be composed entirely of bottlenose dolphins of the offshore ecotype.

All the cetacean species found in the oceanic northern Gulf of Mexico almost certainly occur in similar habitat beyond U.S. boundaries in the southern Gulf. There are fewer cetacean sighting and stranding records in the southern Gulf due to more limited effort. Nevertheless, there are records for most oceanic species in the southern Gulf (e.g., Ortega Ortiz 2002; Ortega-Argueta et al. 2005; Jefferson et al. 2008; Vázquez Castán et al. 2009; Whitt et al. 2011). This is therefore likely a transboundary stock with both Cuba and Mexico. Because U.S. waters only comprise about 40% of the entire Gulf of Mexico and 35% of the oceanic (i.e., >200 m) Gulf of Mexico (Mullin and Fulling 2004), abundance and stock boundaries of oceanic species are poorly known.
Because there are many confirmed records from Gulf of Mexico waters beyond U.S. boundaries (e.g., Ortega Ortiz 2002), bottlenose dolphins almost certainly occur throughout the oceanic Gulf of Mexico (Jefferson et al. 2008), including waters belonging to Mexico and Cuba, where there is currently little information on cetacean species abundance and distribution. U.S. waters only comprise about 40% of the entire Gulf of Mexico and 35% of the oceanic (i.e., >200 m) Gulf of Mexico.

The northern Gulf of Mexico Oceanic Stock of common bottlenose dolphins is being managed separately from the western North Atlantic Offshore Stock of common bottlenose dolphins. Atlantic Ocean stocks of bottlenose dolphins for management purposes. One line of evidence to support this decision comes from Baron et al. (2008), who found that Gulf of Mexico common bottlenose dolphin whistles (collected from oceanic waters) were significantly different from those in the western North Atlantic Ocean (collected from continental shelf and oceanic waters) in duration, number of inflection points and number of steps. Coupled with evidence for population structure in other areas and the fact that the western North Atlantic and Gulf of Mexico belong to distinct marine ecoregions (Spalding et al. 2007), designation delimitation of the two stocks is reasonable and consistent with maintaining stocks as functioning elements of their ecosystems. Restricted genetic exchange has been documented among offshore populations within the Gulf of Mexico, suggesting multiple demographically-independent populations of the offshore morphotype exist (Vollmer and Rosel 2017).

POPULATION SIZE

The best abundance estimate (Nbest) for the northern Gulf of Mexico Oceanic Stock of common bottlenose dolphins is 7,462 (CV=0.31; Table 1). This estimate is from summer 2017 and summer/fall 2018 oceanic surveys covering waters from the 200-m isobath to the seaward extent of the U.S. EEZ (Garrison et al. 2020). This estimate is from a summer 2009 oceanic survey covering waters from the 200 m isobath to the seaward extent of the U.S. EEZ.

Earlier abundance estimates

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

Recent surveys and abundance estimates

An abundance estimate for the oceanic stock of common bottlenose dolphins was generated from vessel surveys conducted in the northern Gulf of Mexico from the continental shelf edge (~200-m isobath) to the seaward extent of the U.S. EEZ (Garrison et al. 2020). One survey was conducted from 2 July to 25 August 2017 and consisted of 5,104 km of on-effort trackline, and the second survey was conducted from 11 August to 6 October 2018 and consisted of 5,205 km of on-effort trackline within the surveyed strata. Both surveys used a double-platform data-collection procedure, which allowed estimation of the detection probability on the trackline using the independent observer approach assuming point independence (Laake and Borchers 2004). Abundance was calculated using mark-recapture distance sampling implemented in package mrds (version 2.21, Laake et al. 2020) in the R statistical programming language. This approach accounted for the effects of covariates (e.g., sea state, glare) on detection probability within the surveyed strip. The surveys were conducted in "passing mode" (e.g., Schwarz et al. 2010) while all prior surveys in the Gulf of Mexico have been conducted in "closing mode." The abundance estimate for this stock included sightings of unidentified dolphins that were apportioned among identified species based on their relative density within the survey strata (Garrison et al. 2020). The inverse variance weighted mean abundance estimate for common bottlenose dolphins in oceanic waters during 2017 and 2018 was 7,462 (CV=0.31; Table 1; Garrison et al. 2020). Unlike previous abundance estimates, this estimate was corrected for the probability of detection on the trackline. During summer 2009, a vessel-based line-transect survey dedicated to estimating the abundance of oceanic cetaceans was conducted in the northern Gulf of Mexico. Survey lines were stratified in relation to depth and the location of the Loop Current. The abundance estimate for bottlenose dolphins in oceanic waters during 2009 was 5,806 (CV=0.39; Table 1).

Table 1. Most recent abundance estimate (Nbest) and coefficient of variation (CV) of northern Gulf of Mexico common bottlenose dolphins in oceanic waters (200 m to the offshore extent of the EEZ) based on an average from summer 2017 and summer/fall 2018 vessel surveys.
Table 1. Summary of recent abundance estimates for the northern Gulf of Mexico oceanic stock of bottlenose dolphins. Month, year and area covered during each abundance survey, and resulting abundance estimate (Nbest) and coefficient of variation (CV).

<table>
<thead>
<tr>
<th>Month/Year</th>
<th>Area</th>
<th>Nbest</th>
<th>CV</th>
</tr>
</thead>
<tbody>
<tr>
<td>Jun-Aug 2009</td>
<td>Oceanic waters</td>
<td>5,806</td>
<td>0.39</td>
</tr>
<tr>
<td>2017, 2018</td>
<td>Gulf of Mexico</td>
<td>7,462</td>
<td>0.31</td>
</tr>
</tbody>
</table>

**Minimum Population Estimate**

The minimum population estimate (Nmin) is the lower limit of the two-tailed 60% confidence interval of the log-normal distributed abundance estimate. This is equivalent to the 20th percentile of the log-normal distributed abundance estimate as specified by Wade and Angliss (1997). The best estimate of abundance for common bottlenose dolphins is \( 7,462 \pm 5,806 \) (CV=0.3139). The minimum population estimate for the northern Gulf of Mexico oceanic stock of common bottlenose dolphin is 5,769 (Table 2). 4,230 bottlenose dolphins.

**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 intervals between surveys. For example, the power to detect a precipitous decline in abundance (i.e., 50% decrease in 15 years) with estimates of low precision (e.g., CV > 0.30) remains below 80% (alpha = 0.30) unless surveys are conducted on an annual basis (Taylor et al. 2007). Three point estimates of oceanic bottlenose dolphin abundance have been made based on data from surveys covering 1996-2009. The estimates vary by a maximum factor of more than two. Nevertheless, differences in temporal abundance estimates will still be difficult to interpret without a Gulf of Mexico-wide understanding of oceanic bottlenose dolphin abundance. The oceanography of the Gulf of Mexico is quite dynamic, and the spatial scale of the Gulf is small relative to the ability of most cetacean species to travel. Studies based on abundance and distribution surveys restricted to U.S. waters are unable to detect temporal shifts in distribution beyond U.S. waters that might account for any changes in abundance. In addition, because these surveys are restricted to U.S. waters, it is not possible to distinguish between changes in population size and Gulf-wide shifts in spatial distribution. Five point estimates of oceanic common bottlenose dolphin abundance have been made based on data from surveys during 2003 (June–August), 2004 (April–June), 2009 (July–August), 2017 (July–August), and 2018 (August–October). Each of these surveys had a similar design and was conducted using the same vessel or a vessel with a similar observation platform. Prior year surveys only employed a single survey team. The estimates for 2017 and 2018 were conducted in "passing" mode which resulted in increased numbers of unidentified sightings and may have affected group size estimation for distant groups of dolphins and small whales. Apportioning unidentified species among identified taxa addresses the first issue, and there were no apparent relationships between sighting distance and estimated group size. The model for detection probability on the trackline from the summer 2017 survey was applied to the abundance estimates from the 2003, 2004, and 2009 surveys. The resulting abundance estimates were: 2003, N=21,350 (CV=0.47); 2004, N=8,864 (CV=0.50); 2009, N=9,640 (CV=0.66); 2017, N=8,756 (CV=0.41); and 2018, N=5,833 (CV=0.46). Pairwise comparisons of the log-transformed means were conducted between years, and significant differences were assessed at alpha=0.10. P-values were adjusted for multiple comparisons. There were no significant differences between survey years.

**CURRENT AND MAXIMUM NET PRODUCTIVITY RATES**

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

**POTENTIAL BIOLOGICAL REMOVAL**

Potential Biological Removal (PBR) is the product of minimum population size, one-half the maximum productivity rate and a recovery factor (MMPA Sec. 3. 16 U.S.C. 1362; Wade and Angliss 1997). The minimum population size is \(5,769\). The maximum productivity rate is 0.04, the default value for cetaceans. The recovery factor, which accounts for endangered, depleted, threatened stocks, or stocks of unknown status relative to optimum sustainable population (OSP), is assumed to be 0.5 because the stock is of unknown status. PBR for the Gulf of Mexico oceanic common bottlenose dolphin is 58 (Table 2).

### Table 2. Best and minimum abundance estimates for northern Gulf of Mexico oceanic common bottlenose dolphins with Maximum Productivity Rate (Rmax), Recovery Factor (Fr) and PBR.

<table>
<thead>
<tr>
<th>N_{best}</th>
<th>CV</th>
<th>N_{min}</th>
<th>Fr</th>
<th>R_{max}</th>
<th>PBR</th>
</tr>
</thead>
<tbody>
<tr>
<td>7,462</td>
<td>0.31</td>
<td>5,769</td>
<td>0.5</td>
<td>0.04</td>
<td>58</td>
</tr>
</tbody>
</table>

**ANNUAL HUMAN-CAUSED MORTALITY AND SERIOUS INJURY**

Total annual estimated fishery-related mortality and serious injury to this stock during 2014–2018 was presumed to be zero, as there were no reports of mortalities or serious injuries to oceanic bottlenose dolphins in the Gulf of Mexico (Table 3). Mean annual mortality and serious injury during 2014–2018 due to other human-caused actions (the Deepwater Horizon oil spill) was predicted to be 32. The minimum total mean annual human-caused mortality and serious injury for this stock during 2014–2018 was, therefore, 32. The estimated annual average fishery-related mortality and serious injury to this stock during 2008–2012 was 6.5 bottlenose dolphins (CV=0.65; Table 2).

### Table 3. Total annual estimated fishery-related mortality and serious injury for northern Gulf of Mexico oceanic common bottlenose dolphins.

<table>
<thead>
<tr>
<th>Years</th>
<th>Source</th>
<th>Annual Avg.</th>
<th>CV</th>
</tr>
</thead>
<tbody>
<tr>
<td>2014–2018</td>
<td>U.S. fisheries using observer data</td>
<td>0</td>
<td></td>
</tr>
</tbody>
</table>

**New Serious Injury Guidelines**

NMFS updated its serious injury designation and reporting process, which uses guidance from previous serious injury workshops, expert opinion, and analysis of historic injury cases to develop new criteria for distinguishing serious from non-serious injury (Angliss and DeMaster 1998; Andersen et al. 2008; NOAA 2012). NMFS defines serious injury as an “injury that is more likely than not to result in mortality”. Injury determinations for stock assessments revised in 2013 or later incorporate the new serious injury guidelines, based on the most recent 5-year period for which data are available.

**Fisheries Information**

There are three commercial fisheries that interact, or that could potentially interact, with this stock in the Gulf of Mexico: the Category I Atlantic Highly Migratory Species (high seas) longline fishery and Atlantic Ocean, Caribbean, Gulf of Mexico large pelagics longline fishery; and the Category III Gulf of Mexico butterfish trawl fishery (Appendix III).

Percent observer coverage (percentage of sets observed) for the two Category I longline fisheries for each year during 2014–2018 was 18, 19, 23, 13 and 20, respectively. There is very little effort within the Gulf of Mexico by the Atlantic Highly Migratory Species (high seas) longline fishery, and no takes of common bottlenose dolphins within high seas waters of the Gulf of Mexico have been observed or reported thus far. Pelagic swordfish, tunas and billfish are the targets of the large pelagics longline fishery operating in the northern Gulf of Mexico and during 2014–2018 there were no observed mortalities or serious injuries to common bottlenose dolphins by this fishery (Garrison and Stokes 2016; 2017; 2019; 2020; in review). The commercial fisheries that could potentially interact with this stock in the Gulf of Mexico are the Category I Atlantic Ocean, Caribbean, Gulf of Mexico large pelagics longline fishery and
the Atlantic Highly Migratory Species (high seas longline) fishery. The Category III Gulf of Mexico butterfish trawl fishery may also interact with this stock (Appendix III). There is very little effort within the Gulf of Mexico by the high seas longline fishery, and no takes of bottlenose dolphins within high seas waters of the Gulf of Mexico have been observed or reported thus far.

Pelagic swordfish, tunas and billfish are the targets of the pelagic longline fishery operating in the northern Gulf of Mexico. There was 6.5 animals (CV=0.65; Table 2). There were no reports of mortality or serious injury to bottlenose dolphins by this fishery in the northern Gulf of Mexico during 1999-2008 (Yeung 1999; Yeung 2001; Garrison 2003; Garrison and Richards 2004; Garrison 2005; Fairfield Walsh and Garrison 2006; Fairfield Walsh and Garrison 2007; Fairfield and Garrison 2008; Garrison et al. 2009). However, during 2009, 1 serious injury of a bottlenose dolphin was observed during the second quarter (Garrison and Stokes 2010). During 2010, 1 serious injury was observed in the second quarter during experimental fishing to test the effectiveness of “weak” hooks as a potential bycatch mitigation tool. There was 100% observer coverage of all experimental sets, and the experimental fishing is not included in extrapolated bycatch estimates because it is not representative of the normal fishing effort (Garrison and Stokes 2012a). Again during 2012, 1 serious injury of a bottlenose dolphin was observed during the fourth quarter (Garrison and Stokes 2013). From earlier years, 1 bottlenose dolphin was observed entangled and released alive in the northern Gulf of Mexico during 2007. All longline gear was removed and the animal was presumed to have no serious injuries. One bottlenose dolphin serious injury was observed in the pelagic longline fishery in 1998, and estimated serious injuries attributable to the pelagic longline fishery in the Gulf of Mexico region during quarter 1 of that year were 22 (CV=1.00; Yeung 1999).

The Category III Gulf of Mexico butterfish trawl fishery may also interact with this stock (Appendix III). A trawl fishery for butterfish was monitored by NMFS observers for a short period in the 1980's with no records of incidental take of marine mammals (Burn and Scott 1988; NMFS unpublished data), although an experimental set by NMFS resulted in the death of 2 common bottlenose dolphins (Burn and Scott 1988). There are no other data available with regard to this fishery.

### Table 2

<table>
<thead>
<tr>
<th>Fishery</th>
<th>Years</th>
<th>Vessels</th>
<th>Data Type</th>
<th>Observ. Coverage</th>
<th>Observed Serious Injury</th>
<th>Estimated Serious Injury</th>
<th>Estimated Combined Mortality</th>
<th>Est. -CVs</th>
<th>Mean Annual Mortality</th>
</tr>
</thead>
<tbody>
<tr>
<td>Pelagic Longline</td>
<td>08-12</td>
<td>53, 47, 46, 42, 47</td>
<td>Obs. Logbook</td>
<td>0.26, 0.22, 0.28, 0.18, 0.14</td>
<td>0.0, 0.0, 0.0, 0.0, 0.0</td>
<td>0.3, 0.2, 0.3, 0.4, 0.5</td>
<td>0.0, 0.0, 0.0, 0.0, 0.0</td>
<td>NA, 1.0, NA, 1.0, 4.0</td>
<td>6.5 (0.65)</td>
</tr>
</tbody>
</table>
Other Mortality

A total of 1,764 common bottlenose dolphins were found stranded in the northern Gulf of Mexico from 2008 through 2012 (NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 13 September 2012 and 1 May 2019 [for 2008-2011 data] and 15 April 2017 [for 2012 data]). Of these, 177 showed evidence of human interactions (e.g., gear entanglement, mutilation, gunshot wounds). The vast majority of stranded common bottlenose dolphins are assumed to belong to the coastal stocks or to bay, sound, and estuary stocks. Nevertheless, it is possible that some of the stranded common bottlenose dolphins belonged to the continental shelf or oceanic stocks and that they were among those strandings with evidence of human interactions. Stratifications do occur for other cetacean species whose primary range in the Gulf of Mexico is near coastal shelf or oceanic waters, but oceanic stocks in the Gulf of Mexico are less likely to strand than nearshore coastal stocks or shelf stocks (Williams et al. 2011).

An Unusual Mortality Event (UME) was declared for cetaceans in the northern Gulf of Mexico beginning 1 February and ending 31 July 2014 (Litz et al. 2014; https://www.fisheries.noaa.gov/national/marine-life-distress/2010-2014-cetacean-unusual-mortality-event-northern-gulf-mexico) and, as of 2013, the event is still ongoing. It includes cetaceans that stranded prior to the Deepwater Horizon (DWH) oil spill (see “Habitat Issues” below), during the spill, and after. Exposure to the DWH oil spill was determined to be the primary underlying cause of the elevated stranding numbers in the northern Gulf of Mexico after the spill (e.g., Schwacke et al. 2014; Venn-Watson et al. 2015; Colegrove et al. 2016; DWH NRDAT 2016; see Habitat Issues section). During 2010, 221 bottlenose dolphins were considered to be part of the UME; during 2011, 324 bottlenose dolphins, and during 2012, 151 bottlenose dolphins. During 2014, 126 common bottlenose dolphins were considered to be part of the UME. The vast majority of stranded common bottlenose dolphins are assumed to come from stocks that live nearest to land, namely the bay, sound and estuary stocks and the three coastal stocks. Nevertheless, it is possible that some of the stranded common bottlenose dolphins considered part of the UME belonged to the continental shelf or oceanic stocks, given the overlap in distribution between the spill and distribution of this population.

A population model was developed to estimate the injury and time to recovery for stocks affected by the DWH oil spill, taking into account long-term effects resulting from mortality, reproductive failure, reduced survival rates, and the proportion of the stock exposed to DWH oil (DWH MMIQT 2015). Overall, the model estimated that this stock experienced a 4% maximum reduction in population size due to the oil spill (DWH MMIQT 2015). The mortality projected for the years 2010–2013 due to the spill has not been reported previously. Based on the population model, it was projected that 308 oceanic common bottlenose dolphins died during 2010–2013 (four year annual average of 77) due to elevated mortality associated with oil exposure (see Appendix VI). For the 2014–2018 reporting period of this SAR, the population model, estimated 160 oceanic common bottlenose dolphins died due to elevated mortality associated with oil exposure. The population model used to predict oceanic common bottlenose dolphin mortality due to the DWH event has a number of sources of uncertainty. Model parameters (e.g., survival rates, reproductive rates, and life-history parameters) were derived from literature sources for common bottlenose dolphins occupying waters outside of the Gulf of Mexico. In addition, proxy values for the effects of DWH oil exposure on both survival rates and reproductive success were applied based upon estimated values for common bottlenose dolphins in Barataria Bay. Finally, there was no estimation of uncertainty in model parameters or outputs.

HABITAT ISSUES

The Deepwater Horizon (DWH) MC252 drilling platform, located approximately 50 miles (80 km) southeast of the
Mississippi River Delta in waters about 1,500 m deep, exploded on 20 April 2010. The rig sank, and over 87 days ~3.24.9 million barrels of oil were discharged from the wellhead until it was capped on 15 July 2010 (McNutt et al. 2012; DWH NRDAT 2016). During the response effort dispersants were applied extensively at the seafloor and at the sea surface (Lehr et al. 2010; OSAT 2010). In situ burning, or controlled burning of oil at the surface, was also used extensively as a response tool (Lehr et al. 2010). The oil, dispersant and burn residue compounds present ecological concerns. The magnitude of this oil spill was unprecedented in U.S. history, causing impacts to wildlife, natural habitats and human communities along coastal areas from western Louisiana to the Florida Panhandle (NOAA 2011). It could be years before the entire scope of damage is ascertained (NOAA 2011).

Shortly after the oil spill, the Natural Resource Damage Assessment (NRDA) process was initiated under the Oil Pollution Act of 1990. A variety of NRDA research studies were conducted to determine potential impacts of the spill on marine mammals. These studies estimated that 10% (95% CI: 5–10) of oceanic common bottlenose dolphins in the Gulf were exposed to oil, that 5% (95% CI: 2–6) of females suffered from reproductive failure, and 4% (95% CI: 1–6) of oceanic common bottlenose dolphins suffered adverse health effects (DWH MMIQT 2015). A population model estimated that the stock experienced a 4% maximum reduction in population size (see Other Mortality section above). These studies have focused on identifying the type, magnitude, severity, length and impact of oil exposure to oceanic, continental shelf, coastal and estuarine marine mammals. The research is ongoing and likely will continue for some time. For continental shelf and oceanic cetaceans, the NOAA-led efforts include: aerial surveys to document the distribution, abundance, species and exposure of marine mammals and turtles relative to oil from DWH spill; and ship surveys to evaluate exposure to oil and other chemicals and to assess changes in animal behavior and distribution relative to oil exposure through visual and acoustic surveys, deployment of passive acoustic monitoring systems, collection of tissue samples, and deployment of satellite tags on sperm and Bryde’s whales.

Aerial surveys have observed bottlenose dolphins, Risso’s dolphins, spinner dolphins, pantropical spotted dolphins, striped dolphins and sperm whales swimming in oil in offshore waters. Some bottlenose dolphins were seen swimming in oil near the wellhead, where water depths would suggest these dolphins belonged to the Oceanic Stock. The effects of oil exposure on marine mammals depend on a number of factors including the type and mixture of chemicals involved, the amount, frequency and duration of exposure, the route of exposure (inhaled, ingested, absorbed, or external) and biomedical risk factors of the particular animal (Geraci 1990). In general, direct external contact with petroleum compounds or dispersants with skin may cause skin irritation, chemical burns and infections. Inhalation of volatile petroleum compounds or dispersants may irritate or injure the respiratory tract, which could lead to pneumonia or inflammation. Ingestion of petroleum compounds may cause injury to the gastrointestinal tract, which could affect an animal’s ability to digest or absorb food. Absorption of petroleum compounds or dispersants may damage kidney, liver and brain function in addition to causing immune suppression and anemia. Long term chronic effects such as lowered reproductive success and decreased survival may occur (Geraci 1990).

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

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

STATUS OF STOCK

Common bottlenose dolphins are not listed as threatened or endangered under the Endangered Species Act, and the northern Gulf of Mexico Oceanic Stock is not considered strategic under the MMPA. No fishery-related mortality or serious injury has been observed in recent years; therefore, total fishery-related mortality and serious injury can be considered to be insignificant and approaching zero mortality and serious injury rate. 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 average annual human-related mortality and serious injury does not exceed PBR. The status of bottlenose dolphins, relative to OSP, in the northern Gulf of Mexico oceanic waters is unknown. There was no statistically significant trend in population size for this stock in the northern Gulf of Mexico. There are insufficient data to determine population trends for this stock.

REFERENCES CITED


DWH MMiQT. 2015. Models and analyses for the quantification of injury to Gulf of Mexico cetaceans from the Deepwater Horizon Oil Spill. MM TR.01 Schwacke Quantification.of.Injury.to.GOM.Cetaceans.


PANTROPICAL SPOTTED DOLPHIN (Stenella attenuata attenuata): Northern Gulf of Mexico Stock

STOCK DEFINITION AND GEOGRAPHIC RANGE

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). The Atlantic spotted dolphin occurs in two forms which may be distinct sub-species (Perrin et al. 1987, 1994; Rice 1998; Viricel and Rosel 2014): the large, heavily spotted form which inhabits the continental shelf and is usually found inside or near the 200m isobath; and the smaller, less spotted island and offshore form which occurs in the Atlantic Ocean but is not known to occur in the Gulf of Mexico (Fulling et al. 2003; Mullin and Fulling 2003; Mullin and Fulling 2004; Viricel and Rosel 2014). Where they co-occur, the offshore form of the Atlantic spotted dolphin and the pantropical spotted dolphin can be difficult to differentiate at sea. The pantropical spotted dolphin is distributed worldwide in tropical and some subtropical oceans (Perrin et al. 1987; Perrin and Hohn 1994). In the northern Gulf of Mexico (i.e., U.S. Gulf of Mexico), sightings of this species are common during visual surveys in oceanic waters >200 m and in all seasons of the northern Gulf of Mexico (i.e., U.S. Gulf of Mexico) (Figure 1; Hansen et al. 1996; Mullin and Hoggard 2000; Mullin and Fulling 2004; Maze-Foley and Mullin 2006; Garrison and Aichinger Dias 2020). Pantropical spotted dolphins were seen in all seasons during GulfCet aerial surveys of the northern Gulf of Mexico between 1992 and 1998 (Hansen et al. 1996; Mullin and Hoggard 2000). All the cetacean species found in the oceanic northern Gulf of Mexico almost certainly occur in similar habitat beyond U.S. boundaries in the southern Gulf. There are fewer cetacean sighting and stranding records in the southern Gulf due to more limited effort. Nevertheless there are records for most oceanic species in the southern Gulf (e.g., Jefferson and Schiro 1997; Ortega Ortiz 2002; Ortega-Arangueta et al. 2005; Jefferson et al. 2008; Vázquez Castán et al. 2009; Whitt et al. 2011). Because there are many confirmed records from Gulf of Mexico waters beyond U.S. boundaries (e.g., Jefferson and Schiro 1997, Ortega Ortiz 2002), pantropical spotted dolphins almost certainly occur throughout the oceanic Gulf of Mexico (Jefferson et al. 2008). This is therefore likely a transboundary stock with Cuba and/or Mexico. Because U.S. waters only comprise about 40% of the entire Gulf of Mexico and 35% of the oceanic (i.e., >200 m) Gulf of Mexico (Mullin and Fulling 2004),
abundance and stock boundaries of oceanic species are poorly known, which is also composed of waters belonging to Mexico and Cuba where there is currently little information on cetacean species abundance and distribution. U.S. waters comprise about 40% of the entire Gulf of Mexico and 35% of the oceanic (i.e., >200 m) Gulf of Mexico.

Some of the Pacific Ocean populations have been divided into different geographic stocks based on morphological characteristics (Perrin et al. 1987; Perrin and Hohn 1994). Pantropical spotted dolphins in the northern Gulf of Mexico are managed separately from those in the western North Atlantic. Although there have been no directed studies of the degree of demographic independence between the two areas, such separation is consistent with evidence for population structure in other areas, including more pelagic waters of the eastern tropical Pacific (Leslie and Morin 2016), and is further supported because the two stocks occupy distinct marine ecoregions (Spalding et al. 2007; Moore and Merrick 2011). There are insufficient data to determine whether the northern Gulf of Mexico stock comprises multiple demographically independent populations. Additional morphological, acoustic, genetic, and/or behavioral data are needed to further delineate population structure within the Gulf of Mexico and across the broader geographic area.

The Gulf of Mexico population is being considered a separate stock for management purposes, although there is currently no information to differentiate this stock from the Atlantic Ocean stock(s). Additional morphological, genetic and/or behavioral data are needed to provide further information on stock delineation.

**Figure 1.** Distribution of pantropical spotted dolphin sightings from SEFSC vessel surveys during summer 2003 and spring 2004, and during summer 2009. All on-effort sightings are shown, though not all were used to estimate abundance. Solid lines indicate the 20 m and 200 m isobaths and the offshore extent of the U.S. EEZ.

**POPULATION SIZE**

The best abundance estimate (Nbest) for the northern Gulf of Mexico pantropical spotted dolphin is 37,195 (CV=0.24; Table 1). This estimate is from summer 2017 and summer/fall 2018 oceanic surveys covering waters from the 200-m isobath to the seaward extent of the U.S. EEZ (Garrison et al. 2020). The best abundance estimate available for northern Gulf of Mexico pantropical spotted dolphins is 50,880 (CV=0.27; Table 1). This estimate is from a summer 2009 oceanic survey covering waters from the 200m isobath to the seaward extent of the U.S. EEZ from Texas to Florida.

**Earlier abundance estimates**

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

**Recent surveys and abundance estimates**

An abundance estimate for pantropical spotted dolphins was generated from vessel surveys conducted in the northern Gulf of Mexico from the continental shelf edge (~200-m isobath) to the seaward extent of the U.S. EEZ (Table 1; Garrison et al. 2020). One survey was conducted from 2 July to 25 August 2017 and consisted of 7,302 km of on-effort trackline, and the second survey was conducted from 11 August to 6 October 2018 and consisted of 6,473 km of on-effort trackline within the surveyed strata. Both surveys used a double-platform data-collection procedure, which allowed estimation of the detection probability on the trackline using the independent observer approach assuming point independence (Laake and Borchers 2004). Abundance was calculated using mark-recapture distance sampling implemented in package mrds (version 2.21, Laake et al. 2020) in the R statistical programming language. This approach accounted for the effects of covariates (e.g., sea state, glare) on detection probability within the surveyed area. The surveys were conducted in "passing mode" (e.g., Schwarz et al. 2010) while all prior surveys in the Gulf of Mexico have been conducted in "closing mode." The abundance estimate for this stock included sightings of unidentified dolphins that were apportioned among identified species based on their relative density within the survey strata (Garrison et al. 2020). The inverse variance weighted mean abundance estimate for pantropical spotted dolphins in oceanic waters during 2017 and 2018 was 37,195 (CV=0.24; Table 1; Garrison et al. 2020). Unlike previous abundance estimates, this estimate was corrected for the probability of detection on the trackline. During summer 2009, a vessel-based line-transect survey dedicated to estimating the abundance of oceanic cetaceans was conducted in the northern Gulf of Mexico. Survey lines were stratified in relation to depth and the location of the Loop Current. The abundance estimate for pantropical spotted dolphins in oceanic waters during 2009 was 50,880 (CV=0.27; Table 1).

**Table 1.** Most recent abundance estimate (Nbest) and coefficient of variation (CV) of northern Gulf of Mexico pantropical spotted dolphins in oceanic waters (200 m to the offshore extent of the EEZ) based on an average from summer 2017 and summer/fall 2018 vessel surveys.
Table 1. Summary of abundance estimates for northern Gulf of Mexico pantropical spotted dolphins. Month, year and area covered during each abundance survey, and resulting abundance estimate (Nbest) and coefficient of variation (CV).

<table>
<thead>
<tr>
<th>Month/Year</th>
<th>Area</th>
<th>Nbest</th>
<th>CV</th>
</tr>
</thead>
<tbody>
<tr>
<td>Apr-Jun 1991-1994</td>
<td>Oceanic waters</td>
<td>31,320</td>
<td>0.20</td>
</tr>
<tr>
<td>Apr-Jun 1996-2001 (excluding 1998)</td>
<td>Oceanic waters</td>
<td>91,321</td>
<td>0.16</td>
</tr>
<tr>
<td>Jun-Aug 2003, Apr-Jun 2004 (pooled)</td>
<td>Oceanic waters</td>
<td>34,067</td>
<td>0.18</td>
</tr>
<tr>
<td>Jun-Aug 2009</td>
<td>Oceanic waters</td>
<td>50,880</td>
<td>0.27</td>
</tr>
</tbody>
</table>

Minimum Population Estimate

The minimum population estimate (Nmin) is the lower limit of the two-tailed 60% confidence interval of the log-normal distributed abundance estimate. This is equivalent to the 20th percentile of the log-normal distributed abundance estimate as specified by Wade and Angliss (1997). The best estimate of abundance for pantropical spotted dolphins is 37,19550,880 (CV=0.2427). The minimum population estimate for the northern Gulf of Mexico pantropical spotted dolphin is 30,377 (Table 2)40,699 pantropical spotted dolphins.

Current Population Trend

The statistical power to detect a trend in abundance for this stock is poor due to the relatively imprecise abundance estimates and long intervals between surveys. For example, the power to detect a precipitous decline in abundance (i.e., 50% decrease in 15 years) with estimates of low precision (e.g., CV>0.30) remains below 80% (alpha = 0.30) unless surveys are conducted on an annual basis (Taylor et al. 2007). In addition, because these surveys are restricted to U.S. waters, it is not possible to distinguish between changes in population size and Gulf-wide shifts in spatial distribution. Five point estimates of pantropical spotted dolphin abundance have been made based on data from surveys during 2003 (June–August), 2004 (April–June), 2009 (July–August), 2017 (July–August), and 2018 (August–October). Each of these surveys had a similar design and was conducted using the same vessel or a vessel with a similar observation platform. Prior year surveys only employed a single survey team. The estimates for 2017 and 2018 were conducted in "passing" mode which resulted in increased numbers of unidentified sightings and may have affected group size estimation for distant groups of dolphins and small whales. Apportioning unidentified species among identified taxa addresses the first issue, and there were no apparent relationships between sighting distance and estimated group size. The model for detection probability on the trackline from the summer 2017 survey was applied to the abundance estimates from the 2003, 2004, and 2009 surveys. The resulting abundance estimates were: 2003, N=72,901 (CV=0.20); 2004, N=78,879 (CV=0.41); 2009, N=84,047 (CV=0.36); 2017, N=27,362 (CV=0.27); and 2018, N=58,726 (CV=0.41). Pairwise comparisons of the log-transformed means were conducted between years, and significant differences were assessed at alpha=0.10. P-values were adjusted for multiple comparisons. There were significant differences between the 2003 and 2017 estimates (p.adjusted = 0.016) and the 2009 and 2017 estimates.
suggesting a possible decline in abundance during recent years. A trend analysis has not been conducted for this stock. Four point estimates of pantropical spotted dolphin abundance have been made based on data from surveys covering 1991-2009 (Table 1). The estimates vary by a maximum factor of nearly three. To determine whether changes in abundance have occurred over this period, an analysis of all the survey data needs to be conducted which incorporates covariates (e.g., survey conditions, season) that could potentially affect estimates. It should be noted that since this is a transboundary stock and the abundance estimates are for U.S. waters only, it will be difficult to interpret any detected trends.

CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

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

POTENTIAL BIOLOGICAL REMOVAL

Potential Biological Removal (PBR) is the product of the minimum population size, one half the maximum net productivity rate, and a recovery factor (MPA Sec. 3.16 U.S.C. 1362; Wade and Angliss 1997). The minimum population size is 30,377. The maximum productivity rate is 0.04, the default value for cetaceans. The recovery factor is 0.5 because the stock is of unknown status. PBR for the northern Gulf of Mexico pantropical spotted dolphin stock is 304 (Table 2).

<table>
<thead>
<tr>
<th>Nbest</th>
<th>CV</th>
<th>Nmin</th>
<th>Fr</th>
<th>Rmax</th>
<th>PBR</th>
</tr>
</thead>
<tbody>
<tr>
<td>37,195</td>
<td>0.24</td>
<td>30,377</td>
<td>0.5</td>
<td>0.04</td>
<td>304</td>
</tr>
</tbody>
</table>

ANNUAL HUMAN- CAUSED MORTALITY AND SERIOUS INJURY

Total annual estimated fishery-related mortality and serious injury to this stock during 2014–2018 was presumed to be zero, as there were no reports of mortalities or serious injuries to pantropical spotted dolphins in the Gulf of Mexico (Table 3). Mean annual mortality and serious injury during 2014–2018 due to other human-caused actions (the Deepwater Horizon oil spill) was predicted to be 241. The minimum total mean annual human-caused mortality and serious injury for this stock during 2014–2018 was, therefore, 241. The estimated mean annual fishery-related mortality and serious injury for this stock during 2009–2013 was 3.8 pantropical spotted dolphins (CV=0.59; Table 2) due to interactions with the pelagic longline fishery. Additional mean annual mortality and serious injury due to other human-caused actions (fishery research) was 0.6. The total mean annual human-caused mortality and serious injury for this stock during 2009–2013 was 4.4.

Table 3. Total annual estimated fishery-related mortality and serious injury for northern Gulf of Mexico pantropical spotted dolphins.

<table>
<thead>
<tr>
<th>Years</th>
<th>Source</th>
<th>Annual Avg.</th>
<th>CV</th>
</tr>
</thead>
<tbody>
<tr>
<td>2014–2018</td>
<td>U.S. fisheries using observer data</td>
<td>0</td>
<td>-</td>
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</tbody>
</table>

Fisheries Information

There are two commercial fisheries that interact, or that potentially could interact, with this stock in the Gulf of Mexico. These are the Category I Atlantic Highly Migratory Species (high seas longline) fishery and the Category I Atlantic Ocean, Caribbean, Gulf of Mexico large pelagics longline fishery (Appendix III). Percent observer coverage (percentage of sets observed) for these longline fisheries for each year during 2014–2018 was 18, 19, 23, 13 and 20, respectively. There is very little effort within the Gulf of Mexico by the high seas longline fishery, and no takes of pantropical spotted dolphins within high seas waters of the Gulf of Mexico have been observed or reported thus far. Pelagic swordfish, tunas and billfish are the targets of the large pelagic longline fishery operating in the northern Gulf.
of Mexico. During 2014–2018 there were no observed mortalities or serious injuries to pantropical spotted dolphins by this fishery (Garrison and Stokes 2016; 2017; 2019; 2020; in review). The average annual serious injury and mortality in the Gulf of Mexico pelagic longline fishery for the 5-year period from 2009 to 2013 is 3.8 (CV=0.59; Table 2). There were no reports of mortality or serious injury to pantropical spotted dolphins by this fishery during 1998–2008 (Yeung 1999; Yeung 2001; Garrison 2003; Garrison and Richards 2004; Garrison 2005; Fairfield-Walsh and Garrison 2006; Fairfield-Walsh and Garrison 2007; Fairfield and Garrison 2008; Garrison et al. 2009). However, during 2009, 4 pantropical spotted dolphins were observed to be seriously injured (3 during quarter 2 and 1 during quarter 4) and 1 pantropical spotted dolphin was released alive with no presumed serious injury after entanglement interactions with the pelagic longline fishery (Garrison and Stokes 2010). During 2010, 2 pantropical spotted dolphins were released alive with no presumed serious injuries after entanglement interactions with the pelagic longline fishery (Garrison and Stokes 2012a). One of the entanglements occurred during experimental fishing to test the effectiveness of “weak” hooks as a potential bycatch mitigation tool. There was 100% observer coverage of all experimental sets. During 2011 there were no reports of mortality or serious injury to pantropical spotted dolphins (Garrison and Stokes 2012b). During 2012, 1 mortality of a pantropical spotted dolphin occurred during an experimental set (during quarter 2; Garrison and Stokes 2013). During 2013, 1 pantropical spotted dolphin was observed to be seriously injured (during quarter 2), and 2 additional dolphins were released alive with no presumed serious injuries (Garrison and Stokes 2014). During the second quarters (15 April – 15 June) of 2009–2013, observer coverage in the Gulf of Mexico pelagic longline fishery was greatly enhanced (approaching 55%) to collect more robust information on the interactions between pelagic longline vessels and spawning bluefin tuna. Therefore, the high annual observer coverage rates during 2009–2013 (Table 2) primarily reflect high coverage rates during the second quarter of each year. During the second quarter, this elevated coverage results in an increased probability that relatively rare interactions will be detected. Species within the oceanic Gulf of Mexico are presumed to be resident year-round; however, it is unknown if the bycatch rate observed during the second quarter is representative of that which occurs throughout the year.

Table 2. Summary of the incidental mortality and serious injury of northern Gulf of Mexico pantropical spotted dolphins in the pelagic longline commercial fishery including the years sampled (Years), the number of vessels active within the fishery (Vessels), the type of data used (Data Type), the annual observer coverage (Observer Coverage), the observed mortalities and serious injuries recorded by on-board observers, the estimated annual mortality and serious injury, the combined annual estimates of mortality and serious injury (Estimated Combined Mortality), the estimated CV of the combined estimates (Estimated CVs) and the mean of the combined estimates (CV in parentheses).

<table>
<thead>
<tr>
<th>Fishery</th>
<th>Years</th>
<th>Vessels</th>
<th>Data Type</th>
<th>Observer Coverage</th>
<th>Observed Serious Injury</th>
<th>Estimated Serious Injury</th>
<th>Estimated Mortality</th>
<th>Estimated Combined Mortality</th>
<th>Est. CVs</th>
<th>Mean Annual Mortality</th>
</tr>
</thead>
<tbody>
<tr>
<td>Pelagic Longline</td>
<td>09–13</td>
<td>47, 46, 42, 47</td>
<td>Obs. Data Logbook</td>
<td>22.2, 23.1, 18.1, 22.5</td>
<td>0, 0, 0, 0</td>
<td>16.0, 0, 0, 2.4</td>
<td>0, 0, 0, 0</td>
<td>16.0, 0, 0, 2.4</td>
<td>0.69</td>
<td>3.8 (0.59)</td>
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</table>

During 2014–2018 there were no observed mortalities or serious injuries to pantropical spotted dolphins by this fishery (Garrison and Stokes 2016; 2017; 2019; 2020; in review). The average annual serious injury and mortality in the Gulf of Mexico pelagic longline fishery for the 5-year period from 2009 to 2013 is 3.8 (CV=0.59; Table 2). There were no reports of mortality or serious injury to pantropical spotted dolphins by this fishery during 1998–2008 (Yeung 1999; Yeung 2001; Garrison 2003; Garrison and Richards 2004; Garrison 2005; Fairfield-Walsh and Garrison 2006; Fairfield-Walsh and Garrison 2007; Fairfield and Garrison 2008; Garrison et al. 2009). However, during 2009, 4 pantropical spotted dolphins were observed to be seriously injured (3 during quarter 2 and 1 during quarter 4) and 1 pantropical spotted dolphin was released alive with no presumed serious injury after entanglement interactions with the pelagic longline fishery (Garrison and Stokes 2010). During 2010, 2 pantropical spotted dolphins were released alive with no presumed serious injuries after entanglement interactions with the pelagic longline fishery (Garrison and Stokes 2012a). One of the entanglements occurred during experimental fishing to test the effectiveness of “weak” hooks as a potential bycatch mitigation tool. There was 100% observer coverage of all experimental sets. During 2011 there were no reports of mortality or serious injury to pantropical spotted dolphins (Garrison and Stokes 2012b). During 2012, 1 mortality of a pantropical spotted dolphin occurred during an experimental set (during quarter 2; Garrison and Stokes 2013). During 2013, 1 pantropical spotted dolphin was observed to be seriously injured (during quarter 2), and 2 additional dolphins were released alive with no presumed serious injuries (Garrison and Stokes 2014). During the second quarters (15 April – 15 June) of 2009–2013, observer coverage in the Gulf of Mexico pelagic longline fishery was greatly enhanced (approaching 55%) to collect more robust information on the interactions between pelagic longline vessels and spawning bluefin tuna. Therefore, the high annual observer coverage rates during 2009–2013 (Table 2) primarily reflect high coverage rates during the second quarter of each year. During the second quarter, this elevated coverage results in an increased probability that relatively rare interactions will be detected. Species within the oceanic Gulf of Mexico are presumed to be resident year-round; however, it is unknown if the bycatch rate observed during the second quarter is representative of that which occurs throughout the year.
Other Mortality

Three research-related mortalities were documented during 2009–2013. In 2011, 3 pantropical spotted dolphins were incidentally captured and killed during a research mid-water trawl. These mortalities were included in the stranding database and in Table 3.

Seven Five pantropical spotted dolphins were reported stranded in the Gulf of Mexico during 2009–2014–2018–2013 (Table 3; NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 21 May 2019–1 June 2014). Three strandings were documented in 2014, all in Florida, and two strandings were documented in 2018, one each in Alabama and in Texas. Evidence of human interaction was detected for 3 strandings (mortalities), which were the result of incidental capture in a research trawling net. No evidence of human interaction was detected for 2 one stranded animals, and for the remaining 2 four animals, it could not be determined if there was evidence of human interaction. Stranding data probably underestimate the extent of human and fishery-related mortality and serious injury because not all of the dolphins that die or are seriously injured in human interactions wash ashore, or, if they do, they are not all recovered (Peltier et al. 2012; Wells et al. 2015). In particular, oceanic stocks in the Gulf of Mexico are less likely to strand than nearshore coastal stocks or shelf stocks (Williams et al. 2011). Additionally, not all carcasses will show evidence of human interaction, entanglement or other fishery-related interaction due to decomposition, scavenger damage, etc. (Byrd et al. 2014). Finally, the level of technical expertise among stranding network personnel varies widely as does the ability to recognize signs of human interaction.

An Unusual Mortality Event (UME) was declared for cetaceans in the northern Gulf of Mexico beginning 1 February–March 2010 and ending 31 July 2014 (Litz et al. 2014; https://www.fisheries.noaa.gov/national/marine-life-distress/2010-2014-cetacean-unusual-mortality-event-northern-gulf-mexico), and as of September 2014, the event is still ongoing (Litz et al. 2014). It included cetaceans that stranded prior to the Deepwater Horizon (DWH) oil spill (see “Habitat Issues” below), during the spill, and after. Exposure to the DWH oil spill was determined to be the primary underlying cause of the elevated stranding numbers in the northern Gulf of Mexico after the spill (e.g., Schwacke et al. 2014; Venn-Watson et al. 2015; Colegrove et al. 2016; DWH NRDAT 2016; see Habitat Issues section). Three pantropical spotted dolphin strandings during 2011 were considered to be part of this UME. During 2010, no animals from this stock were considered to be part of the UME, but the 3 strandings during 2011 were included in the UME.

A population model was developed to estimate the injury and time to recovery for stocks affected by the DWH oil spill, taking into account long-term effects resulting from mortality, reproductive failure, reduced survival rates, and the proportion of the stock exposed to DWH oil (DWH MMIQT 2015). Overall, the model estimated that this stock experienced a 9% maximum reduction in population size due to the oil spill (DWH MMIQT 2015). The mortality projected for the years 2010–2013 due to the spill has not been reported previously. Based on the population model, it was projected that 2,367 pantropical spotted dolphins died during 2010–2013 (four year annual average of 592) due to elevated mortality associated with oil exposure (see Appendix VI). For the 2014–2018 reporting period of this SAR, the population model estimated 1,203 pantropical spotted dolphins died due to elevated mortality associated with oil exposure. The population model used to predict pantropical spotted dolphin mortality due to the DWH event has a number of sources of uncertainty. Model parameters (e.g., survival rates, reproductive rates, and life-history parameters) were derived from literature sources for pantropical spotted dolphins occupying waters outside of the Gulf of Mexico. In addition, proxy values for the effects of DWH oil exposure on both survival rates and reproductive
success were applied based upon estimated values for common bottlenose dolphins in Barataria Bay. Finally, there was no estimation of uncertainty in model parameters or outputs.

Table 3. Pantropical spotted dolphin (*Stenella attenuata*) strandings along the northern Gulf of Mexico coast, 2009–2013.

<table>
<thead>
<tr>
<th>STATE</th>
<th>2009</th>
<th>2010</th>
<th>2011</th>
<th>2012</th>
<th>2013</th>
<th>TOTAL</th>
</tr>
</thead>
<tbody>
<tr>
<td>Alabama</td>
<td>1</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>1</td>
</tr>
<tr>
<td>Florida</td>
<td>1</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>1</td>
</tr>
<tr>
<td>Louisiana</td>
<td>0</td>
<td>0</td>
<td>3^a</td>
<td>0</td>
<td>0</td>
<td>3</td>
</tr>
<tr>
<td>Mississippi</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
</tr>
<tr>
<td>Texas</td>
<td>1</td>
<td>1</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>2</td>
</tr>
<tr>
<td>TOTAL</td>
<td>3</td>
<td>1</td>
<td>3</td>
<td>0</td>
<td>0</td>
<td>7</td>
</tr>
</tbody>
</table>

^a These 3 strandings were incidental takes during a research trawl. They are included in the Northern Gulf of Mexico UME.

HABITAT ISSUES

The Deepwater Horizon (DWH) MC252 drilling platform, located approximately 50 miles/80 km southeast of the Mississippi River Delta in waters about 1,500 m deep, exploded on 20 April 2010. The rig sank, and over 87 days up to ~3.248 million barrels of oil were discharged from the wellhead until it was capped on 15 July 2010 (*DWH NRDAT 2016; McNutt et al. 2012*). During the response effort dispersants were applied extensively at the seafloor and at the sea surface (*Lehr et al. 2010; OSAT 2010*). In-situ burning, or controlled burning of oil at the surface, was also used extensively as a response tool (*Lehr et al. 2010*). The oil, dispersant and burn residue compounds present ecological concerns (*Buist et al. 1999; NOAA 2011*). The magnitude of this oil spill was unprecedented in U.S. history, causing impacts to wildlife, natural habitats and human communities along coastal areas from western Louisiana to the Florida Panhandle (*NOAA 2011*). It could be years before the entire scope of damage is ascertained (*NOAA 2011*).

Shortly after the oil spill, the Natural Resource Damage Assessment (NRDA) process was initiated under the Oil Pollution Act of 1990. A variety of NRDA research studies were conducted to determine potential impacts of the spill on marine mammals. These studies estimated that 20% (95% CI: 15–26) of pantropical spotted dolphins in the Gulf were exposed to oil, that 9% (95% CI: 4–13) of females suffered from reproductive failure, and 7% (95% CI: 3–11) of pantropical spotted dolphins suffered adverse health effects (*DWH MMIQT 2015*). A population model estimated the stock experienced a maximum 9% reduction in population size (see Other Mortality section above). Anthropogenic sound in the world’s oceans has been shown to affect marine mammals, with vessel traffic, seismic surveys, and active naval sonars being the main anthropogenic contributors to low- and mid-frequency noise in oceanic waters (*e.g., Nowacek et al. 2015; Gomez et al. 2016; NMFS 2018*). The long-term and population consequences of...
these impacts are less well-documented and likely vary by species and other factors. Impacts on marine mammal prey from sound are also possible (Carroll et al. 2017), but the duration and severity of any such prey effects on marine mammals are unknown. These studies have focused on identifying the type, magnitude, severity, length and impact of oil exposure to oceanic, continental shelf, coastal and estuarine marine mammals. For continental shelf and oceanic cetaceans, the NOAA-led efforts include: aerial surveys to document the distribution, abundance, species and exposure relative to oil from the DWH spill; and ship surveys to evaluate exposure to oil and other chemicals and to assess changes in animal behavior and distribution relative to oil exposure through visual and acoustic surveys, deployment of passive acoustic monitoring systems, collection of tissue samples, and deployment of satellite tags on sperm and Bryde’s whales.

Vessel and aerial surveys documented common bottlenose dolphins, Atlantic spotted dolphins, rough-toothed dolphins, spinner dolphins, pantropical spotted dolphins, Risso’s dolphins, striped dolphins, sperm whales, dwarf/pygmy sperm whales and a Cuvier’s beaked whale swimming in oil or potentially oil-derived substances (e.g., sheen, mousse) in offshore waters of the northern Gulf of Mexico following the DWH oil spill. The effects of oil exposure on marine mammals depend on a number of factors including the type and mixture of chemicals involved, the amount, frequency and duration of exposure, the route of exposure (inhaled, ingested, absorbed, or external) and biomedical risk factors of the particular animal (Geraci 1990). In general, direct external contact with petroleum compounds or dispersants with skin may cause skin irritation, chemical burns and infections. Inhalation of volatile petroleum compounds or dispersants may irritate or injure the respiratory tract, which could lead to pneumonia or inflammation. Ingestion of petroleum compounds may cause injury to the gastrointestinal tract, which could affect an animal’s ability to digest or absorb food. Absorption of petroleum compounds or dispersants may damage kidney, liver and brain function in addition to causing immune suppression and anemia. Long term chronic effects such as lowered reproductive success and decreased survival may occur (Geraci 1990).

STATUS OF STOCK

Pantropical spotted dolphins are not listed as threatened or endangered under the Endangered Species Act, and the northern Gulf of Mexico stock is not considered strategic under the MOPA. No fishery-related mortality or serious injury has been observed in recent years; therefore, total fishery-related mortality and serious injury Total fishery-related mortality and serious injury for this stock is less than 10% of PBR and can be considered to be insignificant and approaching zero mortality and serious injury rate. The status of pantropical spotted dolphins in the northern Gulf of Mexico, relative to OSP, is unknown. There were statistically significant decreases in population size in recent years for this stock in the northern Gulf of Mexico. There are insufficient data to determine the population trends for this stock.

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DWH MMIQT. 2015. Models and analyses for the quantification of injury to Gulf of Mexico cetaceans from the Deepwater Horizon Oil Spill. MM TR.01 Schwacke Quantification of Injury to GOM.Cetaceans. Southeast Fisheries Science Center, Protected Resources and Biodiversity Division, 75 Virginia Beach Dr., Miami, Florida 33140. PRBD Contribution #: PRBD-2020-02. Available from:


Garrison, L.P. and L. Stokes. in review. Estimated bycatch of marine mammals and sea turtles in the U.S. Atlantic pelagic longline fleet during 2018. Southeast Fisheries Science Center, Protected Resources and Biodiversity Division, 75 Virginia Beach Dr., Miami, Florida 33140. PRD Contribution # PRD-2020-08. 55 pp. Available from: https://repository.library.noaa.gov/view/noaa/26505


STRIPED DOLPHIN (Stenella coeruleoalba):
Northern Gulf of Mexico Stock

STOCK DEFINITION AND GEOGRAPHIC RANGE

The striped dolphin is distributed worldwide in tropical to temperate oceanic waters (Leatherwood and Reeves 1983; Perrin et al. 1994). In the northern Gulf of Mexico (i.e., U.S. Gulf of Mexico), sightings of this species occur in waters >200 m deep, with most observations in waters ≥1,000 m deep (Figure 1; Mullin and Fulling 2004; Maze-Foley and Mullin 2006; Garrison and Aichinger Dias 2020). Like spinner dolphins, the majority of sightings are east of the Mississippi River, but striped dolphins are also observed over the continental slope in the western Gulf and out in the deeper central basin. Striped dolphins have been seen in all seasons during NMFS visual surveys (Mullin and Hoggard 2000). All the cetacean species found in the oceanic northern Gulf of Mexico almost certainly occur in similar habitat beyond U.S. boundaries in the southern Gulf. There are fewer cetacean sighting and stranding records in the southern Gulf due to more limited effort. Nevertheless there are records for most oceanic species in the southern Gulf (e.g., Jefferson and Schiro 1997; Ortega Ortiz 2002; Ortega-Argueta et al. 2005; Jefferson et al. 2008; Vázquez Castán et al. 2009; Whitt et al. 2011). Although there are only a few records from Gulf of Mexico waters beyond U.S. boundaries (e.g., Jefferson and Schiro 1997, Ortega Ortiz 2002), striped dolphins almost certainly occur throughout the oceanic Gulf of Mexico (Jefferson et al. 2008). This is therefore likely a transboundary stock with Cuba and/or Mexico. Because U.S. waters only comprise about 40% of the entire Gulf of Mexico and 35% of the oceanic (i.e., >200 m) Gulf of Mexico (Mullin and Fulling 2004), abundance and stock boundaries of oceanic species are poorly known, which is also composed of waters belonging to Mexico and Cuba where there is currently little information on cetacean species abundance and distribution. U.S. waters only comprise about 40% of the entire Gulf of Mexico and 65% of oceanic waters are south of the U.S. Exclusive Economic Zone (EEZ).

Striped dolphins in the northern Gulf of Mexico are managed separately from those in the western North Atlantic. Although there have been no directed studies of the degree of demographic independence between the two areas, this management structure is consistent with the fact that the Gulf of Mexico and western North Atlantic belong to distinct marine ecoregions (Spalding et al. 2007; Moore and Merrick 2011). There are insufficient data to determine whether
The northern Gulf of Mexico stock comprises multiple demographically independent populations. Additional morphological, acoustic, genetic, and/or behavioral data are needed to further delineate population structure within the Gulf of Mexico and across the broader geographic area. The Gulf of Mexico population is provisionally being considered a separate stock for management purposes, although there is currently no information to differentiate this stock from the Atlantic Ocean stock(s). Additional morphological, genetic and/or behavioral data are needed to provide further information on stock delineation.

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

POPULATION SIZE

The best abundance estimate (N_best) for the northern Gulf of Mexico striped dolphin is 1,817 (CV=0.56; Table 1). This estimate is from summer 2017 and summer/fall 2018 oceanic surveys covering waters from the 200-m isobath to the seaward extent of the U.S. EEZ (Garrison et al. 2020). The best abundance estimate available for northern Gulf of Mexico striped dolphins is 1,849 (CV=0.77; Table 1). This estimate is from a summer 2009 oceanic survey covering waters from the 200-m isobath to the seaward extent of the U.S. EEZ.

Earlier abundance estimates

Please see Appendix IV for a summary of abundance estimates, including earlier estimates and survey descriptions. All estimates of abundance were derived through the application of distance sampling analysis (Buckland et al. 2001) and the computer program DISTANCE (Thomas et al. 1998) to line transect survey data collected from ships in the oceanic northern Gulf of Mexico (i.e., 200m isobath to seaward extent of the U.S. EEZ) and are summarized in Appendix IV.

From 1991 through 1994, and from 1996 through 2001 (excluding 1998), annual surveys were conducted during spring along a fixed plankton-sampling trackline. Due to limited survey effort in any given year, the survey effort-weighted estimated average abundance of striped dolphins for all surveys combined was estimated. For 1991 to 1994, the estimate was 4,858 (CV=0.44) (Hansen et al. 1995), and for 1996 to 2001, 6,505 (CV=0.43) (Mullin and Fulling 2004; Table 1).

During summer 2003 and spring 2004, surveys dedicated to estimating cetacean abundance were conducted along a grid of uniformly spaced transect lines from a random start. The abundance estimate for striped dolphins, pooled from 2003 to 2004, was 3,325 (CV=0.48) (Mullin 2007; Table 1).

Recent surveys and abundance estimates

An abundance estimate for striped dolphins was generated from vessel surveys conducted in the northern Gulf of Mexico from the continental shelf edge (~200-m isobath) to the seaward extent of the U.S. EEZ (Table 1; Garrison et al. 2020). One survey was conducted from 2 July to 25 August 2017 and consisted of 7,302 km of on-effort trackline, and the second survey was conducted from 11 August to 6 October 2018 and consisted of 6,473 km of on-effort trackline within the surveyed strata. Both surveys used a double-platform data-collection procedure, which allowed estimation of the detection probability on the trackline using the independent observer approach assuming point independence (Laake and Borchers 2004). Abundance was calculated using mark-recapture distance sampling implemented in package mrds (version 2.21, Laake et al. 2020) in the R statistical programming language. This approach accounted for the effects of covariates (e.g., sea state, glare) on detection probability within the surveyed strip. The surveys were conducted in "passing mode" (e.g., Schwarz et al. 2010), while all prior surveys in the Gulf of Mexico have been conducted in "closing mode." The abundance estimate for this stock included sightings of unidentified dolphins that were apportioned among identified species based on their relative density within the survey strata (Garrison et al. 2020). There were sightings of striped dolphins in 2018 but there were none in 2017. The inverse variance weighted mean abundance estimate for striped dolphins in oceanic waters during 2017 and 2018 is 1,817 (CV=0.56; Table 1; Garrison et al. 2020). Unlike previous abundance estimates, this estimate was corrected for the probability of detection on the trackline.
During the summer of 2009, a line transect survey dedicated to estimating the abundance of oceanic cetaceans was conducted in the northern Gulf of Mexico.Survey lines were stratified in relation to depth and the location of the Loop Current. The abundance estimate for striped dolphins in oceanic waters during 2009 was 1,849 (CV=0.77; Table 1).

Table 1. Most recent abundance estimate (Nbest) and coefficient of variation (CV) of northern Gulf of Mexico striped dolphins in oceanic waters (200 m to the offshore extent of the EEZ) based on an average from summer 2017 and summer/fall 2018 vessel surveys.

<table>
<thead>
<tr>
<th>Years</th>
<th>Area</th>
<th>Nbest</th>
<th>CV</th>
</tr>
</thead>
<tbody>
<tr>
<td>2017, 2018</td>
<td>Gulf of Mexico</td>
<td>1,817</td>
<td>0.56</td>
</tr>
</tbody>
</table>

Table 1. Summary of abundance estimates for northern Gulf of Mexico striped dolphins. Month, year and area covered during each abundance survey, and resulting abundance estimate (Nbest) and coefficient of variation (CV).

<table>
<thead>
<tr>
<th>Month/Year</th>
<th>Area</th>
<th>Nbest</th>
<th>CV</th>
</tr>
</thead>
<tbody>
<tr>
<td>Apr–Jun 1991–1994</td>
<td>Oceanic waters</td>
<td>4,858</td>
<td>0.44</td>
</tr>
<tr>
<td>Apr–Jun 1996–2001</td>
<td>Oceanic waters</td>
<td>6,505</td>
<td>0.43</td>
</tr>
<tr>
<td>(excluding 1998)</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Jun–Aug 2003, Apr–Jun 2004</td>
<td>Oceanic waters</td>
<td>3,325</td>
<td>0.48</td>
</tr>
<tr>
<td>Jun–Aug 2009</td>
<td>Oceanic waters</td>
<td>1,849</td>
<td>0.77</td>
</tr>
</tbody>
</table>

Minimum Population Estimate

The minimum population estimate (Nmin) is the lower limit of the two-tailed 60% confidence interval of the log-normal distributed abundance estimate. This is equivalent to the 20th percentile of the log-normal distributed abundance estimate as specified by Wade and Angliss (1997). The best estimate of abundance for striped dolphins is 1,817–1,849 (CV=0.56–0.77). The minimum population estimate for the northern Gulf of Mexico striped dolphin is 1,172 (Table 2)–1,041 striped dolphins.

Current Population Trend

The statistical power to detect a trend in abundance for this stock is poor due to the relatively imprecise abundance estimates and long intervals between surveys. For example, the power to detect a precipitous decline in abundance (i.e., 50% decrease in 15 years) with estimates of low precision (e.g., CV>0.30) remains below 80% (alpha=0.30) unless surveys are conducted on an annual basis (Taylor et al. 2007). In addition, because these surveys are restricted to U.S. waters, it is not possible to distinguish between changes in population size and Gulf-wide shifts in spatial distribution. Five point estimates of striped dolphin abundance have been made based on data from surveys during: 2003 (June–August), 2004 (April–June), 2009 (July–August), 2017 (July–August), and 2018 (August–October). Each of these surveys had a similar design and was conducted using the same vessel or a vessel with a similar observation platform. Prior year surveys only employed a single survey team. The estimates for 2017 and 2018 were conducted in
"passing" mode which resulted in increased numbers of unidentified sightings and may have affected group size estimation for distant groups of dolphins and small whales. Apportioning unidentified species among identified taxa addresses the first issue, and there were no apparent relationships between sighting distance and estimated group size. The model for detection probability on the trackline from the summer 2017 survey was applied to the abundance estimates from the 2003, 2004, and 2009 surveys. The resulting abundance estimates were: 2003, N=5,494 (CV=0.43); 2004, N=10,764 (CV=0.51); 2009, N=3,060 (CV=0.73); 2017, N=0 (CV=NA); and 2018, N=3,633 (CV=0.56). Pairwise comparisons of the non-zero log-transformed means were conducted between years, and significant differences were assessed at alpha=0.10. P-values were adjusted for multiple comparisons. There were no significant differences between survey years. Four point estimates of striped dolphin abundance have been made based on data from surveys covering 1991-2009. The estimates vary by a maximum factor of more than three. To determine whether changes in abundance have occurred over this period, an analysis of all the survey data needs to be conducted which incorporates covariates (e.g., survey conditions, season) that could potentially affect estimates. Nevertheless, differences in temporal abundance estimates will still be difficult to interpret without a Gulf of Mexico-wide understanding of striped dolphin abundance. The oceanography of the Gulf of Mexico is quite dynamic, and the spatial scale of the Gulf is small relative to the ability of most cetacean species to travel. Studies based on abundance and distribution surveys restricted to U.S. waters are unable to detect temporal shifts in distribution beyond U.S. waters that might account for any changes in abundance.

CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

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

POTENTIAL BIOLOGICAL REMOVAL

Potential Biological Removal (PBR) is the product of the minimum population size, one half the maximum net productivity rate and a recovery factor (MMPA Sec. 3.16 U.S.C. 1362; Wade and Angliss 1997). The minimum population size is 1,172, the maximum productivity rate is 0.04, the default value for cetaceans. The recovery factor, which accounts for endangered, depleted, threatened stocks, or stocks of unknown status relative to optimum sustainable population (OSP), is assumed to be 0.5 because the stock is of unknown status. PBR for the northern Gulf of Mexico striped dolphin is 12 (Table 2).

Table 2. Best and minimum abundance estimates for the northern Gulf of Mexico striped dolphin with Maximum Productivity Rate (Rmax), Recovery Factor (Fr) and PBR.

<table>
<thead>
<tr>
<th>Nbest</th>
<th>CV</th>
<th>Nmin</th>
<th>Fr</th>
<th>Rmax</th>
<th>PBR</th>
</tr>
</thead>
<tbody>
<tr>
<td>1,817</td>
<td>0.56</td>
<td>1,172</td>
<td>0.50</td>
<td>0.04</td>
<td>12</td>
</tr>
</tbody>
</table>

ANNUAL HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

Total annual estimated fishery-related mortality and serious injury to this stock during 2014–2018 was presumed to be zero, as there were no reports of mortalities or serious injuries to striped dolphins in the northern Gulf of Mexico (Table 3). Mean annual mortality and serious injury during 2014–2018 due to other human-caused actions (the Deepwater Horizon oil spill) was predicted to be 13. The minimum total mean annual human-caused mortality and serious injury for this stock during 2014–2018 was, therefore, 13. There has been no reported fishing-related mortality or serious injury of striped dolphins during 1998-2010 (Yeung 1999; Yeung 2001; Garrison 2003; Garrison and Richards 2004; Garrison 2005; Fairfield Walsh and Garrison 2006; Fairfield Walsh and Garrison 2007; Fairfield and Garrison 2008; Garrison et al. 2009; Garrison and Stokes 2010; 2011).

Table 3. Total annual estimated fishery-related mortality and serious injury for the northern Gulf of Mexico striped dolphin.

<table>
<thead>
<tr>
<th>Years</th>
<th>Source</th>
<th>Annual Avg.</th>
<th>CV</th>
</tr>
</thead>
<tbody>
<tr>
<td>2014–2018</td>
<td>U.S. fisheries using observer data</td>
<td>0</td>
<td>-</td>
</tr>
</tbody>
</table>
There are two commercial fisheries that interact, or that potentially could interact, with this stock in the Gulf of Mexico. These are the Category I Atlantic Highly Migratory Species (high seas) longline fishery and the Category I Atlantic Ocean, Caribbean, Gulf of Mexico large pelagic longline fishery (Appendix III). Percent observer coverage (percentage of sets observed) for these longline fisheries for each year during 2014–2018 was 18, 19, 23, 13 and 20, respectively. There is very little effort within the Gulf of Mexico by the Atlantic Highly Migratory Species (high seas) longline fishery, and no takes of striped dolphins within high seas waters of the Gulf of Mexico have been observed or reported thus far. The commercial fishery which potentially could interact with this stock in the Gulf of Mexico is the Atlantic Ocean, Caribbean, Gulf of Mexico large pelagic longline fishery (Appendix III). Pelagic swordfish, tunas and billfish are the targets of the longline fishery operating in the northern Gulf of Mexico. During 2014–2018 there were no observed mortalities or serious injuries to striped dolphins by this fishery (Garrison and Stokes 2016; 2017; 2019; 2020; in review). There were no reports of mortality or serious injury to striped dolphins by this fishery during 1998–2010 (Yeung 1999; Yeung 2001; Garrison 2003; Garrison and Richards 2004; Garrison 2005; Fairfield-Walsh and Garrison 2006; Fairfield-Walsh and Garrison 2007; Fairfield and Garrison 2008; Garrison et al. 2009; Garrison and Stokes 2010; 2011).

Other Mortality

Three There was one reported stranding of a striped dolphin were reported stranded in the Gulf of Mexico during 20062014–20182010 (NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 21 May 201916 November 2011). This animal stranded during 2015 in Florida, and it could not be determined if there was evidence of human interaction. Stranding data probably underestimate the extent of human and fishery-related mortality and serious injury because not all of the dolphins that die or are seriously injured in human interactions wash ashore, or, if they do, they are not all recovered (Peltier et al. 2012; Wells et al. 2015). In particular, oceanic stocks in the Gulf of Mexico are less likely to strand than nearshore coastal stocks or shelf stocks (Williams et al. 2011). Additionally, not all carcasses will show evidence of human interaction, entanglement or other fishery-related interaction due to decomposition, scavenger damage, etc. (Byrd et al. 2014). Finally, the level of technical expertise among stranding network personnel varies widely as does the ability to recognize signs of human interaction. One striped dolphin stranded during 2006 in Florida, 1 stranded during 2007 in Louisiana, and 1 stranded during 2008 in Mississippi. Evidence of human interactions was detected for 1 of the stranded animals, and for the remaining 2, it could not be determined if there was evidence of human interactions. Stranding data probably underestimate the extent of fishery-related mortality and serious injury because not all of the marine mammals which die or are seriously injured in fishery interactions wash ashore, nor all that wash ashore are discovered, reported or investigated, 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 interactions.

An Unusual Mortality Event (UME) was declared for cetaceans in the northern Gulf of Mexico beginning 1 FebruaryMarch 2010 and ending 31 July 2014 (Litz et al. 2014; https://www.fisheries.noaa.gov/national/marine-life-distress/2010-2014-cetacean-unusual-mortality-event-northern-gulf-mexico) and, as of early 2012, the event is still ongoing. It included cetaceans that stranded prior to the Deepwater Horizon (DWH) oil spill (see “Habitat Issues” below), during the spill, and after. Exposure to the DWH oil spill was determined to be the primary underlying cause of the elevated stranding numbers in the northern Gulf of Mexico after the spill (e.g., Schwacke et al. 2014; Venn-Watson et al. 2015; Colegrove et al. 2016; DWH NRDAT 2016; see Habitat Issues section). However, there were no striped dolphin strandings recovered within the spatial and temporal boundaries of this UME. During 2010, no animals from this stock were considered to be part of the UME. A population model was developed to estimate the injury and time to recovery for stocks affected by the DWH oil spill, taking into account long-term effects resulting from mortality, reproductive failure, reduced survival rates, and the proportion of the stock exposed to DWH oil (DWH MMIQT 2015). Overall, the model estimated that this stock experienced a 6% maximum reduction in population size due to the oil spill (DWH MMIQT 2015). The mortality projected for the years 2010–2013 due to the spill has not been reported previously. Based on the population model, it was projected that 124 striped dolphins died during 2010–2013 (four year annual average of 31) due to elevated mortality associated with oil exposure (see Appendix VI). For the 2014–2018 reporting period of this SAR, the population model estimated 63 striped dolphins died due to elevated mortality associated with oil exposure. The population model used to predict striped dolphin mortality due to the DWH event has a number of sources of uncertainty. Model parameters (e.g., survival rates, reproductive rates, and life-history parameters) were derived from...
literature sources for striped dolphins occupying waters outside of the Gulf of Mexico. In addition, proxy values for the effects of DWH oil exposure on both survival rates and reproductive success were applied based upon estimated values for common bottlenose dolphins in Barataria Bay. Finally, there was no estimation of uncertainty in model parameters or outputs.

HABITAT ISSUES

The Deepwater Horizon (DWH) MC252 drilling platform, located approximately 50 miles80 km southeast of the Mississippi River Delta in waters about 1,500 m deep, exploded on 20 April 2010. The rig sank, and over 87 days ~3.2 millions of barrels of oil and gas were discharged from the wellhead until it was capped on 15 July 2010 (DWH NRDAT 2016). During the response effort dispersants were applied extensively at the seafloor and at the sea surface (Lehr et al. 2010; OSAT 2010). In-situ burning, or controlled burning of oil at the surface, was also used extensively as a response tool (Lehr et al. 2010). The oil, dispersant and burn residue compounds present ecological concerns. The magnitude of this oil spill was unprecedented in U.S. history, causing impacts to wildlife, natural habitats and human communities along coastal areas from western Louisiana to the Florida Panhandle (NOAA 2011). It could be years before the entire scope of damage is ascertained (NOAA 2011).

Shortly after the oil spill, the Natural Resource Damage Assessment (NRDA) process was initiated under the Oil Pollution Act of 1990. A variety of NRDA research studies are being conducted to determine potential impacts of the spill on marine mammals. These studies estimated that 13% (95% CI: 8–22) of striped dolphins in the Gulf were exposed to oil, that 6% (95% CI: 3–9) of females suffered from reproductive failure, and 5% (95% CI: 2–8) of striped dolphins suffered adverse health effects (DWH MMIQT 2015). A population model estimated the stock experienced a maximum 6% reduction in population size (see Other Mortality section above).

Anthropogenic sound in the world’s oceans has been shown to affect marine mammals, with vessel traffic, seismic surveys, and active naval sonars being the main anthropogenic contributors to low- and mid-frequency noise in oceanic waters (e.g., Nowacek et al. 2015; Gomez et al. 2016; NMFS 2018). The long-term and population consequences of these impacts are less well-documented and likely vary by species and other factors. Impacts on marine mammal prey from sound are also possible (Carroll et al. 2017), but the duration and severity of any such prey effects on marine mammals are unknown. These studies have focused on identifying the type, magnitude, severity, length and impact of oil exposure to oceanic, coastal and estuarine marine mammals. The research is ongoing and likely will continue for some time. For continental shelf and oceanic cetaceans, the NOAA led efforts include: aerial surveys to document the distribution, abundance, species and exposure of marine mammals and turtles relative to oil from DWH spill; and ship surveys to evaluate exposure to oil and other chemicals and to assess changes in animal behavior and distribution relative to oil exposure through visual and acoustic surveys, deployment of passive acoustic monitoring systems, collection of tissue samples, and deployment of satellite tags on sperm and Bryde's whales.

Aerial surveys have observed Risso’s dolphins, spinner dolphins, pantropical spotted dolphins, striped dolphins, bottlenose dolphins and sperm whales swimming in oil in offshore waters (NOAA 2010a). The effects of oil exposure on marine mammals depend on a number of factors including the type and mixture of chemicals involved, the amount, frequency and duration of exposure, the route of exposure (inhaled, ingested, absorbed, or external) and biomedical risk factors of the particular animal (Geraci 1990; NOAA 2010b). In general, direct external contact with petroleum compounds or dispersants with skin may cause skin irritation, chemical burns and infections. Inhalation of volatile petroleum compounds or dispersants may irritate or injure the respiratory tract, which could lead to pneumonia or inflammation. Ingestion of petroleum compounds may cause injury to the gastrointestinal tract, which could affect an animal’s ability to digest or absorb food. Absorption of petroleum compounds or dispersants may damage kidney, liver and brain function in addition to causing immune suppression and anemia. Long term chronic effects such as lowered reproductive success and decreased survival may occur (Geraci 1990; NOAA 2010b).

STATUS OF STOCK

Striped dolphins are not listed as threatened or endangered under the Endangered Species Act, but the northern Gulf of Mexico stock is considered strategic under the MMPA because the mean modeled annual human-caused mortality and serious injury due to the DWH oil spill exceeds PBR. No fishery-related mortality or serious injury has been observed in recent years; therefore, total fishery-related mortality and serious injury can be considered to be insignificant and approaching zero mortality and serious injury rate. The status of striped dolphins in the northern Gulf of Mexico, relative to OSP, is unknown. There was no statistically significant trend in population size for this stock in the northern Gulf of Mexico. The species is not listed as threatened or endangered under the Endangered Species
Act. There are insufficient data to determine the population trends for this species. Total human-caused mortality and serious injury for this stock is not known. There is insufficient information available to determine whether the total fishery-related mortality and serious injury for this stock is insignificant and approaching zero mortality and serious injury rate. This is not a strategic stock because it is assumed that the average annual human-related mortality and serious injury does not exceed PBR.

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DWH MMIQT. 2015. Models and analyses for the quantification of injury to Gulf of Mexico cetaceans from the Deepwater Horizon Oil Spill. MM TR.01 Schwacke Quantification of Injury to GOM Cetaceans. Southeast Fisheries Science Center, Protected Resources and Biodiversity Division, 75 Virginia Beach Dr., Miami, Florida 33140. PRBD Contribution #: PRBD-2020-02. Available from: https://repository.library.noaa.gov/view/noaa/25568


Garrison, L.P. and L. Stokes, in review. Estimated bycatch of marine mammals and sea turtles in the U.S. Atlantic pelagic longline fleet during 2018. Southeast Fisheries Science Center, Protected Resources and Biodiversity Division, 75 Virginia Beach Dr., Miami, Florida 33140. PRD Contribution # PRD-2020-08. 55 pp. Available from: https://repository.library.noaa.gov/view/noaa/26505


common bottlenose dolphins (*Tursiops truncatus*) found dead following the Deepwater Horizon Oil Spill.


SPINNER DOLPHIN (Stenella longirostris longirostris):
Northern Gulf of Mexico Stock

STOCK DEFINITION AND GEOGRAPHIC RANGE

- The spinner dolphin is distributed worldwide in tropical to temperate oceanic and coastal waters (Leatherwood and Reeves 1983; Perrin and Gilpatrick 1994). In the northern Gulf of Mexico (i.e., U.S. Gulf of Mexico), sightings of this species occur in waters >200 m and are concentrated over the continental slope, particularly east of the Mississippi River (Figure 1; Mullin and Fulling 2004; Maze-Foley and Mullin 2006; Garrison and Aichinger Dias 2020). Spinner dolphins have been seen in all seasons during NMFS visual surveys during GulfCet aerial surveys of the northern Gulf of Mexico between 1992 and 1998 (Hansen et al. 1996; Mullin and Hoggard 2000).

Figure 1. Distribution of spinner dolphin on-effort sightings from SEFSC vessel surveys during spring 1996-2001, summer 2003, spring 2004, summer 2009, summer 2017, and summer/fall 2018. Isobaths are the 200-m, 1,000-m, and 2,000-m depth contours. The darker line indicates the U.S. EEZ.

All the cetacean species found in the oceanic northern Gulf of Mexico almost certainly occur in similar habitat beyond U.S. boundaries in the southern Gulf. There are fewer cetacean sighting and stranding records in the southern Gulf due to more limited effort. Nevertheless, there are records for most oceanic species in the southern Gulf (e.g., Jefferson and Schiro 1997; Ortega Ortiz 2002; Ortega-Argueta et al. 2005; Jefferson et al. 2008; Vázquez Castán et al. 2009; Whitt et al. 2011). Although there are only a few records from Gulf of Mexico waters beyond U.S. boundaries (e.g., Jefferson and Schiro 1997, Ortega Ortiz 2002), spinner dolphins almost certainly occur throughout the oceanic Gulf of Mexico (Jefferson et al. 2008). This is therefore likely a transboundary stock with Cuba and/or Mexico. Because U.S. waters only comprise about 40% of the entire Gulf of Mexico and 35% of the oceanic (i.e., >200 m) Gulf of Mexico (Mullin and Fulling 2004), abundance and stock boundaries of oceanic species are poorly known, which is also composed of waters belonging to Mexico and Cuba where there is currently little information on cetacean species abundance and distribution. U.S. waters only comprise about 40% of the entire Gulf of Mexico, and 65% of oceanic waters are south of the U.S. Exclusive Economic Zone (EEZ).
Figure 1. Distribution of spinner dolphin sightings from SEFSC vessel surveys during spring 1996-2001, summer 2003 and spring 2004, and summer 2009. All the on-effort sightings are shown, though not all were used to estimate abundance. Solid lines indicate the 100m and 1,000m isobaths and the offshore extent of the U.S. EEZ.

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

POPULATION SIZE

The best abundance estimate (Nbest) available for the northern Gulf of Mexico spinner dolphin is 2,991 (CV=0.54; Table 1). This estimate is from summer 2017 and summer/fall 2018 oceanic surveys covering waters from the 200-m isobath to the seaward extent of the U.S. EEZ (Garrison et al. 2020). The best abundance estimate available for northern Gulf of Mexico spinner dolphins is 11,441 (CV=0.83; Table 1). This estimate is from a summer 2009 oceanic survey covering waters from the 200m isobath to the seaward extent of the U.S. EEZ.

Earlier abundance estimates

Please see Appendix IV for a summary of abundance estimates, including earlier estimates and survey descriptions. All estimates of abundance were derived through the application of distance sampling analysis (Buckland et al. 2001) and the computer program DISTANCE (Thomas et al. 1998) to line transect survey data collected from ships in the oceanic northern Gulf of Mexico (i.e., 200m isobath to seaward extent of the U.S. EEZ) and are summarized in Appendix IV.

— From 1991 through 1994, and from 1996 through 2001 (excluding 1998), annual surveys were conducted during spring along a fixed plankton-sampling trackline. Due to limited survey effort in any given year, the survey effort-weighted estimated average abundance of spinner dolphins for all surveys combined was estimated. For 1991 to 1994, the estimate was 6,316 (CV=0.43) (Hansen et al. 1995), and for 1996 to 2001, 11,971 (CV=0.71) (Mullin and Fulling 2004; Table 1).

— During summer 2003 and spring 2004, surveys dedicated to estimating cetacean abundance were conducted along a grid of uniformly-spaced transect lines from a random start. The abundance estimate for spinner dolphins, pooled from 2003 to 2004, was 1,989 (CV=0.48) (Mullin 2007; Table 1).

Recent surveys and abundance estimates

An abundance estimate for spinner dolphins was generated from vessel surveys conducted in the northern Gulf of Mexico from the continental shelf edge (~200-m isobath) to the seaward extent of the U.S. EEZ (Garrison et al. 2020). One survey was conducted from 2 July to 25 August 2017 and consisted of 7,302 km of on-effort trackline, and the second survey was conducted from 11 August to 6 October 2018 and consisted of 6,473 km of on-effort trackline within the surveyed strata. Both surveys used a double-platform data-collection procedure, which allowed estimation of the detection probability on the trackline using the independent observer approach assuming point independence (Laake and Borchers 2004). Abundance was calculated using mark-recapture distance sampling implemented in package mrds (version 2.21, Laake et al. 2020) in the R statistical programming language. This approach accounted for the effects of covariates (e.g., sea state, glare) on detection probability within the surveyed
strip. The surveys were conducted in "passing mode" (e.g., Schwarz et al. 2010), while all prior surveys in the Gulf of Mexico have been conducted in "closing mode." The abundance estimate for this stock included sightings of unidentified dolphins that were apportioned among identified species based on their relative density within the survey strata (Garrison et al. 2020). There were sightings of spinner dolphins in 2017 but there were none in 2018. The inverse variance weighted mean abundance estimate for spinner dolphins in oceanic waters during 2017 and 2018 is 2,991 (CV=0.54; Table 1; Garrison et al. 2020). Unlike previous abundance estimates, this estimate was corrected for the probability of detection on the trackline. During summer 2009, a line-transect survey dedicated to estimating the abundance of oceanic cetaceans was conducted in the northern Gulf of Mexico. Survey lines were stratified in relation to depth and the location of the Loop Current. The abundance estimate for spinner dolphins in oceanic waters during 2009 was 11,441 (CV=0.83; Table 1).

Table 1. Most recent abundance estimate (Nbest) and coefficient of variation (CV) of northern Gulf of Mexico spinner dolphins in oceanic waters (200 m to the offshore extent of the EEZ) based on an average from summer 2017 and summer/fall 2018 vessel surveys.

<table>
<thead>
<tr>
<th>Years</th>
<th>Area</th>
<th>Nbest</th>
<th>CV</th>
</tr>
</thead>
<tbody>
<tr>
<td>2017, 2018</td>
<td>Gulf of Mexico</td>
<td>2,991</td>
<td>0.54</td>
</tr>
</tbody>
</table>

Table 1. Summary of abundance estimates for northern Gulf of Mexico spinner dolphins. Month, year and area covered during each abundance survey, and resulting abundance estimate (Nbest) and coefficient of variation (CV).

<table>
<thead>
<tr>
<th>Month/Year</th>
<th>Area</th>
<th>Nbest</th>
<th>CV</th>
</tr>
</thead>
<tbody>
<tr>
<td>Apr-Jun 1991-1994</td>
<td>Oceanic waters</td>
<td>6,316</td>
<td>0.43</td>
</tr>
<tr>
<td>Apr-Jun 1996-2001 (excluding 1998)</td>
<td>Oceanic waters</td>
<td>11,971</td>
<td>0.71</td>
</tr>
<tr>
<td>Jun-Aug 2003, Apr-Jun 2004</td>
<td>Oceanic waters</td>
<td>1,989</td>
<td>0.48</td>
</tr>
<tr>
<td>Jun-Aug 2009</td>
<td>Oceanic waters</td>
<td>11,441</td>
<td>0.83</td>
</tr>
</tbody>
</table>

Minimum Population Estimate

The minimum population estimate (Nmin) is the lower limit of the two-tailed 60% confidence interval of the log-normal distributed abundance estimate. This is equivalent to the 20th percentile of the log-normal distributed abundance estimate as specified by Wade and Angliss (1997). The best estimate of abundance for spinner dolphins is 2,99111,441 (CV=0.5483). The minimum population estimate for the northern Gulf of Mexico spinner dolphin is 1,954 (Table 2) 6,221 spinner dolphins.

Current Population Trend

Four point estimates of spinner dolphin abundance have been made based on data from surveys covering 1991-2009. The statistical power to detect a trend in abundance for this stock is poor due to the relatively imprecise abundance estimates and long intervals between surveys. For example, the power to detect a precipitous decline in abundance (i.e., 50% decrease in 15 years) with estimates of low precision (e.g., CV>0.30) remains below 80%.
(alpha=0.30) unless surveys are conducted on an annual basis (Taylor et al. 2007). In addition, because these surveys are restricted to U.S. waters, it is not possible to distinguish between changes in population size and Gulf-wide shifts in spatial distribution. Five point estimates of spinner dolphin abundance have been made based on data from surveys during: 2003 (June–August), 2004 (April–June), 2009 (July–August), 2017 (July–August), and 2018 (August–October). Each of these surveys had a similar design and was conducted using the same vessel or a vessel with a similar observation platform. Prior year surveys only employed a single survey team. The estimates for 2017 and 2018 were conducted in "passing" mode which resulted in increased numbers of unidentified sightings and may have affected group size estimation for distant groups of dolphins and small whales. Apportioning unidentified species among identified taxa addresses the first issue, and there were no apparent relationships between sighting distance and estimated group size. The model for detection probability on the trackline from the summer 2017 survey was applied to the abundance estimates from the 2003, 2004, and 2009 surveys. The resulting abundance estimates were: 2003, N=5,160 (CV=0.55); 2004, N=24,536 (CV=0.58); 2009, N=19,678 (CV=0.53); 2017, N=5,982 (CV=0.54); and 2018, N=0 (CV=NA). Pairwise comparisons of the non-zero log-transformed means were conducted between years, and significant differences were assessed at alpha=0.10. P-values were adjusted for multiple comparisons. There were no significant differences between survey years. The estimates vary by a maximum factor of six. To determine whether changes in abundance have occurred over this period, an analysis of all the survey data needs to be conducted which incorporates covariates (e.g., survey conditions, season) that could potentially affect estimates. Nevertheless, differences in temporal abundance estimates will still be difficult to interpret without a Gulf of Mexico wide understanding of spinner dolphin abundance. The oceanography of the Gulf of Mexico is quite dynamic, and the spatial scale of the Gulf is small relative to the ability of most cetacean species to travel. Studies based on abundance and distribution surveys restricted to U.S. waters are unable to detect temporal shifts in distribution beyond U.S. waters that might account for any changes in abundance.

**CURRENT AND MAXIMUM NET PRODUCTIVITY RATES**

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

**POTENTIAL BIOLOGICAL REMOVAL**

Potential Biological Removal (PBR) is the product of the minimum population size, one half the maximum net productivity rate and a recovery factor (MMPA Sec. 3.16 U.S.C. 1362; Wade and Angliss 1997). The minimum population size is 1,954. The maximum productivity rate is 0.04, the default value for cetaceans. The recovery factor, which accounts for endangered, depleted, threatened stocks, or stocks of unknown status relative to optimum sustainable population (OSP), is assumed to be 0.5 because the stock is of unknown status. PBR for the northern Gulf of Mexico spinner dolphin is 20 (Table 2).

<table>
<thead>
<tr>
<th>Nb&lt;sub&gt;es&lt;/sub&gt;</th>
<th>CV</th>
<th>N&lt;sub&gt;min&lt;/sub&gt;</th>
<th>Fr</th>
<th>R&lt;sub&gt;max&lt;/sub&gt;</th>
<th>PBR</th>
</tr>
</thead>
<tbody>
<tr>
<td>2,991</td>
<td>0.54</td>
<td>1,954</td>
<td>0.50</td>
<td>0.04</td>
<td>20</td>
</tr>
</tbody>
</table>

**ANNUAL HUMAN-CAUSED MORTALITY AND SERIOUS INJURY**

Total annual estimated fishery-related mortality and serious injury to this stock during 2014–2018 was presumed to be zero, as there were no reports of mortalities or serious injuries to spinner dolphins in the Gulf of Mexico (Table 3). Mean annual mortality and serious injury during 2014–2018 due to other human-caused actions (the Deepwater Horizon oil spill) was predicted to be 113. The minimum total mean annual human-caused mortality and serious injury for this stock during 2014–2018 was, therefore, 113.

Table 3. Total annual estimated fishery-related mortality and serious injury for the northern Gulf of Mexico spinner dolphin.

<table>
<thead>
<tr>
<th>Years</th>
<th>Source</th>
<th>Annual Avg.</th>
<th>CV</th>
</tr>
</thead>
<tbody>
<tr>
<td>2014–2018</td>
<td>U.S. fisheries using observer data</td>
<td>0</td>
<td>-</td>
</tr>
</tbody>
</table>

Fisheries Information

The commercial fishery which potentially could interact with this stock in the Gulf of Mexico is the Category I Atlantic Ocean, Caribbean, Gulf of Mexico large pelagics longline fishery (Appendix III). Percent observer coverage (percentage of sets observed) for this fishery for each year during 2014–2018 was 18, 19, 23, 13 and 20, respectively. Pelagic swordfish, tunas and billfish are the targets of the longline fishery operating in the northern Gulf of Mexico. During 2014–2018 there were no observed mortalities or serious injuries to spinner dolphins by this fishery (Garrison and Stokes 2016; 2017; 2019; 2020; in review). There were no reports of mortality or serious injury to spinner dolphins by this fishery during 1998–2010 (Yeung 1999; Yeung 2001; Garrison 2003; Garrison and Richards 2004; Garrison 2005; Fairfield-Walsh and Garrison 2006; Fairfield-Walsh and Garrison 2007; Fairfield and Garrison 2008; Garrison et al. 2009; Garrison and Stokes 2010; 2011).

Other Mortality

There were 13 Eleven spinner dolphins were reported stranded of spinner dolphins in the Gulf of Mexico during 2006–2014–2018–2010, including one mass stranding of 11 individuals in Florida during 2016 (Table 4; NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 21 May 2019). Evidence of human interactions was detected for 1 stranded spinner dolphin, which stranded alive visibly oiled during June 2010 near Pensacola, Florida. No evidence of human interaction was detected for the remaining 12 stranded spinner dolphins, and for the remaining 9 eight spinner dolphins, it could not be determined if there was evidence of human interactions. Stranding data probably underestimate the extent of human and fishery-related mortality and serious injury because not all of the dolphins that die or are seriously injured in human interactions wash ashore, or, if they do, they are not all recovered (Peltier et al. 2012; Wells et al. 2015). In particular, oceanic stocks in the Gulf of Mexico are less likely to strand than nearshore coastal stocks or shelf stocks (Williams et al. 2011). Additionally, not all carcasses will show evidence of human interaction, entanglement or other fishery-related interaction due to decomposition, scavenger damage, etc. (Byrd et al. 2014). Finally, the level of technical expertise among stranding network personnel varies widely as does the ability to recognize signs of human interaction. Stranding data probably underestimate the extent of fishery-related mortality and serious injury because not all of the marine mammals which die or are seriously injured in fishery interactions wash ashore, not all that wash ashore are discovered, reported or investigated, 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 interactions.

An Unusual Mortality Event (UME) was declared for cetaceans in the northern Gulf of Mexico beginning 1 February 2010 and ending 31 July 2014 (Litz et al. 2014; https://www.fisheries.noaa.gov/national/marine-life-distress/2010-2014-cetacean-unusual-mortality-event-northern-gulf-mexico); and, as of early 2012, the event is still ongoing. It includes cetaceans that stranded prior to the Deepwater Horizon (DWH) oil spill (see “Habitat Issues” below), during the spill, and after. Exposure to the DWH oil spill was determined to be the primary underlying cause of the elevated stranding numbers in the northern Gulf of Mexico after the spill (e.g., Schwacke et al. 2014; Venn-Watson et al. 2015; Colegrove et al. 2016; DWH NRDAT 2016; see Habitat Issues section). Twelve spinner dolphin strandings were considered to be part of this UME, one of which occurred during 2014. During 2010–2013, 7 animals from this stock were considered to be part of the UME.

A population model was developed to estimate the injury and time to recovery for stocks affected by the DWH oil spill, taking into account long-term effects resulting from mortality, reproductive failure, reduced survival rates, and the proportion of the stock exposed to DWH oil (DWH MMIQT 2015). Overall, the model estimated that this stock experienced a 23% maximum reduction in population size due to the oil spill (DWH MMIQT 2015). The mortality projected for the years 2010–2013 due to the spill has not been reported previously. Based on the population model, it was projected that 1114 spinner dolphins died during 2010–2013 (four year annual average of 278) due to elevated mortality associated with oil exposure (see Appendix VI). For the 2014–2018 reporting period of this SAR,
the population model estimated 566 spinner dolphins died due to elevated mortality associated with oil exposure. The population model used to predict spinner dolphin mortality due to the DWH event has a number of sources of uncertainty. Model parameters (e.g., survival rates, reproductive rates, and life-history parameters) were derived from literature sources for spinner dolphins occupying waters outside of the Gulf of Mexico. In addition, proxy values for the effects of DWH oil exposure on both survival rates and reproductive success were applied based upon estimated values for common bottlenose dolphins in Barataria Bay. Finally, there was no estimation of uncertainty in model parameters or outputs.

Table 4. Spinner dolphin strandings along the northern Gulf of Mexico coast, 2014–2018. Data are from the NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 21 May 2019. There were no strandings of spinner dolphins in Alabama or Mississippi.

<table>
<thead>
<tr>
<th>State</th>
<th>2014</th>
<th>2015</th>
<th>2016</th>
<th>2017</th>
<th>2018</th>
<th>Total</th>
</tr>
</thead>
<tbody>
<tr>
<td>Florida</td>
<td>0</td>
<td>0</td>
<td>11(^b)</td>
<td>0</td>
<td>0</td>
<td>11</td>
</tr>
<tr>
<td>Louisiana</td>
<td>1(^a)</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>1</td>
</tr>
<tr>
<td>Texas</td>
<td>0</td>
<td>1</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>1</td>
</tr>
<tr>
<td>TOTAL</td>
<td>1</td>
<td>1</td>
<td>11</td>
<td>0</td>
<td>0</td>
<td>13</td>
</tr>
</tbody>
</table>

\(^{a}\) This stranding was part of the Northern Gulf of Mexico UME.

\(^{b}\) This was a mass strandings of 11 animals.

Table 2. Spinner dolphin (Stenella longirostris longirostris) strandings along the northern Gulf of Mexico coast, 2006–2010.

<table>
<thead>
<tr>
<th>STATE</th>
<th>2006</th>
<th>2007</th>
<th>2008</th>
<th>2009</th>
<th>2010</th>
<th>TOTAL</th>
</tr>
</thead>
<tbody>
<tr>
<td>Alabama</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>3</td>
<td>0</td>
<td>3</td>
</tr>
<tr>
<td>Florida</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>2(^*)</td>
<td>2</td>
</tr>
<tr>
<td>Louisiana</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
</tr>
<tr>
<td>Mississippi</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
</tr>
<tr>
<td>Texas</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>1</td>
<td>1</td>
</tr>
<tr>
<td>TOTAL</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>3</td>
<td>8</td>
<td>11</td>
</tr>
</tbody>
</table>

\(^{*}\)These strandings are included in the Northern Gulf of Mexico UME.
HABITAT ISSUES

The Deepwater Horizon (DWH) MC252 drilling platform, located approximately 50 miles southeast of the Mississippi River Delta in waters about 1,500 m deep, exploded on 20 April 2010. The rig sank, and over 87 days ~3.2 million barrels of oil and gas were discharged from the wellhead until it was capped on 15 July 2010 [DWH NRDAT 2016]. During the response effort dispersants were applied extensively at the seafloor and at the sea surface (Lehr et al. 2010; OSAT 2010). In situ burning, or controlled burning of oil at the surface, was also used extensively as a response tool (Lehr et al. 2010). The oil, dispersant and burn residue compounds present ecological concerns. The magnitude of this oil spill was unprecedented in U.S. history, causing impacts to wildlife, natural habitats and human communities along coastal areas from western Louisiana to the Florida Panhandle (NOAA 2011). It could be years before the entire scope of damage is ascertained (NOAA 2011).

Shortly after the oil spill, the Natural Resource Damage Assessment (NRDA) process was initiated under the Oil Pollution Act of 1990. A variety of NRDA research studies are being conducted to determine potential impacts of the spill on marine mammals. These studies estimated that 47% (95% CI: 24–91) of spinner dolphins in the Gulf were exposed to oil, that 21% (95% CI: 10–30) of females suffered from reproductive failure, and 17% (95% CI: 6–27) of spinner dolphins suffered adverse health effects (DWH MMIQT 2015). A population model estimated the stock experienced a maximum 23% reduction in population size (see Other Mortality section above).

Anthropogenic sound in the world’s oceans has been shown to affect marine mammals, with vessel traffic, seismic surveys, and active naval sonars being the main anthropogenic contributors to low- and mid-frequency noise in oceanic waters (e.g., Nowacek et al. 2015; Gomez et al. 2016; NMFS 2018). The long-term and population consequences of these impacts are less well-documented and likely vary by species and other factors. Impacts on marine mammal prey from sound are also possible (Carroll et al. 2017), but the duration and severity of any such prey effects on marine mammals are unknown. These studies have focused on identifying the type, magnitude, severity, length and impact of oil exposure to oceanic, coastal and estuarine marine mammals. The research is ongoing and likely will continue for some time. For continental shelf and oceanic cetaceans, the NOAA led efforts include: aerial surveys to document the distribution, abundance, species and exposure of marine mammals and turtles relative to oil from DWH spill; and ship surveys to evaluate exposure to oil and other chemicals and to assess changes in animal behavior and distribution relative to oil exposure through visual and acoustic surveys, deployment of passive acoustic monitoring systems, collection of tissue samples, and deployment of satellite tags on sperm and Bryde’s whales.

Aerial surveys have observed Risso’s dolphins, spinner dolphins, pantropical spotted dolphins, striped dolphins, bottlenose dolphins and sperm whales swimming in oil in offshore waters (NOAA 2010a). The effects of oil exposure on marine mammals depend on a number of factors including the type and mixture of chemicals involved, the amount, frequency and duration of exposure, the route of exposure (inhaled, ingested, absorbed, or external) and biomedical risk factors of the particular animal (Geraci 1990; NOAA 2010b). In general, direct external contact with petroleum compounds or dispersants with skin may cause skin irritation, chemical burns and infections. For large whales, oil can foul the baleen they use to filter-feed. Inhalation of volatile petroleum compounds or dispersants may irritate or injure the respiratory tract, which could lead to pneumonia or inflammation. Ingestion of petroleum compounds may cause injury to the gastrointestinal tract, which could affect an animal's ability to digest or absorb food. Absorption of petroleum compounds or dispersants may damage kidney, liver and brain function in addition to causing immune suppression and anemia. Long term chronic effects such as lowered reproductive success and decreased survival may occur (Geraci 1990; NOAA 2010b).

STATUS OF STOCK

Spinner dolphins are not listed as threatened or endangered under the Endangered Species Act, but the northern Gulf of Mexico stock is considered strategic under the MMPA because the mean modeled annual human-caused mortality and serious injury due to the DWH oil spill exceeds PBR. No fishery-related mortality or serious injury has been observed in recent years; therefore, total fishery-related mortality and serious injury can be considered to be insignificant and approaching zero mortality and serious injury rate. The status of spinner dolphins in the northern Gulf of Mexico, relative to OSP, is unknown. There was no statistically significant trend in population size for this stock in the northern Gulf of Mexico. The species is not listed as threatened or endangered under the Endangered Species Act. There are insufficient data to determine the population trends for this species. Total human caused mortality and serious injury for this stock is not known. There is insufficient information available to determine whether the total fishery-related mortality and serious injury for this stock is insignificant and approaching zero...
mortality and serious injury rate. This is not a strategic stock because it is assumed that the average annual human-related mortality and serious injury does not exceed PBR.

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Garrison, L.P. and L. Stokes. in review. Estimated bycatch of marine mammals and sea turtles in the U.S. Atlantic pelagic longline fleet during 2018. Southeast Fisheries Science Center, Protected Resources and Biodiversity Division, 75 Virginia Beach Dr., Miami, Florida 33140. PRD Contribution #: PRD-2020-08. 55 pp.


ROUGH-TOOTHED DOLPHIN (Steno bredanensis): Northern Gulf of Mexico Stock

STOCK DEFINITION AND GEOGRAPHIC RANGE

Rough-toothed dolphins (Steno bredanensis) are distributed worldwide in the Atlantic, Pacific, and Indian Oceans, generally in warm temperate, subtropical, or tropical waters. They are commonly reported in a wide range of water depths, from shallow, nearshore waters to oceanic waters (West et al. 2011). The rough-toothed dolphin is distributed worldwide in tropical to warm temperate waters (Leatherwood and Reeves 1983; Miyazaki and Perrin 1994; West et al. 2011). In the northern Gulf of Mexico (i.e., U.S. Gulf of Mexico), rough-toothed dolphins occur in oceanic and to a lesser extent continental shelf waters in the northern Gulf of Mexico (i.e., U.S. Gulf of Mexico) (Figure 1; Fulling et al. 2003; Mullin and Fulling 2004; Maze-Foley and Mullin 2006; Garrison and Aichinger Dias 2020). They have been observed rough-toothed dolphins were seen in all seasons during NMFS visual surveys in the Gulf of Mexico GulfCet aerial surveys of the northern Gulf of Mexico between 1992 and 1998 (Hansen et al. 1996; Mullin and Hoggard 2000) but are not seen every survey year attesting to their low density in this region. Four dolphins from a mass stranding of 62 animals in the Florida Panhandle in December 1997 were rehabilitated and released in 1998, and satellite-linked transmitters on three of these were tracked for 4 to 112 days. A report after five months indicated that the animals returned to, and remained in, northeastern Gulf waters averaging about 195 m deep offshore of the original stranding site (Wells et al. 1999).

All the cetacean species found in the oceanic northern Gulf of Mexico almost certainly occur in similar habitat beyond U.S. boundaries in the southern Gulf. There are fewer cetacean sighting and stranding records in the southern Gulf due to more limited effort. Nevertheless there are records for most oceanic species in the southern Gulf (e.g., Jefferson and Schiro 1997; Ortega Ortiz 2002; Ortega-Argueta et al. 2005; Jefferson et al. 2008; Vázquez Castán et al. 2009; Whitt et al. 2011). This is therefore likely a transboundary stock with Cuba and/or Mexico. Because U.S. waters only comprise about 40% of the entire Gulf of Mexico and 35% of the oceanic (i.e., >200 m) Gulf of Mexico (Mullin and Fulling 2004), abundance and stock boundaries of oceanic species are poorly known. Although there are only a few records from Gulf of Mexico waters beyond U.S. boundaries (e.g., Jefferson and Schiro 1997, Ortega Ortiz 2002), rough-toothed dolphins almost certainly occur throughout the oceanic Gulf of Mexico (Jefferson et al. 2008), which is also composed of waters belonging to Mexico and Cuba where there is currently little information on cetacean

Figure 1. Distribution of rough-toothed dolphin on-effort sightings from SEFSC vessel surveys during spring 1996-2001, summer 2003, spring 2004, summer 2009, summer 2017, and summer/fall 2018. Isobaths are the 200-m, 1,000-m, and 2,000-m depth contours. The darker line indicates the U.S. EEZ.
species abundance and distribution. U.S. waters comprise only about 40% of the entire Gulf of Mexico and 35% of the oceanic (i.e., >200 m) Gulf of Mexico.

Rough-toothed dolphins in the northern Gulf of Mexico are managed separately from those in the western North Atlantic. Several lines of evidence support this distinction. Four dolphins from a mass stranding of 62 animals in the Florida Panhandle in December 1997 were rehabilitated and released in 1998, and satellite-linked transmitters on three of these were tracked for up to 112 days. A report after five months indicated that the animals returned to, and remained in, northeastern Gulf waters (Wells et al. 2008), providing evidence for fidelity to the Gulf. In addition, analyses of worldwide genetic differentiation in Steno indicate animals in the western Atlantic Ocean are strongly differentiated from those in the Pacific and Indian Oceans (Albertson 2014; da Silva et al. 2015). Albertson (2014) illustrated that this species may exhibit fine-scale population structure and da Silva et al. (2015) provided evidence for multiple populations in the western South Atlantic. Finally, the separation of Atlantic and Gulf of Mexico stocks is consistent with the fact that the two areas belong to distinct marine ecoregions (Spalding et al. 2007; Moore and Merrick 2011).

There are insufficient data to determine whether the northern Gulf of Mexico stock comprises multiple demographically independent populations. Additional morphological, acoustic, genetic, and/or behavioral data are needed to further delineate population structure within the Gulf of Mexico and across the broader geographic area. The Gulf of Mexico population is being considered 1 stock for management purposes, although there is currently no information to differentiate this stock from the Atlantic Ocean stock(s), nor information on whether more than 1 stock may exist in the Gulf of Mexico. Additional morphological, genetic and or behavioral data are needed to provide further information on stock delineation.

**POPULATION SIZE**

The best abundance estimate (Nbest) for the northern Gulf of Mexico rough-toothed dolphin is unknown (Table 1) since no sightings of this species were made during the summer 2017 or summer/fall 2018 oceanic surveys covering waters from the 200-m isobath to the seaward extent of the U.S. EEZ (Garrison et al. 2020). Current population size for the rough-toothed dolphin in the northern Gulf of Mexico is 624 (CV=0.99; Table 1; Garrison 2016). This estimate is from a summer 2009 oceanic survey covering waters from the 200-m isobath to the seaward extent of the U.S. Exclusive Economic Zone (EEZ).

**Earlier abundance estimates**

Estimates of abundance were derived through the application of distance sampling analysis (Buckland et al. 2001) and the computer program DISTANCE (Thomas et al. 1998) to line transect survey data. During summer 2003 and spring 2004, ship surveys dedicated to estimating cetacean abundance were conducted in oceanic waters along a grid of uniformly-spaced transect lines from a random start. The abundance estimate for rough-toothed dolphins in oceanic waters, pooled from 2003 to 2004, was 1,508 (CV=0.39) (Mullin 200 Please see Appendix IV for a summary of abundance estimates, including earlier estimates and survey descriptions.

**Recent surveys and abundance estimates**

Two vessel surveys were conducted in the northern Gulf of Mexico from the continental shelf edge (~200-m isobath) to the seaward extent of the U.S. EEZ (Garrison et al. 2020). One survey was conducted from 2 July to 25 August 2017 and consisted of 7,302 km of on-effort trackline, and the second survey was conducted from 11 August to 6 October 2018 and consisted of 6,473 km of on-effort trackline within the surveyed strata. Both surveys used a double-platform data-collection procedure, which allowed estimation of the detection probability on the trackline using the independent observer approach assuming point independence (Laake and Borchers 2004). The surveys were conducted in "passing mode" (e.g., Schwarz et al. 2010) while all prior surveys in the Gulf of Mexico have been conducted in "closing mode." No sightings of rough-toothed dolphins were made during these two vessel surveys; therefore, the abundance estimate for rough-toothed dolphins is unknown.

During summer 2009, a line transect shipboard survey dedicated to estimating the abundance of oceanic cetaceans was conducted in the northern Gulf of Mexico covering waters depths from the 200-m isobath to the seaward extent of the U.S. EEZ (Garrison 2016). Survey lines were stratified in relation to depth and the location of the Loop Current. In total, 4,600 km of trackline were surveyed using a single visual observation team. The abundance estimate for rough-toothed dolphins in oceanic waters during 2009 was 624 (CV=0.99; Table 1; Garrison 2016). This is the most reliable current estimate for the northern Gulf of Mexico but it is probably an underestimate. This estimate does not
include Gulf of Mexico continental shelf waters where an estimate based on 1998–2001 surveys was over 1,000 rough-toothed dolphins (Fulling et al. 2003). There is not a recent estimate for continental shelf waters.

**Table 1. Summary of recent abundance estimates for rough-toothed dolphins in the northern Gulf of Mexico oceanic waters (200 m to the offshore extent of the EEZ) by month, year, and area covered during each abundance survey and the resulting abundance estimate (N_best) and coefficient of variation (CV).**

<table>
<thead>
<tr>
<th>Years</th>
<th>Area</th>
<th>N_best</th>
<th>CV</th>
</tr>
</thead>
<tbody>
<tr>
<td>2017, 2018</td>
<td>Gulf of Mexico</td>
<td>Unknown</td>
<td>-</td>
</tr>
</tbody>
</table>

**Minimum Population Estimate**

The minimum population estimate (Nmin) is the lower limit of the two-tailed 60% confidence interval of the log-normal distributed abundance estimate. This is equivalent to the 20th percentile of the log-normal distributed abundance estimate as specified by Wade and Angliss (1997). The best and minimum estimates of abundance for rough-toothed dolphins are unknown is 624 (CV=0.99). The minimum population estimate for northern Gulf of Mexico rough-toothed dolphins is 311.

**Current Population Trend**

The statistical power to detect a trend in abundance for this stock is poor due to the relatively imprecise abundance estimates and long intervals between surveys. For example, the power to detect a precipitous decline in abundance (i.e., 50% decrease in 15 years) with estimates of low precision (e.g., CV>0.30) remains below 80% (alpha=0.30) unless surveys are conducted on an annual basis (Taylor et al. 2007). In addition, because these surveys are restricted to U.S. waters, it is not possible to distinguish between changes in population size and Gulf-wide shifts in spatial distribution. Five point estimates of rough-toothed dolphin abundance have been made based on data from surveys during: 2003 (June−August), 2004 (April−June), 2009 (July−August), 2017 (July−August), and 2018 (August−October). Each of these surveys had a similar design, employed similar methods, and was conducted using the same vessel or a vessel with a similar observation platform. These estimates are: 2003, N=9,253 (CV=0.78); 2004, N=0 (CV=NA); 2009, N=3,509 (CV=0.67); 2017, N=0 (CV=NA); and 2018, N=0 (CV=NA). A pairwise comparison of the non-zero log-transformed means was conducted between years, and significant difference was assessed at alpha=0.10. There was no significant difference between survey years. A trend analysis has not been conducted for this stock. Two point estimates of rough-toothed dolphin abundance have been made based on data from oceanic surveys during 2003–2004 and 2009 (Table 1). The estimates vary by a factor of more than two. To determine whether changes in oceanic abundance have occurred over this period, an analysis of all the survey data needs to be conducted which incorporates covariates (e.g., survey conditions, season) that could potentially affect estimates. It should be noted that since this is a transboundary stock and the abundance estimates are for U.S. waters only, it will be difficult to interpret any detected trends. Additionally, the extent to which rough-toothed dolphins inhabit continental shelf waters and whether there is movement between these waters and oceanic waters needs to be resolved.
CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

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

POTENTIAL BIOLOGICAL REMOVAL

Potential Biological Removal (PBR) is currently undetermined. PBR is the product of the minimum population size, one half the maximum net productivity rate and a recovery factor (MMPA Sec. 3.16 U.S.C. 1362; Wade and Angliss 1997). The minimum population size is unknown. The maximum productivity rate is 0.04, the default value for cetaceans. The recovery factor is 0.40 because the CV of the average mortality estimate is greater than 0.8 (Wade and Angliss 1997; Table 2). PBR for the northern Gulf of Mexico rough-toothed dolphin is 3.1.

Table 2. Best and minimum abundance estimates for the northern Gulf of Mexico rough-toothed dolphin stock with Maximum Productivity Rate (Rmax), Recovery Factor (Fr) and PBR.

<table>
<thead>
<tr>
<th>Nbest</th>
<th>CV</th>
<th>Nmin</th>
<th>Fr</th>
<th>Rmax</th>
<th>PBR</th>
</tr>
</thead>
<tbody>
<tr>
<td>Unknown</td>
<td>-</td>
<td>Unknown</td>
<td>0.40</td>
<td>0.04</td>
<td>Undetermined</td>
</tr>
</tbody>
</table>

ANNUAL HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

The estimated mean annual fishery-related mortality and serious injury for this stock during 2014–2018 was 0.8 rough-toothed dolphins (CV=1.00; Table 2) due to interactions with the large pelagics longline fishery, and 0.2 rough-toothed dolphins due to an interaction with the hook and line fishery (see Fisheries Information sections below; Tables 3–4). Mean annual mortality and serious injury during 2014–2018 due to other human-caused actions (the Deepwater Horizon oil spill) was predicted to be 38. The minimum total mean annual human-caused mortality and serious injury for this stock during 2014–2018 was, therefore, 39 (Table 5).

Table 3. Total annual estimated fishery-related mortality and serious injury for the northern Gulf of Mexico rough-toothed dolphin stock.

<table>
<thead>
<tr>
<th>Years</th>
<th>Source</th>
<th>Annual Avg.</th>
<th>CV</th>
</tr>
</thead>
<tbody>
<tr>
<td>2014–2018</td>
<td>U.S. fisheries using observer data</td>
<td>1.0</td>
<td>1.00</td>
</tr>
</tbody>
</table>

New Serious Injury Guidelines

— NMFS updated its serious injury designation and reporting process, which uses guidance from previous serious injury workshops, expert opinion, and analysis of historic injury cases to develop new criteria for distinguishing serious from non-serious injury (Angliss and DeMaster 1998; Andersen et al. 2008; NMFS 2012). NMFS defines serious injury as an “injury that is more likely than not to result in mortality.” Injury determinations for stock assessments revised in 2013 or later incorporate the new serious injury guidelines, based on the most recent 5-year period for which data are available.

Fisheries Information

There are three commercial fisheries that interact, or that potentially could interact, with this stock in the Gulf of Mexico. These include two Category I fisheries, the Atlantic Highly Migratory Species (high seas) longline fishery, and the Atlantic Ocean, Caribbean, Gulf of Mexico large pelagics longline fishery, and one Category III fishery, the Atlantic Ocean, Gulf of Mexico, Caribbean commercial passenger fishing vessel (hook and line) fishery (Appendix III).

Longline
There is very little effort within the Gulf of Mexico by the Atlantic Highly Migratory Species (high seas) longline fishery, and no takes of rough-toothed dolphins within high seas waters of the Gulf of Mexico have been observed or reported thus far. Pelagic swordfish, tunas and billfish are the targets of the large pelagics longline fishery operating in the northern Gulf of Mexico. The commercial fishery that interacts, or that potentially could interact, with this stock in the Gulf of Mexico is the Atlantic Ocean, Caribbean, Gulf of Mexico large pelagics longline fishery (Appendix III). Pelagic swordfish, tunas and billfish are the targets of the large pelagics longline fishery operating in the northern Gulf of Mexico. For the five-year period 2010–2014, the estimated annual combined serious injury and mortality attributable to the large pelagics longline fishery in the northern Gulf of Mexico was 0.8 (CV=1.00) rough-toothed dolphins (Table 4; Garrison and Stokes 2016; 2017; 2019; 2020; in review). During the second quarter of 2014, one serious injury was observed (Garrison and Stokes 2016). There were no reports of mortality or serious injury to rough-toothed dolphins by this fishery in the northern Gulf of Mexico during 1998–2013 (Yeung 1999; Yeung 2001; Garrison 2003; Garrison and Richards 2004; Garrison 2005; Fairfield Walsh and Garrison 2006; Fairfield-Walsh and Garrison 2007; Fairfield and Garrison 2008; Garrison et al. 2009; Garrison and Stokes 2010, 2012a,b, 2013, 2014; 2016).

Percent observer coverage (percentage of sets observed) for the two longline fisheries for each year during 2014–2018 was 18, 19, 23, 13 and 20, respectively. During the first and second quarters (15 April–15 June) of 2014–2018, observer coverage in the Gulf of Mexico large pelagics longline fishery was greatly enhanced (approaching 55%) to collect more robust information on the interactions between pelagic longline vessels and spawning bluefin tuna. Therefore, the high annual observer coverage rates during 2014–2018 (Table 4) primarily reflect high coverage rates during the first and second quarters of each year. During these second quarters, this elevated coverage results in an increased probability that relatively rare interactions will be detected. Species within the oceanic Gulf of Mexico are presumed to be resident year-round; however, it is unknown if the bycatch rates observed during the first and second quarters are representative of that which occurs throughout the year.

Table 4. Summary of the incidental mortality and serious injury of rough-toothed dolphins by the pelagic longline commercial fishery including the years sampled (Years), the type of data used (Data Type), the annual observer coverage (Observer Coverage), the annual observed serious injury and mortality recorded by on-board observers, the annual estimated serious injury and mortality, the combined annual estimates of serious injury and mortality (Est. Combined Mortality), the estimated CV of the combined annual mortality estimates (Est. CVs), the mean of the combined annual mortality estimates, and the CV of the mean combined annual mortality estimate (CV of Mean).

<table>
<thead>
<tr>
<th>Fishery</th>
<th>Years</th>
<th>Data Type*</th>
<th>Observer Coverage*</th>
<th>Observed Serious Injury*</th>
<th>Observed Mortality</th>
<th>Estimated Serious Injury*</th>
<th>Est. Mort.</th>
<th>Est. Combined Mortality</th>
<th>Est. CVs</th>
<th>Mean Combined Annual Mortality</th>
<th>CV of Mean</th>
</tr>
</thead>
<tbody>
<tr>
<td>Pelagic Longline</td>
<td>2014</td>
<td>Obs. Data, Trip Logbook</td>
<td>0.18</td>
<td>1</td>
<td>0</td>
<td>4.2</td>
<td>0</td>
<td>4.2</td>
<td>1</td>
<td></td>
<td></td>
</tr>
<tr>
<td>2015</td>
<td>Obs. Data, Trip Logbook</td>
<td>0.19</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>2016</td>
<td>Obs. Data, Trip Logbook</td>
<td>0.23</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>2017</td>
<td>Obs. Data, Trip Logbook</td>
<td>0.13</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>2018</td>
<td>Obs. Data, Trip Logbook</td>
<td>0.20</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0.8</td>
<td>1.00</td>
<td></td>
<td></td>
</tr>
<tr>
<td>TOTAL</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td>0.8</td>
<td>1.00</td>
<td></td>
</tr>
</tbody>
</table>

* Observed data excludes data where observer coverage was not calculated.
Table 2. Summary of the incidental mortality and serious injury of northern Gulf of Mexico rough-toothed dolphins in the pelagic longline commercial fishery including the years sampled (Years), the number of vessels active within the fishery (Vessels), the type of data used (Data Type), the annual observer coverage (Observer Coverage), the annual observed serious injury and mortality recorded by on-board observers, the annual estimated serious injury and mortality, the combined annual estimates of serious injury and mortality (Estimated Combined Mortality), the estimated CV of the combined annual mortality estimates (Est. CVs) and the mean of the combined mortality estimates (CV in parentheses).

<table>
<thead>
<tr>
<th>Fisher y</th>
<th>Years</th>
<th>Vesselsa</th>
<th>Data Typeb</th>
<th>Observ er Cover agec</th>
<th>Observ ed Seriou s Injury</th>
<th>Observ ed Mortali ty</th>
<th>Estimat ed Serious Injury</th>
<th>Estimat ed Mortali ty</th>
<th>Estimat ed Combin ed Mortali ty</th>
<th>Est. CVs</th>
<th>Mean Annual Mortali ty</th>
</tr>
</thead>
<tbody>
<tr>
<td>Pelagi c Longli ne</td>
<td>2010–2014</td>
<td>46, 42, 47, 47, 44</td>
<td>Obs. Data Logb ook</td>
<td>28, 18, 11, 25, 18</td>
<td>0, 0, 0, 0, 1</td>
<td>0, 0, 0, 0</td>
<td>0, 0, 0, 0, 0</td>
<td>0, 0, 0, 0, 0</td>
<td>NA, NA, NA, NA, 1.0</td>
<td>NA, NA, NA, NA, 0.8</td>
<td>0.8 (1.0)</td>
</tr>
</tbody>
</table>

a Number of vessels in the fishery is based on vessels reporting effort to the pelagic longline logbook.
b Observer data (Obs. Data) are used to measure bycatch rates, and the data are collected within the Northeast Fisheries Observer Program. Mandatory logbook data were used to measure total effort for the longline fishery. These data are collected at the Southeast Fisheries Science Center (SEFSC). Observer coverage in the GOM is dominated by very high coverage rates during April–June associated with efforts to improve estimates of bluefin tuna bycatch.
c Proportion of sets observed.

Hook and Line (Rod and Reel)

During 2014–2018, stranding data included one mortality and one serious injury for which hook and line gear entanglement or ingestion were documented. For the mortality, the stranding data suggested the hook and line gear interaction was not a contributing factor to cause of death. Therefore, only the serious injury (Maze-Foley and Garrison 2020) was included in the annual human-caused mortality and serious injury total for this stock (Table 5). Both cases occurred in 2018 and were included in the stranding database and are included in the stranding totals presented in Table 6 (Southeast Regional Marine Mammal Stranding Network; NOAA National Marine Mammal Health and
It should be noted that, in general, it cannot be determined if rod and reel hook and line gear originated from a commercial (i.e., commercial fisherman, charter boat, or headboat) or recreational angler because the gear type used by both sources is typically the same. Also, it is not possible to estimate the total number of interactions with hook and line gear because there is no systematic observer program. The documented interactions in this gear represent a minimum known count of interactions in the last five years.

**Table 5. Summary of the incidental mortality and serious injury of rough-toothed dolphins during 2014–2018 from all sources, including observed commercial fisheries, unobserved commercial fisheries, and other sources.**

<table>
<thead>
<tr>
<th>Description</th>
<th>Value</th>
</tr>
</thead>
<tbody>
<tr>
<td>Mean Annual Mortality due to the observed commercial large pelagics longline fishery (2014–2018, Table 4)</td>
<td>0.8</td>
</tr>
<tr>
<td>Mean Annual Mortality due to the unobserved hook and line fishery (2014–2018)</td>
<td>0.2</td>
</tr>
<tr>
<td>Mean Annual Mortality due to Other Human-Caused Sources (DWH oil spill) (2014–2018)</td>
<td>38</td>
</tr>
<tr>
<td>Minimum Total Mean Annual Human-Caused Mortality and Serious Injury (2014–2018)</td>
<td>39</td>
</tr>
</tbody>
</table>

**Other Mortality**

There were six stranded rough-toothed dolphins in the northern Gulf of Mexico during 2010–2014 (Table 6; NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 21 May 2019). Evidence of human interaction was detected for two of the stranded animals, both of which were classified as fishery interactions. No evidence of human interaction was detected for one stranded animal, and for the remaining three, it could not be determined if there was evidence of human interaction. Stranding data probably underestimate the extent of human and fishery-related mortality and serious injury because not all of the dolphins that die or are seriously injured in human interactions wash ashore, or, if they do, they are not all recovered (Peltier et al. 2012; Wells et al. 2015). In particular, oceanic stocks in the Gulf of Mexico are less likely to strand than nearshore coastal stocks or shelf stocks (Williams et al. 2011). Additionally, not all carcasses will show evidence of human interaction, entanglement or other fishery-related interaction due to decomposition, scavenger damage, etc. (Byrd et al. 2014). Finally, the level of technical expertise among stranding network personnel varies widely as does the ability to recognize signs of human interaction.

An Unusual Mortality Event (UME), involving primarily bottlenose dolphins, was declared for cetaceans in the northern Gulf of Mexico beginning 1 February 2010 and ending 31 July 2014 (Litz et al. 2014; http://www.nmfs.noaa.gov/pr/health/mmume/cetacean_gulfofmexico.htm, accessed 1 June 2016). It included cetaceans that stranded prior to the Deepwater Horizon (DWH) oil spill (see “Habitat Issues” below), during the spill, and after. Exposure to the DWH oil spill was determined to be the primary underlying cause of the elevated stranding numbers in the northern Gulf of Mexico after the spill (e.g., Schwacke et al. 2014; Venn-Watson et al. 2015; Colegrove et al. 2016; DWH NRDAT 2016; see Habitat Issues section). Investigations to date have determined that the DWH oil spill is the primary underlying cause of the elevated stranding numbers in the northern Gulf of Mexico after the spill (e.g., Schwacke et al. 2014; Venn-Watson et al. 2015). One stranding of a rough-toothed dolphin in 2013 was considered to be part of this UME. During 2010–2014, 1 animal from this stock was considered to be part of the UME, a 2013 stranding in Destin, Florida.

A population model was developed to estimate the injury and time to recovery for stocks affected by the DWH oil spill, taking into account long-term effects resulting from mortality, reproductive failure, reduced survival rates, and the proportion of the stock exposed to DWH oil (DWH MMIQT 2015). Overall, the model estimated that this stock experienced a 17% maximum reduction in population size due to the oil spill (DWH MMIQT 2015).
mortality projected for the years 2010–2013 due to the spill has not been reported previously. Based on the population model, it was projected that 362 rough-toothed dolphins died during 2010–2013 (four year annual average of 91) due to elevated mortality associated with oil exposure (see Appendix VI). For the 2014–2018 reporting period of this SAR, the population model estimated 188 rough-toothed dolphins died due to elevated mortality associated with oil exposure. The population model used to predict rough-toothed dolphin mortality due to the DWH event has a number of sources of uncertainty. Model parameters (e.g., survival rates, reproductive rates, and life-history parameters) were derived from literature sources for rough-toothed dolphins occupying waters outside of the Gulf of Mexico. In addition, proxy values for the effects of DWH oil exposure on both survival rates and reproductive success were applied based upon estimated values for common bottlenose dolphins in Barataria Bay. Finally, there was no estimation of uncertainty in model parameters or outputs.

Table 6. Rough-toothed dolphin strandings along the northern Gulf of Mexico coast, 2014–2018. Data are from the NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 21 May 2019. There were no strandings of rough-toothed dolphins in Alabama or Texas.

<table>
<thead>
<tr>
<th>State</th>
<th>2014</th>
<th>2015</th>
<th>2016</th>
<th>2017</th>
<th>2018</th>
<th>Total</th>
</tr>
</thead>
<tbody>
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<td>0</td>
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<td>4</td>
</tr>
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<td>0</td>
<td>0</td>
<td>1</td>
<td>0</td>
</tr>
<tr>
<td>Mississippi</td>
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<td>0</td>
<td>0</td>
<td>0</td>
<td>1</td>
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<td>TOTAL</td>
<td>0</td>
<td>3</td>
<td>0</td>
<td>1</td>
<td>0</td>
<td>4</td>
</tr>
</tbody>
</table>

a. Both 2018 animals were classified as fishery interactions.

HABITAT ISSUES

The DWH MC252 drilling platform, located approximately 50 miles southeast of the Mississippi River Delta in waters about 1,500 m deep, exploded on 20 April 2010. The rig sank, and over 87 days up to ~3.24.9 million barrels of oil were discharged from the wellhead until it was capped on 15 July 2010 (McNutt et al. 2012; DWH NRDAT 2016). During the response effort dispersants were applied extensively at the seafloor and at the sea surface (Lehr et al. 2010; OSAT 2010). In-situ burning, or controlled burning of oil at the surface, was also used extensively as a response tool (Lehr et al. 2010). The oil, dispersant and burn residue compounds present ecological concerns (Buist et al. 1999; NOAA 2011). The magnitude of this oil spill was unprecedented in U.S. history, causing impacts to wildlife, natural habitats and human communities along coastal areas from western Louisiana to the Florida Panhandle (NOAA 2011). It could be years before the entire scope of damage is ascertained (NOAA 2011).

Shortly after the oil spill, the Natural Resource Damage Assessment (NRDA) process was initiated under the Oil Pollution Act of 1990. A variety of NRDA research studies are ongoing to determine potential impacts of the spill on marine mammals. These studies estimated that 41% (95% CI: 16–100) of rough-toothed dolphins in the Gulf were exposed to oil, that 19% (95% CI: 9–26) of females suffered from reproductive failure, and 15% (95% CI: 6–23) of rough-toothed dolphins suffered adverse health effects (DWH MMIQT 2015). A population model estimated the stock experienced a maximum 17% reduction in population size (see Other Mortality section above).

Anthropogenic sound in the world’s oceans has been shown to affect marine mammals, with vessel traffic, seismic surveys, and active naval sonars being the main anthropogenic contributors to low- and mid-frequency noise in oceanic waters (e.g., Nowacek et al. 2015; Gomez et al. 2016; NMFS 2018). The long-term and population consequences of these impacts are less well-documented and likely vary by species and other factors. Impacts on marine mammal prey from sound are also possible (Carroll et al. 2017), but the duration and severity of any such prey effects on marine mammals are unknown. These studies have focused on identifying the type, magnitude, severity, length and impact of oil exposure to oceanic, continental shelf, coastal and estuarine marine mammals. For continental shelf and oceanic cetaceans, the NOAA-led efforts include: aerial surveys to document the distribution, abundance, species and exposure relative to oil from the DWH spill; and ship surveys to evaluate exposure to oil and other chemicals and to assess changes in animal behavior and distribution relative to oil exposure through visual and acoustic surveys, deployment of passive acoustic monitoring systems, collection of tissue samples, and deployment of satellite-linked tags on sperm and Bryde’s whales.

Vessel and aerial surveys documented common bottlenose dolphins, Atlantic spotted dolphins, rough-toothed dolphins, spinner dolphins, pantropical spotted dolphins, Risso’s dolphins, striped dolphins, sperm whales,
dwarf/pygmy sperm whales and a Cuvier's beaked whale swimming in oil or potentially oil-derived substances (e.g., sheen, mousse) in offshore waters of the northern Gulf of Mexico following the DWH oil spill. The effects of oil exposure on marine mammals depend on a number of factors including the type and mixture of chemicals involved, the amount, frequency and duration of exposure, the route of exposure (inhaled, ingested, absorbed, or external) and biomedical risk factors of the particular animal (Geraci 1990; Helm et al. 2015). In general, direct external contact with petroleum compounds or dispersants with skin may cause skin irritation, chemical burns and infections. Inhalation of volatile petroleum compounds or dispersants may irritate or injure the respiratory tract, which could lead to pneumonia or inflammation. Ingestion of petroleum compounds may cause injury to the gastrointestinal tract, which could affect an animal's ability to digest or absorb food. Absorption of petroleum compounds or dispersants may damage kidney, liver and brain function in addition to causing immune suppression and anemia. Long-term chronic effects such as lowered reproductive success and decreased survival may occur (Geraci 1990; Helm et al. 2015).

STATUS OF STOCK

Rough-toothed dolphins are not listed as threatened or endangered under the Endangered Species Act. The most recent abundance surveys (2017–2018) observed no rough-toothed dolphins, rendering PBR undetermined. The northern Gulf of Mexico stock is therefore not considered strategic under the MMPA. However, the mean modeled annual human-caused mortality and serious injury due to the DWH oil spill (38 animals) greatly exceeds the previous, but expired, estimate of PBR for this stock (2.5) based on 2009 surveys. Total fishery-related mortality and serious injury for this stock was 0.8, which is not less than 10% of the previously calculated PBR, and therefore it is likely that fishery-related mortality and serious injury cannot be considered to be insignificant and approaching zero mortality and serious injury rate. The status of rough-toothed dolphins in the northern Gulf of Mexico, relative to OSP, is unknown. There was no statistically significant trend in population size for this stock in the northern Gulf of Mexico. There are insufficient data to determine the population trends for this stock.

REFERENCES


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**CLYMENE DOLPHIN (Stenella clymene): Northern Gulf of Mexico Stock**

**STOCK DEFINITION AND GEOGRAPHIC RANGE**

The Clymene dolphin is endemic to tropical and subtropical waters of the Atlantic (Leatherwood and Reeves 1983; Perrin and Mead 1994). Sightings of these animals in the northern Gulf of Mexico (i.e., U.S. Gulf of Mexico) occur primarily over the deeper waters off the continental shelf and primarily west of the Mississippi River (Mullin et al. 1994; Maze-Foley and Mullin 2006; Garrison and Aichinger Dias 2020; Figure 1). Clymene dolphins were seen in the winter, spring and summer during GulfCet aerial surveys of the northern Gulf of Mexico during 1992 to 1998.

![Figure 1. Distribution of Clymene dolphin on-effort sightings from SEFSC vessel surveys during spring 1996-2001, summer 2003, spring 2004, summer 2009, summer 2017, and summer/fall 2018. Isobaths are the 200-m, 1,000-m, and 2,000-m depth contours. The darker line indicates the U.S. EEZ.](image)

(Hansen et al. 1996; Mullin and Hoggard 2000).
All the cetacean species found in the oceanic northern Gulf of Mexico almost certainly occur in similar habitat beyond U.S. boundaries in the southern Gulf. There are fewer cetacean sighting and stranding records in the southern Gulf due to more limited effort. Nevertheless, there are records for most oceanic species in the southern Gulf (e.g., Jefferson and Schiro 1997; Ortega-Argueta et al. 2005; Jefferson et al. 2008; Vázquez Castán et al. 2009; Whitt et al. 2011). Although there are only a few records from Gulf of Mexico waters beyond U.S. boundaries (e.g., Jefferson and Schiro 1997), Clymene dolphins almost certainly occur throughout the oceanic Gulf of Mexico (Jefferson et al. 2008). This is therefore likely a transboundary stock with Cuba and/or Mexico. Because U.S. waters only comprise about 40% of the entire Gulf of Mexico and 35% of the oceanic (i.e., >200 m) Gulf of Mexico (Mullin and Fulling 2004), abundance and stock boundaries of oceanic species are poorly known, which is also composed of waters belonging to Mexico and Cuba where there is currently little information on cetacean species abundance and distribution. U.S. waters only comprise about 40% of the entire Gulf of Mexico, and 65% of oceanic waters are south of the U.S. Exclusive Economic Zone (EEZ).

Nara et al. (2017) analyzed mitochondrial DNA sequence data from Clymene dolphin samples collected in the western North Atlantic, Gulf of Mexico, and western South Atlantic and found significant genetic differentiation among all three regions, supporting delimitation of separate western North Atlantic and Gulf of Mexico stocks. The Gulf of Mexico population is provisionally being considered a separate stock for management purposes, although there is currently no information to differentiate this stock from the Atlantic Ocean stock(s). There are insufficient data to determine whether the northern Gulf of Mexico stock comprises multiple demographically independent populations. Additional morphological, acoustic, genetic and/or behavioral data are needed to provide further delineate population structure within the Gulf of Mexico and across the broader geographic area of information on stock delineation.

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

POPULATION SIZE

The best abundance estimate (Nbest) available for northern Gulf of Mexico Clymene dolphins is 513,429 (CV=1.03; Table 1). This estimate is from summer 2017 and summer/fall 2018 oceanic surveys covering waters from the 200-m isobath to the seaward extent of the U.S. EEZ (Garrison et al. 2020). This estimate is from a summer 2009 oceanic survey covering waters from the 200m isobath to the seaward extent of the U.S. EEZ.

Earlier abundance estimates

Please see Appendix IV for a summary of abundance estimates, including earlier estimates and survey descriptions. All estimates of abundance were derived through the application of distance sampling analysis (Buckland et al. 2001) and the computer program DISTANCE (Thomas et al. 1998) to line-transect survey data collected from ships in the oceanic northern Gulf of Mexico (i.e., 200m isobath to seaward extent of the U.S. EEZ) and are summarized in Appendix IV.

—From 1991 through 1994, and from 1996 through 2001 (excluding 1998), annual surveys were conducted during spring along a fixed plankton-sampling trackline. Due to limited survey effort in any given year, the survey effort-weighted estimated average abundance of Clymene dolphins for all surveys combined was estimated. For 1991 to 1994, the estimate was 5,871 (CV=0.37) (Hansen et al. 1995), and for 1996 to 2001, 17,355 (CV=0.65) (Mullin and Fulling 2004; Table 1).

—During summer 2003 and spring 2004, surveys dedicated to estimating cetacean abundance were conducted along a grid of uniformly spaced transect lines from a random start. The abundance estimate for Clymene dolphins, pooled from 2003 to 2004, was 6,575 (CV=0.36) (Mullin 2007; Table 1).

Recent surveys and abundance estimates

—During summer 2009, a line-transect survey dedicated to estimating the abundance of oceanic cetaceans was
conducted in the northern Gulf of Mexico using NOAA Ship Gordon Gunter. Survey lines were stratified in relation to depth and the location of the Loop Current. The abundance estimate for Clymene dolphins in oceanic waters during 2009 was 129 (CV=1.00; Table 1). An abundance estimate for Clymene dolphins was generated from vessel surveys conducted in the northern Gulf of Mexico from the continental shelf edge (~200-m isobath) to the seaward extent of the U.S. EEZ (Garrison et al. 2020). One survey was conducted from 2 July to 25 August 2017 and consisted of 7,302 km of on-effort trackline, and the second survey was conducted from 11 August to 6 October 2018 and consisted of 6,473 km of on-effort trackline within the surveyed strata. Both surveys used a double-platform data-collection procedure, to allow estimation of the detection probability on the trackline using the independent observer approach assuming point independence (Laake and Borchers 2004). However, this species was observed during tracklines when only one survey team was on effort. Therefore, abundance estimates were derived using MCDS distance sampling methods that accounted for the effects of covariates (e.g., sea state, glare) on detection probability within the surveyed strip (Thomas et al. 2010) implemented in package mrds (version 2.21, Laake et al. 2020) in the R statistical programming language. The estimated detection probability on the trackline for similar species was then applied to develop the final abundance estimate. The abundance estimate for this stock included sightings of unidentified dolphins that were apportioned among species based on their relative density within the survey strata (Garrison et al. 2020). The surveys were conducted in "passing mode" (e.g., Schwarz et al. 2010) while all prior surveys in the Gulf of Mexico have been conducted in "closing mode." There were sightings of Clymene dolphins in 2017 but there were none in 2018. The inverse variance weighted mean abundance estimate for Clymene dolphins in oceanic waters during 2017 and 2018 was 513 (CV=1.03; Table 1; Garrison et al. 2020). Unlike previous abundance estimates, this estimate was corrected for the probability of detection on the trackline.

Table 1. Most recent abundance estimate (Nbest) and coefficient of variation (CV) of northern Gulf of Mexico Clymene dolphins in oceanic waters (200 m to the offshore extent of the EEZ) based on an average from summer 2017 and summer/fall 2018 vessel surveys.

<table>
<thead>
<tr>
<th>Years</th>
<th>Area</th>
<th>Nbest</th>
<th>CV</th>
</tr>
</thead>
<tbody>
<tr>
<td>2017, 2018</td>
<td>Gulf of Mexico</td>
<td>513</td>
<td>1.03</td>
</tr>
</tbody>
</table>

Table 1. Summary of abundance estimates for northern Gulf of Mexico Clymene dolphins. Month, year and area covered during each abundance survey, and resulting abundance estimate (Nbest) and coefficient of variation (CV).

<table>
<thead>
<tr>
<th>Month/Year</th>
<th>Area</th>
<th>Nbest</th>
<th>CV</th>
</tr>
</thead>
<tbody>
<tr>
<td>Apr-Jun 1991-1994</td>
<td>Oceanic waters</td>
<td>5,571</td>
<td>0.37</td>
</tr>
<tr>
<td>Apr-Jun 1996-2001 (excluding 1998)</td>
<td>Oceanic waters</td>
<td>17,355</td>
<td>0.65</td>
</tr>
<tr>
<td>Jun-Aug 2003, Apr-Jun 2004</td>
<td>Oceanic waters</td>
<td>6,575</td>
<td>0.36</td>
</tr>
<tr>
<td>Jun-Aug 2009</td>
<td>Oceanic waters</td>
<td>129</td>
<td>1.00</td>
</tr>
</tbody>
</table>

Minimum Population Estimate

The minimum population estimate (Nmin) is the lower limit of the two-tailed 60% confidence interval of the log-normal distributed abundance estimate. This is equivalent to the 20th percentile of the log-normal distributed abundance estimate as specified by Wade and Angliss (1997). The best estimate of abundance for Clymene dolphins...
is 513.429 (CV=1.034.00). The minimum population estimate for the northern Gulf of Mexico stock of Clymene dolphins is 250 (Table 2).64 Clymene dolphins.

Current Population Trend

The statistical power to detect a trend in abundance for this stock is poor due to the relatively imprecise abundance estimates and long intervals between surveys. For example, the power to detect a precipitous decline in abundance (i.e., 50% decrease in 15 years) with estimates of low precision (e.g., CV>0.30) remains below 80% (alpha=0.30) unless surveys are conducted on an annual basis (Taylor et al. 2007). In addition, because these surveys are restricted to U.S. waters, it is not possible to distinguish between changes in population size and Gulf-wide shifts in spatial distribution. Five point estimates of Clymene dolphin abundance have been made based on data from surveys during: 2003 (June–August), 2004 (April–June), 2009 (July–August), 2017 (July–August), and 2018 (August–October). Each of these surveys had a similar design and was conducted using the same vessel or a vessel with a similar observation platform. Prior year surveys only employed a single survey team. The estimates for 2017 and 2018 were conducted in "passing" mode which resulted in increased numbers of unidentified sightings and may have affected group size estimation for distant groups of dolphins and small whales. Apportioning unidentified species among identified taxa addresses the first issue, and there were no apparent relationships between sighting distance and estimated group size. The model for detection probability on the trackline from the summer 2017 survey was applied to the abundance estimates from the 2003, 2004, and 2009 surveys. The resulting abundance estimates were: 2003, N=10,900 (CV=0.42); 2004, N=13,257 (CV=0.81); 2009, N=1,319 (CV=0.78); 2017, N=1,026 (CV=1.03); and 2018, N=0 (CV=NA). Pairwise comparisons of the non-zero log-transformed means were conducted between years, and significant differences were assessed at alpha=0.10. P-values were adjusted for multiple comparisons. Pairwise comparisons indicated significant differences between the 2003 estimates and both the 2009 (p.adjusted=0.024) and 2017 (p.adjusted=0.030), and between the 2004 estimates and both the 2009 (p.adjusted=0.039) and 2017 (p.adjusted=0.039), and there were no Clymene dolphins observed during 2018. Taken together, these differences suggest a possible decline in abundance during recent years. Four point estimates of Clymene dolphin abundance have been made based on data from surveys covering 1991–2009. The estimates vary by a maximum factor of more than 100. To determine whether changes in abundance have occurred over this period, an analysis of all the survey data needs to be conducted which incorporates covariates (e.g., survey conditions, season) that could potentially affect estimates. Nevertheless, differences in temporal abundance estimates will still be difficult to interpret without a Gulf of Mexico-wide understanding of Clymene dolphin abundance. The oceanography of the Gulf of Mexico is quite dynamic, and the spatial scale of the Gulf is small relative to the ability of most cetacean species to travel. Studies based on abundance and distribution surveys restricted to U.S. waters are unable to detect temporal shifts in distribution beyond U.S. waters that might account for any changes in abundance.

Current and Maximum Net Productivity Rates

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

POTENTIAL BIOLOGICAL REMOVAL

Potential Biological Removal (PBR) is the product of the minimum population size, one half the maximum net productivity rate and a recovery factor (MMPA Sec. 3.16 U.S.C. 1362; Wade and Angliss 1997). The minimum population size is 250. The maximum productivity rate is 0.04, the default value for cetaceans. The recovery factor, which accounts for endangered, depleted, threatened stocks, or stocks of unknown status relative to optimum sustainable population (OSP), is assumed to be 0.5 because the stock is of unknown status. PBR for the northern Gulf of Mexico Clymene dolphin is 2.5 (Table 2).64.

Table 2. Best and minimum abundance estimates for northern Gulf of Mexico Clymene dolphins with Maximum Productivity Rate (Rmax), Recovery Factor (Fr) and PBR.

<table>
<thead>
<tr>
<th>Nb</th>
<th>CV</th>
<th>N</th>
<th>Fr</th>
<th>Rmax</th>
<th>PBR</th>
</tr>
</thead>
<tbody>
<tr>
<td>513</td>
<td>1.03</td>
<td>250</td>
<td>0.5</td>
<td>0.04</td>
<td>2.5</td>
</tr>
</tbody>
</table>

ANNUAL HUMAN-CAUSED MORTALITY AND SERIOUS INJURY
Total annual estimated fishery-related mortality and serious injury to this stock during 2014–2018 was presumed to be zero, as there were no reports of mortalities or serious injuries to Clymene dolphins in the Gulf of Mexico (Table 3). Mean annual mortality and serious injury during 2014–2018 due to other human-caused actions (the Deepwater Horizon oil spill) was predicted to be 8.4. The minimum total mean annual human-caused mortality and serious injury for this stock during 2014–2018 was, therefore, 8.4. There has been no reported fishing-related mortality or serious injury of Clymene dolphins during 1998–2010 (Yeung 1999; Yeung 2001; Garrison 2003; Garrison and Richards 2004; Garrison 2005; Fairfield Walsh and Garrison 2006; Fairfield-Walsh and Garrison 2007; Fairfield and Garrison 2008; Garrison et al. 2009; Garrison and Stokes 2010; 2011).

Table 3. Total annual estimated fishery-related mortality and serious injury for northern Gulf of Mexico Clymene dolphins.

<table>
<thead>
<tr>
<th>Years</th>
<th>Source</th>
<th>Annual Avg.</th>
<th>CV</th>
</tr>
</thead>
<tbody>
<tr>
<td>2014–2018</td>
<td>U.S. fisheries using observer data</td>
<td>0</td>
<td>-</td>
</tr>
</tbody>
</table>

Fisheries Information

There are two commercial fisheries that interact, or that could potentially could interact, with this stock in the Gulf of Mexico. These are the Category I Atlantic Highly Migratory Species (high seas) longline fishery and the Category I Atlantic Ocean, Caribbean, Gulf of Mexico large pelagics longline fishery (Appendix III). Percent observer coverage (percentage of sets observed) for these longline fisheries for each year during 2014–2018 was 18, 19, 23, 13 and 20, respectively. There is very little effort within the Gulf of Mexico by the Atlantic Highly Migratory Species (high seas) longline fishery, and no takes of Clymene dolphins within high-seas waters of the Gulf of Mexico have been observed or reported thus far. Pelagic swordfish, tunas and billfish are the targets of the large pelagics longline fishery operating in the northern Gulf of Mexico and there were no observed mortalities or serious injuries to Clymene dolphins by this fishery during 2014–2018 (Garrison and Stokes 2016; 2017; 2019; 2020; in review). The commercial fishery which potentially could interact with this stock in the Gulf of Mexico is the Atlantic Ocean, Caribbean, Gulf of Mexico large pelagic longline fishery (Appendix III). Pelagic swordfish, tunas and billfish are the targets of the longline fishery operating in the northern Gulf of Mexico. There were no reports of mortality or serious injury to Clymene dolphins by this fishery during 1998–2010 (Yeung 1999; Yeung 2001; Garrison 2003; Garrison and Richards 2004; Garrison 2005; Fairfield Walsh and Garrison 2006; Fairfield-Walsh and Garrison 2007; Fairfield and Garrison 2008; Garrison et al. 2009; Garrison and Stokes 2010; 2011).

Other Mortality

There were 16 reported strandings of Clymene dolphins in the Gulf of Mexico during 2006–2014–2018 (NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 16 November 2019; Table 4). For one stranding, there was evidence of human interaction (healed scars). No evidence of human interaction was detected for two strandings, and for the remaining 13 strandings, it could not be determined whether there was evidence of human interaction. Stranding data probably underestimate the extent of human and fishery-related mortality and serious injury because not all of the dolphins that die or are seriously injured in human interactions wash ashore, or, if they do, they are not all recovered (Peltier et al. 2012; Wells et al. 2015). In particular, oceanic stocks in the Gulf of Mexico are less likely to strand than nearshore coastal stocks or shelf stocks (Williams et al. 2011). Additionally, not all carcasses will show evidence of human interaction, entanglement or other fishery-related interaction due to decomposition, scavenger damage, etc. (Byrd et al. 2014). Finally, the level of technical expertise among stranding network personnel varies widely as does the ability to recognize signs of human interaction. Stranding data probably underestimate the extent of fishery-related mortality and serious injury because not all of the marine mammals which die or are seriously injured in fishery interactions wash ashore, not all that wash ashore are discovered, reported or investigated, 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 interactions.

An Unusual Mortality Event (UME) was declared for cetaceans in the northern Gulf of Mexico beginning 1 February/March 2010; and ending 31 July 2014 (Litz et al. 2014; https://www.fisheries.noaa.gov/national/marine-life-distress/2010-2014-cetacean-unusual-mortality-event-northern-gulf-mexico), as of early 2012, the event is still ongoing. It includes cetaceans that stranded prior to the Deepwater Horizon (DWH) oil spill (see “Habitat Issues”
below), during the spill, and after. Exposure to the DWH oil spill was determined to be the primary underlying cause of the elevated stranding numbers in the northern Gulf of Mexico after the spill (e.g., Schwacke et al. 2014; Venn-Watson et al. 2015; Colegrove et al. 2016; DWH NRDAT 2016; see Habitat Issues section). One stranding of a Clymene dolphin in 2010 was considered to be part of this UME. During 2010, no animals from this stock were considered to be part of the UME.

A population model was developed to estimate the injury and time to recovery for stocks affected by the DWH oil spill, taking into account long-term effects resulting from mortality, reproductive failure, reduced survival rates, and the proportion of the stock exposed to DWH oil (DWH MMIQT 2015). Overall, the model estimated that this stock experienced a 3% maximum reduction in population size due to the oil spill (DWH MMIQT 2015). The mortality projected for the years 2010–2013 due to the spill has not been reported previously. Based on the population model, it was projected that 83 Clymene dolphins died during 2010–2013 (four year annual average of 21) due to elevated mortality associated with oil exposure (see Appendix VI). For the 2014–2018 reporting period of this SAR, the population model estimated 42 Clymene dolphins died due to elevated mortality associated with oil exposure. The population model used to predict Clymene dolphin mortality due to the DWH event has a number of sources of uncertainty. Model parameters (e.g., survival rates, reproductive rates, and life-history parameters) were derived from literature sources for Clymene dolphins occupying waters outside of the Gulf of Mexico. In addition, proxy values for the effects of DWH oil exposure on both survival rates and reproductive success were applied based upon estimated values for common bottlenose dolphins in Barataria Bay. Finally, there was no estimation of uncertainty in model parameters or outputs.


<table>
<thead>
<tr>
<th>Area</th>
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<th>2016</th>
<th>2017</th>
<th>2018</th>
<th>Total</th>
</tr>
</thead>
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<td>13</td>
<td>1</td>
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</tr>
<tr>
<td>Texas</td>
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<td>0</td>
<td>0</td>
<td>1</td>
<td>0</td>
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</tr>
<tr>
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<td>1</td>
<td>14</td>
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<td>16</td>
</tr>
</tbody>
</table>

HABITAT ISSUES

The Deepwater Horizon (DWH) MC252 drilling platform, located approximately 50 miles 80 km southeast of the Mississippi River Delta in waters about 1,500 m deep, exploded on 20 April 2010. The rig sank, and for over 87 days ~3.2 millions of barrels of oil and gas were discharged from the wellhead until it was capped on 15 July 2010 (DWH NRDAT 2016). During the response effort dispersants were applied extensively at the seafloor and at the sea surface (Lehr et al. 2010; OSAT 2010). In-situ burning, or controlled burning of oil at the surface, was also used extensively as a response tool (Lehr et al. 2010). The oil, dispersant and burn residue compounds present ecological concerns. The magnitude of this oil spill was unprecedented in U.S. history, causing impacts to wildlife, natural habitats and human communities along coastal areas from western Louisiana to the Florida Panhandle (NOAA 2011). It could be years before the entire scope of damage is ascertained (NOAA 2011).

Shortly after the oil spill, the Natural Resource Damage Assessment (NRDA) process was initiated under the Oil Pollution Act of 1990. A variety of NRDA research studies were conducted to determine potential impacts of the spill on marine mammals. These studies estimated that 7% (95% CI: 3–15) of Clymene dolphins in the Gulf were exposed to oil, that 3% (95% CI: 2–5) of females suffered from reproductive failure, and 3% (95% CI: 1–4) of Clymene dolphins suffered adverse health effects (DWH MMIQT 2015). A population model estimated that the stock experienced a 3% maximum reduction in population size (see Other Mortality section above).

Anthropogenic sound in the world’s oceans has been shown to affect marine mammals, with vessel traffic, seismic surveys, and active naval sonars being the main anthropogenic contributors to low- and mid-frequency noise in oceanic waters (e.g., Nowacek et al. 2015; Gomez et al. 2016; NMFS 2018). The long-term and population consequences of these impacts are less well-documented and likely vary by species and other factors. Impacts on marine mammal prey from sound are also possible (Carroll et al. 2017), but the duration and severity of any such prey effects on marine mammals are unknown. These studies have focused on identifying the type, magnitude, severity, length and impact of oil exposure to oceanic, coastal and estuarine marine mammals. The research is ongoing and likely will continue for
some time. For continental shelf and oceanic cetaceans, the NOAA-led efforts include: aerial surveys to document the
distribution, abundance, species and exposure of marine mammals and turtles relative to oil from DWH spill; and ship
surveys to evaluate exposure to oil and other chemicals and to assess changes in animal behavior and distribution
relative to oil exposure through visual and acoustic surveys, deployment of passive acoustic monitoring systems,
collection of tissue samples, and deployment of satellite tags on sperm and Bryde's whales.

Aerial surveys have observed Risso’s dolphins, spinner dolphins, pantropical spotted dolphins, striped dolphins,
bottlenose dolphins and sperm whales swimming in oil in offshore waters (NOAA 2010a). The effects of oil exposure
on marine mammals depend on a number of factors including the type and mixture of chemicals involved, the amount,
frequency and duration of exposure, the route of exposure (inhaled, ingested, absorbed, or external) and biomedical
risk factors of the particular animal (Geraci 1990; NOAA 2010b). In general, direct external contact with petroleum
compounds or dispersants with skin may cause skin irritation, chemical burns and infections. Inhalation of volatile
petroleum compounds or dispersants may irritate or injure the respiratory tract, which could lead to pneumonia or
inflammation. Ingestion of petroleum compounds may cause injury to the gastrointestinal tract, which could affect an
animal’s ability to digest or absorb food. Absorption of petroleum compounds or dispersants may damage kidney,
liver and brain function in addition to causing immune suppression and anemia. Long-term chronic effects such as
lowered reproductive success and decreased survival may occur (Geraci 1990; NOAA 2010b).

STATUS OF STOCK

Clymene dolphins are not listed as threatened or endangered under the Endangered Species Act, but the northern
Gulf of Mexico stock is considered strategic under the MMPA because the mean modeled annual human-caused
mortality and serious injury due to the DWH oil spill exceeds PBR. No fishery-related mortality or serious injury has
been observed; therefore, total fishery-related mortality and serious injury can be considered insignificant and
approaching the zero mortality and serious injury rate. The status of Clymene dolphins in the U.S. EEZ relative to
OSP is unknown. There were statistically significant decreases in population size in recent years for this stock in the
northern Gulf of Mexico. The status of Clymene dolphins in the northern Gulf of Mexico, relative to OSP, is unknown.
The species is not listed as threatened or endangered under the Endangered Species Act. There are insufficient data to
determine the population trends for this species. Total human caused mortality and serious injury for this stock is not
known but none has been documented. The total level of fishery-related mortality and serious injury for this stock is
unknown, but assumed to be less than 10% of the calculated PBR and can be considered to be insignificant and
approaching zero mortality and serious injury rate. This is not a strategic stock because average annual human-related
mortality and serious injury does not exceed PBR.

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Garrison, L.P. 2003. Estimated bycatch of marine mammals and turtles in the U.S. Atlantic pelagic longline fleet
Garrison, L.P. 2005. Estimated bycatch of marine mammals and turtles in the U.S. Atlantic pelagic longline fleet


FRASER'S DOLPHIN (*Lagenodelphis hosei*):
Northern Gulf of Mexico Stock

STOCK DEFINITION AND GEOGRAPHIC RANGE

Fraser's dolphins are distributed worldwide in tropical waters (Perrin *et al.* 1994), and they have more recently been reported from temperate and subtropical areas of the North Atlantic (Gomes-Pereira *et al.* 2013). They are generally oceanic in distribution but may be seen closer to shore where deep water can be found near the shore, such as in the Lesser Antilles of the Caribbean Sea (Dolar 2009). Sightings in the northern Gulf of Mexico (i.e., U.S. Gulf of Mexico) occur only sporadically during vessel surveys in the northern Gulf of Mexico (i.e., U.S. Gulf of Mexico) and are generally confined to oceanic waters (>200 m) (Figure 1; Maze-Foley and Mullin 2006) and they. Fraser's dolphins have been observed in the northern Gulf of Mexico during all seasons (Leatherwood *et al.* 1993; Hansen *et al.* 1996; Mullin and Hoggard 2000).

All the cetacean species found in the oceanic northern Gulf of Mexico almost certainly occur in similar habitat beyond U.S. boundaries in the southern Gulf. There are fewer cetacean sighting and stranding records in the southern Gulf due to more limited effort. Nevertheless there are records for most oceanic species in the southern Gulf (e.g., Jefferson and Schiro 1997; Ortega Ortiz 2002; Ortega-Argueta *et al.* 2005; Jefferson *et al.* 2008; Vázquez Castán *et al.* 2009; Whitt *et al.* 2011). Although there are only a few records from Gulf of Mexico waters beyond U.S. boundaries (e.g., Jefferson and Schiro 1997, Ortega Ortiz 2002), Fraser’s dolphins almost certainly occur throughout the oceanic Gulf of Mexico (Jefferson *et al.* 2008). This is therefore likely a transboundary stock with Cuba and/or Mexico. Because U.S. waters only comprise about 40% of the entire Gulf of Mexico and 35% of the oceanic (i.e., >200 m) Gulf of Mexico (Mullin and Fulling 2004), abundance and stock boundaries of oceanic species are poorly known, which is also composed of waters belonging to Mexico and Cuba where there is currently little information on cetacean species abundance and distribution. U.S. waters only comprise about 40% of the entire Gulf of Mexico, and 65% of oceanic waters are south of the U.S. Exclusive Economic Zone (EEZ).

Fraser’s dolphins in the northern Gulf of Mexico are managed separately from those in the western North Atlantic. Although there have been no directed studies of the degree of demographic independence between the two areas, this management structure is consistent with the fact that the Gulf of Mexico and western North Atlantic belong to distinct...
marine ecoregions (Spalding et al. 2007; Moore and Merrick 2011). Due to the paucity of sightings in the northern Gulf of Mexico, there are insufficient data to determine whether the northern Gulf of Mexico stock comprises multiple demographically independent populations. Additional morphological, acoustic, genetic, and/or behavioral data are needed to further delineate population structure within the Gulf of Mexico and across the broader geographic area. The Gulf of Mexico population is provisionally being considered 1 stock for management purposes, although there is currently no information to differentiate this stock from the Atlantic Ocean stock(s). Additional morphological, genetic and/or behavioral data are needed to provide further information on stock delineation.

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

POPULATION SIZE

The best abundance estimate (Nbest) available for northern Gulf of Mexico Fraser’s dolphins is 213 (CV=1.03; Table 1) unknown (Table 1). This estimate is from summer 2017 and summer/fall 2018 oceanic surveys covering waters from the 200-m isobath to the seaward extent of the U.S. EEZ (Garrison et al. 2020). No sightings of groups of Fraser’s dolphins were made during a summer 2009 survey. Nevertheless, a small number of Fraser’s dolphins probably continually inhabit the northern Gulf of Mexico. Historically, sightings have been consistently made every 3-4 years since the early 1990’s but have not occurred or have been rare during any given survey.

Earlier abundance estimates

Please see Appendix IV for a summary of abundance estimates, including earlier estimates and survey descriptions. All estimates of abundance were derived through the application of distance sampling analysis (Buckland et al. 2001) and the computer program DISTANCE (Thomas et al. 1998) to line transect survey data collected from ships in the oceanic northern Gulf of Mexico (i.e., 200m isobath to seaward extent of the U.S. EEZ) and are summarized in Appendix IV.

— From 1991 through 1994, and from 1996 through 2001 (excluding 1998), annual surveys were conducted during spring along a fixed plankton-sampling trackline. Due to limited survey effort in any given year, the survey effort-weighted estimated average abundance of Fraser’s dolphins for all surveys combined was estimated. For 1991 to 1994, the estimate was 127 (CV=0.90) (Hansen et al. 1995), and for 1996 to 2001, 726 (CV=0.70) (Mullin and Fulling 2004; Table 1).
— During summer 2003 and spring 2004, surveys dedicated to estimating cetacean abundance were conducted along a grid of uniformly-spaced transect lines from a random start. The abundance estimate for Fraser’s dolphins, pooled from 2003 to 2004, was 0 (Mullin 2007; Table 1).

Recent surveys and abundance estimates

An abundance estimate for Fraser’s dolphins was generated from vessel surveys conducted in the northern Gulf of Mexico from the continental shelf edge (~200-m isobath) to the seaward extent of the U.S. EEZ (Garrison et al. 2020). One survey was conducted from 2 July to 25 August 2017 and consisted of 7,302 km of on-effort trackline, and the second survey was conducted from 11 August to 6 October 2018 and consisted of 6,473 km of on-effort trackline within the surveyed strata. Both surveys used a double-platform data-collection procedure to allow estimation of the detection probability on the trackline using the independent observer approach assuming point independence (Laake and Borchers 2004). However, this species was observed during tracklines when only one survey team was on effort. Therefore, abundance estimates were derived using MCDS distance sampling methods that accounted for the effects of covariates (e.g., sea state, glare) on detection probability within the surveyed strip (Thomas et al. 2010) implemented in package mrds (version 2.21, Laake et al. 2020) in the R statistical programming language. The estimated detection probability on the trackline for similar species was then applied to develop the final abundance estimate. The abundance estimate for this stock included sightings of unidentified dolphins that were apportioned
among species based on their relative density within the survey strata (Garrison et al. 2020). The surveys were conducted in "passing mode" (e.g., Schwarz et al. 2010) while all prior surveys in the Gulf of Mexico have been conducted in "closing mode." There were sightings of Fraser’s dolphins in 2017 but there were none in 2018. The inverse variance weighted mean abundance for Fraser’s dolphins in oceanic waters during 2017 and 2018 was 213 (CV=1.03; Table 1; Garrison et al. 2020). Unlike previous abundance estimates, this estimate was corrected for the probability of detection on the trackline. During summer 2009, a line transect survey dedicated to estimating the abundance of oceanic cetaceans was conducted in the northern Gulf of Mexico. Survey lines were stratified in relation to depth and the location of the Loop Current. The abundance estimate for Fraser’s dolphins in oceanic waters during 2009 was 0 (Table 1). Because sightings of groups of Fraser’s dolphins have historically been uncommon to rare, it is probable that Fraser’s dolphins were in the northern Gulf of Mexico during 2009 but were not encountered.

Table 1. Most recent abundance estimate ($N_{best}$) and coefficient of variation (CV) of northern Gulf of Mexico Fraser’s dolphins in oceanic waters (200 m to the offshore extent of the EEZ) based on an average from summer 2017 and summer/fall 2018 vessel surveys.

<table>
<thead>
<tr>
<th>Years</th>
<th>Area</th>
<th>$N_{best}$</th>
<th>CV</th>
</tr>
</thead>
<tbody>
<tr>
<td>2017, 2018</td>
<td>Gulf of Mexico</td>
<td>213</td>
<td>1.03</td>
</tr>
</tbody>
</table>

Table 1. Summary of abundance estimates for northern Gulf of Mexico Fraser’s dolphins. Month, year and area covered during each abundance survey, and resulting abundance estimate ($N_{best}$) and coefficient of variation (CV).

<table>
<thead>
<tr>
<th>Month/Year</th>
<th>Area</th>
<th>$N_{best}$</th>
<th>CV</th>
</tr>
</thead>
<tbody>
<tr>
<td>Apr-Jun 1991-1994</td>
<td>Oceanic waters</td>
<td>127</td>
<td>0.90</td>
</tr>
<tr>
<td>Apr-Jun 1996-2001 (excluding 1998)</td>
<td>Oceanic waters</td>
<td>726</td>
<td>0.70</td>
</tr>
<tr>
<td>Jun-Aug 2003, Apr-Jun 2004</td>
<td>Oceanic waters</td>
<td>0</td>
<td>-</td>
</tr>
<tr>
<td>Jun-Aug 2009</td>
<td>Oceanic waters</td>
<td>0</td>
<td>-</td>
</tr>
</tbody>
</table>

Minimum Population Estimate

The minimum population estimate ($N_{min}$) is the lower limit of the two-tailed 60% confidence interval of the log-normal distributed abundance estimate. This is equivalent to the 20th percentile of the log-normal distributed abundance estimate as specified by Wade and Angliss (1997). The best estimate of abundance for Fraser’s dolphins is 213 (CV=1.03) unknown. The minimum population estimate for the northern Gulf of Mexico for Fraser’s dolphins is 104 (Table 2) unknown.

Current Population Trend

There are insufficient data to determine the population trends for this species. The best available abundance estimate is unknown. The pooled abundance estimate for 1996-2001 of 726 (CV=0.70) and that for 1991-1994 of 127 (CV=0.89) were not significantly different (P>0.05), but due to the precision of the estimates, the power to detect a difference is low. No estimates of abundance were available from surveys conducted during 2003, 2004, and 2009 that are comparable to the most recent surveys because no Fraser’s dolphins were observed. The large relative
changes/fluctuations in the total abundances of Fraser’s dolphins are probably due to a number of factors. Fraser’s dolphin is most certainly a resident species in the Gulf of Mexico but probably occurs in low numbers and the survey effort is not sufficient to estimate the abundance of uncommon or rare species with precision. Also, because this is likely a transboundary stock, the temporal changes in abundance estimates are difficult to interpret without a Gulf of Mexico-wide understanding of Fraser’s dolphin distribution and abundance. Also, these temporal abundance estimates are difficult to interpret without a Gulf of Mexico-wide understanding of Fraser’s dolphin abundance. Fraser’s dolphin, like all the other oceanic cetacean species in the Gulf, is a mobile predator and this stock is most likely a transboundary stock. The oceanography of the Gulf of Mexico is quite dynamic, and the spatial scale of the Gulf is small relative to the ability of most cetacean species to travel. Studies based on abundance and distribution surveys restricted to U.S. waters are unable to detect temporal shifts in distribution beyond U.S. waters that might account for any changes in abundance.

CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

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

POTENTIAL BIOLOGICAL REMOVAL

Potential Biological Removal (PBR) is the product of the minimum population size, one half the maximum net productivity rate and a recovery factor (MMPA Sec. 3.16 U.S.C. 1362; Wade and Angliss 1997). The minimum population size is 104/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 the stock is of unknown status. PBR for the northern Gulf of Mexico Fraser’s dolphin is 1.0 (Table 2/unknown).

Table 2. Best and minimum abundance estimates for northern Gulf of Mexico Fraser’s dolphins with Maximum Productivity Rate (Rmax), Recovery Factor (Fr) and PBR.

<table>
<thead>
<tr>
<th>Nbest</th>
<th>CV</th>
<th>Nmin</th>
<th>Fr</th>
<th>Rmax</th>
<th>PBR</th>
</tr>
</thead>
<tbody>
<tr>
<td>213</td>
<td>1.03</td>
<td>104</td>
<td>0.5</td>
<td>0.04</td>
<td>1.0</td>
</tr>
</tbody>
</table>

ANNUAL HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

Total annual estimated fishery-related mortality and serious injury to this stock during 2014–2018 was presumed to be zero, as there were no reports of mortalities or serious injuries to Fraser’s dolphins in the Gulf of Mexico (Table 3). Mean annual mortality and serious injury during 2014–2018 due to other human-caused actions (e.g., the Deepwater Horizon oil spill) was unknown (see Habitat Issues section). The minimum total mean annual human-caused mortality and serious injury for this stock during 2014–2018 was, therefore, unknown. There has been no reported fishing-related mortality of a Fraser’s dolphin during 1998-2010 (Yeung 1999; Yeung 2001; Garrison 2003; Garrison and Richards 2004; Garrison 2005; Fairfield Walsh and Garrison 2006; Fairfield-Walsh and Garrison 2007; Fairfield and Garrison 2008; Garrison et al. 2009; Garrison and Stokes 2010; 2011).

Table 3. Total annual estimated fishery-related mortality and serious injury for northern Gulf of Mexico Fraser’s dolphins.

<table>
<thead>
<tr>
<th>Years</th>
<th>Source</th>
<th>Annual Avg.</th>
<th>CV</th>
</tr>
</thead>
<tbody>
<tr>
<td>2014–2018</td>
<td>U.S. fisheries using observer data</td>
<td>0</td>
<td></td>
</tr>
</tbody>
</table>

Fisheries Information

There are two commercial fisheries that interact, or that could potentially interact, with this stock in the Gulf of Mexico. These are the Category I Atlantic Highly Migratory Species (high seas) longline fishery and the Category I Atlantic Ocean, Caribbean, Gulf of Mexico large pelagics longline fishery (Appendix III). The commercial fishery which potentially could interact with this stock in the Gulf of Mexico is the Atlantic Ocean, Caribbean, Gulf of Mexico

400
Surveys, and active naval sonars being the main anthropogenic contributors to low- and mid-frequency noise in oceanic habitats and human communities along coastal areas from western Louisiana to the Florida Panhandle (NOAA 2011). The magnitude of this oil spill was unprecedented in U.S. history, causing impacts to wildlife, natural habitats and human communities along coastal areas from western Louisiana to the Florida Panhandle (NOAA 2011). It could be years before the entire scope of damage is ascertained (NOAA 2011).

Shortly after the oil spill, the Natural Resource Damage Assessment (NRDA) process was initiated under the Oil Pollution Act of 1990. A variety of NRDA research studies are being conducted to determine potential impacts of the spill on marine mammals. These studies did not include Fraser’s dolphins regarding impacts of the spill due to insufficient data to determine the overlap of the DWH oil spill footprint and the range of Fraser’s dolphins (DWH MMIQT 2015). The impact of the spill on Fraser’s dolphins is unknown.

Anthropogenic sound in the world’s oceans has been shown to affect marine mammals, with vessel traffic, seismic surveys, and active naval sonars being the main anthropogenic contributors to low- and mid-frequency noise in oceanic habitats and human communities along coastal areas from western Louisiana to the Florida Panhandle (NOAA 2011). It could be years before the entire scope of damage is ascertained (NOAA 2011).

Other Mortality

There was one mass stranding of five were no reported strandings of Fraser’s dolphins in the Gulf of Mexico during 2006–2010 (NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 21 May 2019; 16 November 2011). The mass stranding occurred off Florida in July 2017, and it could not be determined if there was evidence of human interaction for any of the dolphins. Stranding data probably underestimate the extent of human and fishery-related mortality and serious injury because not all of the whales that die or are seriously injured in human interactions wash ashore, or, if they do, they are not all recovered (Peltier et al. 2012; Wells et al. 2015). In particular, oceanic stocks in the Gulf of Mexico are less likely to strand than nearshore coastal stocks or shelf stocks (Williams et al. 2011). Additionally, not all carcasses will show evidence of human interaction, entanglement or other fishery-related interaction due to decomposition, scavenger damage, etc. (Byrd et al. 2014). Finally, the level of technical expertise among stranding network personnel varies widely as does the ability to recognize signs of human interaction. Stranding data probably underestimate the extent of fishery-related mortality and serious injury because not all of the marine mammals which die or are seriously injured in fishery interactions wash ashore, not all that wash ashore are discovered, reported or investigated, 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 interactions.

An Unusual Mortality Event (UME) was declared for cetaceans in the northern Gulf of Mexico beginning 1 February–March 2010 and ending 31 July 2014 (Litz et al. 2014; https://www.fisheries.noaa.gov/national/marine-life-distress/2010-2014-cetacean-unusual-mortality-event-northern-gulf-mexico) and, as of early 2012, the event is still ongoing. It includes cetaceans that stranded prior to the Deepwater Horizon (DWH) oil spill (see “Habitat Issues” below), during the spill, and after. Exposure to the DWH oil spill was determined to be the primary underlying cause of the elevated stranding numbers in the northern Gulf of Mexico after the spill (e.g., Schwacke et al. 2014; Venn-Watson et al. 2015; Colegrove et al. 2016; DWH NRDAT 2016; see Habitat Issues section). However, there were no Fraser’s dolphin strandings recovered within the spatial and temporal boundaries of this UME. During 2010, no animals from this stock were considered to be part of the UME.

HABITAT ISSUES

The Deepwater Horizon (DWH) MC252 drilling platform, located approximately 80 km southeast of the Mississippi River Delta in waters about 1,500 m deep, exploded on 20 April 2010. The rig sank, and over 87 days ~3.2 million barrels of oil and gas were discharged from the wellhead until it was capped on 15 July 2010 (DWH NRDAT 2016). During the response effort dispersants were applied extensively at the seafloor and at the sea surface (Lehr et al. 2010; OSAT 2010). In situ burning, or controlled burning of oil at the surface, was also used extensively as a response tool (Lehr et al. 2010). The oil, dispersant and burn residue compounds present ecological concerns. The magnitude of this oil spill was unprecedented in U.S. history, causing impacts to wildlife, natural habitats and human communities along coastal areas from western Louisiana to the Florida Panhandle (NOAA 2011). It could be years before the entire scope of damage is ascertained (NOAA 2011).

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Aerial surveys have observed Risso’s dolphins, spinner dolphins, pantropical spotted dolphins, striped dolphins, bottlenose dolphins and sperm whales swimming in oil in offshore waters (NOAA 2010a). The effects of oil exposure on marine mammals depend on a number of factors including the type and mixture of chemicals involved, the amount, frequency and duration of exposure, the route of exposure (inhaled, ingested, absorbed, or external) and biomedical risk factors of the particular animal (Geraci 1990; NOAA 2010b). In general, direct external contact with petroleum compounds or dispersants with skin may cause skin irritation, chemical burns and infections. Inhalation of volatile petroleum compounds or dispersants may irritate or injure the respiratory tract, which could lead to pneumonia or inflammation. Ingestion of petroleum compounds may cause injury to the gastrointestinal tract, which could affect an animal’s ability to digest or absorb food. Absorption of petroleum compounds or dispersants may damage kidney, liver and brain function in addition to causing immune suppression and anemia. Long term chronic effects such as lowered reproductive success and decreased survival may occur (Geraci 1990; NOAA 2010b).

STATUS OF STOCK

Fraser’s dolphins are not listed as threatened or endangered under the Endangered Species Act, and the northern Gulf of Mexico stock is not considered strategic under the Marine Mammal Protection Act. No fishery-related mortality or serious injury has been observed in recent years; therefore, total fishery-related mortality and serious injury can be considered insignificant and approaching the zero mortality and serious injury rate. The status of Fraser’s dolphins in the northern Gulf of Mexico, relative to OSP, is unknown. The species is not listed as threatened or endangered under the Endangered Species Act. There are insufficient data to determine the population trends for this stock species. Total human-caused mortality and serious injury for this stock is not known but none has been documented. There is insufficient information available to determine whether the total fishery-related mortality and serious injury for this stock is insignificant and approaching zero mortality and serious injury rate. Despite an undetermined PBR, this is not a strategic stock because there is no documented human-related mortality and serious injury.

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KILLER WHALE (Orcinus orca): Northern Gulf of Mexico Stock

STOCK DEFINITION AND GEOGRAPHIC RANGE

The killer whale is distributed worldwide from tropical to polar regions (Leatherwood and Reeves 1983). In the northern Gulf of Mexico (i.e., U.S. Gulf of Mexico), sightings occur only sporadically during visual surveys and are generally confined to slope and basin waters >700 m (Hansen et al. 1996; O’Sullivan and Mullin 1997; Mullin and Hoggard 2000; Mullin and Fulling 2004; Maze-Foley and Mullin 2006; Garrison and Aichinger Dias 2020; Figure 1). Sightings of these animals in the northern Gulf of Mexico (i.e., U.S. Gulf of Mexico) during 1921-1995 occurred primarily in oceanic waters ranging from 256 to 2,652 m (averaging 1,242 m) in the north-central Gulf of Mexico (O’Sullivan and Mullin 1997). More recent sightings from NMFS vessel surveys have also occurred in oceanic waters of the north-central Gulf (Figure 1). Despite extensive shelf surveys (O’Sullivan and Mullin 1997), no killer whales have been reported on the Gulf of Mexico shelf waters other than those reported in 1921, 1985 and 1987 by Katona et al. (1988). Killer whales were seen only in the summer during GulfCet aerial surveys of the northern Gulf of Mexico between 1992 and 1998 (Hansen et al. 1996; Mullin and Hoggard 2000), were reported from May through June during vessel surveys (Mullin and Fulling 2004; Maze-Foley and Mullin 2006) and recorded in May, August, September and November by earlier opportunistic ship-based sources (O’Sullivan and Mullin 1997).

All the cetacean species found in the oceanic northern Gulf of Mexico almost certainly occur in similar habitat beyond U.S. boundaries in the southern Gulf. There are fewer cetacean sighting and stranding records in the southern Gulf due to more limited effort. Nevertheless there are records for most oceanic species in the southern Gulf (e.g., Jefferson and Schiro 1997; Ortega Ortiz 2002; Ortega-Argueta et al. 2005; Jefferson et al. 2008; Vázquez Castán et al. 2009; Whitt et al. 2011). This is therefore likely a transboundary stock with Cuba and/or Mexico. Because U.S. waters only comprise about 40% of the entire Gulf of Mexico and 35% of the oceanic (i.e., >200 m) Gulf of Mexico (Mullin and Fulling 2004), abundance and stock boundaries of oceanic species are poorly known. Although there are only a few records from Gulf of Mexico waters beyond U.S. boundaries (e.g., Jefferson and Schiro 1997, Ortega Ortiz 2002), killer whales almost certainly occur throughout the oceanic Gulf of Mexico (Jefferson et al. 2008), which is also composed of waters belonging to Mexico and Cuba where there is currently little information on cetacean species abundance and distribution. U.S. waters only comprise about 40% of the entire Gulf of Mexico, and 65% of oceanic

Figure 1. Distribution of killer whale on-effort sightings from SEFSC vessel surveys during spring 1996–2001, summer 2003, spring 2004, summer 2009, summer 2017, and summer/fall 2018. Isobaths are the 200-m, 1,000-m, and 2,000-m depth contours. The darker line indicates the U.S. EEZ.
waters are south of the U.S. Exclusive Economic Zone (EEZ).

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

Killer whales exhibit significant variation in genetic diversity, color pattern, feeding behavior, body size and vocalizations worldwide and several different ecotypes have been identified (Bigg et al. 1990; Pitman et al. 2007; Foote et al. 2009; Parsons et al. 2009). Morin et al. (2010) analyzed whole mitogenomes and concluded that several ecotypes should be elevated to full species. A single sample from the Gulf of Mexico was included in this study and it grouped most closely with killer whales from the Antarctic to the exclusion of samples collected in the eastern North Atlantic, and a single sample collected in the western North Atlantic (Morin et al. 2010). Further work is needed to determine where killer whales in the Gulf of Mexico fit in the global picture of killer whale taxonomy.

Different stocks have been identified in the northeastern Pacific based on morphological, behavioral and genetic characteristics (Bigg et al. 1990; Hoelzel 1991). There is no information on stock differentiation for the Atlantic Ocean population; although an analysis of vocalizations of killer whales from Iceland and Norway indicated that whales from these areas may represent different stocks (Moore et al. 1988). Thirty-two individuals have been photographically identified to date in the northern Gulf of Mexico, with one individual having been sighted over a 20-year period, four whales resighted over 10 years. Thirty animals have been sighted over a range of more than 1,100 km (O'Sullivan and Mullin 1997). The Gulf of Mexico population is provisionally being considered a separate stock for management purposes, although there is currently no information to differentiate this stock from the Atlantic Ocean stock(s). Additional morphological, genetic and/or behavioral data are needed to provide further information on stock delineation.

Killer whales in the northern Gulf of Mexico are managed separately from those in the western North Atlantic. Although there is currently no information to differentiate the stocks, such separation is consistent with the fact that the two areas belong to distinct marine ecoregions (Spalding et al. 2007; Moore and Merrick 2011) and the photo-identification data suggest some degree of long-term site fidelity to the Gulf of Mexico. Thirty-two individual killer whales have been photographically identified to date in the northern Gulf of Mexico, with one individual having been sighted over a 20-year period, four whales resighted over 15 years, and three whales resighted over 10 years. Due to the paucity of sightings in the northern Gulf of Mexico, there are insufficient data to determine whether the northern Gulf of Mexico stock comprises multiple demographically independent populations. Additional morphological, acoustic, genetic and/or behavioral data are needed to further delineate population structure within the Gulf of Mexico and across the broader geographic area.

**POPULATION SIZE**

The best abundance estimate (Nbest) for the northern Gulf of Mexico killer whale is 267 (CV=0.75; Table 1). This estimate is from summer 2017 and summer/fall 2018 oceanic surveys covering waters from the 200-m isobath to the seaward extent of the U.S. EEZ (Garrison et al. 2020). The best abundance estimate available for northern Gulf of Mexico killer whales is 28 (CV=1.02; Table 1). This estimate is from a summer 2009 oceanic survey covering waters from the 200 m isobath to the seaward extent of the U.S. EEZ.

**Earlier abundance estimates**

Please see Appendix IV for a summary of abundance estimates, including earlier estimates and survey descriptions. All estimates of abundance were derived through the application of distance sampling analysis (Buckland et al. 2001) and the computer program DISTANCE (Thomas et al. 1998) to line transect survey data collected from ships in the oceanic northern Gulf of Mexico (i.e., 200-m isobath to seaward extent of the U.S. EEZ) and are summarized in Appendix IV.

From 1991 through 1994, and from 1996 through 2001 (excluding 1998), annual surveys were conducted during spring along a fixed plankton-sampling trackline. Due to limited survey effort in any given year, the survey
effort-weighted estimated average abundance of killer whales for all surveys combined was estimated. For 1991 to 1994, the estimate was 277 (CV=0.42) (Hansen et al. 1995), and for 1996 to 2001, 133 (CV=0.49) (Mullin and Fulling 2004; Table 1).

During summer 2003 and spring 2004, surveys dedicated to estimating cetacean abundance were conducted along a grid of uniformly spaced transect lines from a random start. The abundance estimate for killer whales, pooled from 2003 to 2004, was 49 (CV=0.77) (Mullin 2007; Table 1).

Recent surveys and abundance estimates

An abundance estimate for killer whales was generated from vessel surveys conducted in the northern Gulf of Mexico from the continental shelf edge (~200-m isobath) to the seaward extent of the U.S. EEZ (Garrison et al. 2020). One survey was conducted from 2 July to 25 August 2017 and consisted of 7,302 km of on-effort trackline, and the second survey was conducted from 11 August to 6 October 2018 and consisted of 6,473 km of on-effort trackline within the surveyed strata. Both surveys used a double-platform data-collection procedure, which allowed estimation of the detection probability on the trackline using the independent observer approach assuming point independence (Laake and Borchers 2004). Abundance was calculated using mark-recapture distance sampling implemented in package mrds (version 2.21, Laake et al. 2020) in the R statistical programming language. This approach accounted for the effects of covariates (e.g., sea state, glare) on detection probability within the surveyed strip. The surveys were conducted in "passing mode" (e.g., Schwarz et al. 2010) while all prior surveys in the Gulf of Mexico have been conducted in "closing mode." The abundance estimate for this stock included sightings of unidentified small whales that were apportioned among identified species based on their relative density within the survey strata (Garrison et al. 2020). There were sightings of killer whales in 2017 but there were none in 2018. Unidentified small whales observed during the 2018 survey were apportioned by the relative density from the summer 2017 survey to develop an abundance estimate for killer whales in 2018. The inverse variance weighted mean abundance estimate for killer whales in oceanic waters during 2017 and 2018 was 267 (CV=0.75; Table 1; Garrison et al. 2020). Unlike previous abundance estimates, this estimate was corrected for the probability of detection on the trackline. During summer 2009, a line-transect survey dedicated to estimating the abundance of oceanic cetaceans was conducted in the northern Gulf of Mexico. Survey lines were stratified in relation to depth and the location of the Loop Current. The abundance estimate for killer whales in oceanic waters during 2009 was 28 (CV=1.02; Table 1).

Table 1. Summary of abundance estimates for northern Gulf of Mexico killer whales. Month, year and area covered during each abundance survey, and resulting abundance estimate (Nbest) and coefficient of variation (CV).

<table>
<thead>
<tr>
<th>Month/Year</th>
<th>Area</th>
<th>Nbest</th>
<th>CV</th>
</tr>
</thead>
<tbody>
<tr>
<td>Apr-Jun 1991-1994</td>
<td>Oceanic waters</td>
<td>277</td>
<td>0.42</td>
</tr>
<tr>
<td>Apr-Jun 1996-2001 (excluding 1998)</td>
<td>Oceanic waters</td>
<td>133</td>
<td>0.49</td>
</tr>
<tr>
<td>Jun-Aug 2003, Apr-Jun 2004</td>
<td>Oceanic waters</td>
<td>49</td>
<td>0.77</td>
</tr>
<tr>
<td>Jun-Aug 2009</td>
<td>Oceanic waters</td>
<td>28</td>
<td>1.02</td>
</tr>
</tbody>
</table>

Table 1. Most recent abundance estimate (Nbest) and coefficient of variation (CV) of northern Gulf of Mexico killer whales in oceanic waters (200 m to the offshore extent of the EEZ) based on an average from summer 2017 and summer/fall 2018 vessel surveys.
Minimum Population Estimate

The minimum population estimate (Nmin) is the lower limit of the two-tailed 60% confidence interval of the log-normal distributed abundance estimate. This is equivalent to the 20th percentile of the log-normal distributed abundance estimate as specified by Wade and Angliss (1997). The best estimate of abundance for killer whales is 26728 (CV=0.751.02). The minimum population estimate for the northern Gulf of Mexico killer whale is 152 (Table 2).器 killer whales.

Current Population Trend

The statistical power to detect a trend in abundance for this stock is poor due to the relatively imprecise abundance estimates and long intervals between surveys. For example, the power to detect a precipitous decline in abundance (i.e., 50% decrease in 15 years) with estimates of low precision (e.g., CV>0.30) remains below 80% (alpha = 0.30) unless surveys are conducted on an annual basis (Taylor et al. 2007). In addition, because these surveys are restricted to U.S. waters, it is not possible to distinguish between changes in population size and Gulf-wide shifts in spatial distribution. Five point estimates of killer whale abundance have been made based on data from surveys during 2003 (June–August), 2004 (April–June), 2009 (July–August), 2017 (July–August), and 2018 (August–October). Each of these surveys had a similar design and was conducted using the same vessel or a vessel with a similar observation platform. Prior year surveys only employed a single survey team. The estimates for 2017 and 2018 were conducted in "passing" mode which resulted in increased numbers of unidentified sightings and may have affected group size estimation for distant groups of dolphins and small whales. Apportioning unidentified species among identified taxa addresses the first issue, and there were no apparent relationships between sighting distance and estimated group size. The model for detection probability on the trackline from the summer 2017 survey was applied to the abundance estimates from the 2003, 2004, and 2009 surveys. The resulting abundance estimates were: 2003, N=0 (CV=NA); 2004, N=198 (CV=1.00); 2009, N=51 (CV=0.97); 2017, N=86 (CV=0.87); and 2018, N=450 (CV=0.88). Pairwise comparisons of the non-zero log-transformed means were conducted between years, and significant differences were assessed at alpha=0.10. P-values were adjusted for multiple comparisons. There were no significant differences between survey years. There are insufficient data to determine the population trends for this species. The pooled abundance estimate for 2003-2004 of 49 (CV=0.77) and that for 1996-2001 of 133 (CV=0.49) are not significantly different (P=0.05), but due to the precision of the estimates, the power to detect a difference is low. The abundance estimate for 1991-1994 was 277 (CV=0.42). The large relative changes in the total abundances of killer whales are probably due to a number of factors. The killer whale is most certainly a resident species in the Gulf of Mexico but probably occurs in low numbers and the survey effort is not sufficient to estimate the abundance of uncommon or rare species with precision. Also, these temporal abundance estimates are difficult to interpret without a Gulf of Mexico-wide understanding of killer whale abundance. The killer whale, like all the other oceanic cetacean species in the Gulf, is a mobile predator and this stock is most likely a transboundary stock. The oceanography of the Gulf of Mexico is quite dynamic, and the spatial scale of the Gulf is small relative to the ability of most cetacean species to travel. Studies based on abundance and distribution surveys restricted to U.S. waters are unable to detect temporal shifts in distribution beyond U.S. waters that might account for any changes in abundance.

CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

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

POTENTIAL BIOLOGICAL REMOVAL

Potential Biological Removal (PBR) is the product of the minimum population size, one half the maximum net productivity rate and a recovery factor (MMPA Sec. 3.16 U.S.C. 1362; Wade and Angliss 1997). The minimum population size is 152.器 killer whales. The maximum productivity rate is 0.04, the default value for cetaceans. The recovery factor, which accounts for endangered, depleted, threatened stocks, or stocks of unknown status relative to optimum sustainable population (OSP), is assumed to be 0.5 because the stock is of unknown status. PBR for the northern Gulf

<table>
<thead>
<tr>
<th>Years</th>
<th>Area</th>
<th>Nbest</th>
<th>CV</th>
</tr>
</thead>
<tbody>
<tr>
<td>2017, 2018</td>
<td>Gulf of Mexico</td>
<td>267</td>
<td>0.75</td>
</tr>
</tbody>
</table>
of Mexico killer whale is 1.50.1 (Table 2).

**Table 2. Best and minimum abundance estimates for northern Gulf of Mexico killer whales with Maximum Productivity Rate (Rmax), Recovery Factor (Fr) and PBR.**

<table>
<thead>
<tr>
<th>Nbest</th>
<th>CV</th>
<th>Nmin</th>
<th>Fr</th>
<th>Rmax</th>
<th>PBR</th>
</tr>
</thead>
<tbody>
<tr>
<td>267</td>
<td>0.75</td>
<td>152</td>
<td>0.5</td>
<td>0.04</td>
<td>1.5</td>
</tr>
</tbody>
</table>

**ANNUAL HUMAN-CAUSED MORTALITY AND SERIOUS INJURY**

Total annual estimated fishery-related mortality and serious injury to this stock during 2014–2018 was presumed to be zero, as there were no reports of mortalities or serious injuries to killer whales in the Gulf of Mexico (Table 3). Mean annual mortality and serious injury during 2014–2018 due to other human-caused actions (e.g., the Deepwater Horizon oil spill) was unknown (see Habitat Issues section). The minimum total mean annual human-caused mortality and serious injury for this stock during 2014–2018 was, therefore, unknown. There has been no reported fishing-related mortality of a killer whale during 1998–2010 (Yeung 1999; 2001; Garrison 2003; Garrison and Richards 2004; Garrison 2005; Fairfield Walsh and Garrison 2006; Fairfield Walsh and Garrison 2007; Fairfield and Garrison 2008; Garrison et al. 2009; Garrison and Stokes 2010; 2011). However, during 2008 there was 1 killer whale released alive with no serious injury after an entanglement interaction with the pelagic longline fishery (Garrison et al. 2009).

**Table 3. Total annual estimated fishery-related mortality and serious injury for northern Gulf of Mexico killer whales.**

<table>
<thead>
<tr>
<th>Years</th>
<th>Source</th>
<th>Annual Avg.</th>
<th>CV</th>
</tr>
</thead>
<tbody>
<tr>
<td>2014–2018</td>
<td>U.S. fisheries using observer data</td>
<td>0</td>
<td>-</td>
</tr>
</tbody>
</table>

**Fisheries Information**

There are two commercial fisheries that interact, or that could potentially interact, with this stock in the Gulf of Mexico. These are the Category I Atlantic Highly Migratory Species longline fishery and the Category I Atlantic Ocean, Caribbean, Gulf of Mexico large pelagic longline fishery (Appendix III). Percent observer coverage (percentage of sets observed) for these longline fisheries for each year during 2014–2018 was 18, 19, 23, 13 and 20, respectively. There is very little effort within the Gulf of Mexico by the Atlantic Highly Migratory Species (high seas) longline fishery, and no takes of killer whales within high seas waters of the Gulf of Mexico have been observed or reported thus far. Pelagic swordfish, tunas and billfish are the targets of the large pelagics longline fishery operating in the northern Gulf of Mexico. During 2014–2018 there were no observed mortalities or serious injuries to killer whales by this fishery (Garrison and Stokes 2016; 2017; 2019; 2020; in review). The commercial fishery which potentially could interact with this stock in the Gulf of Mexico is the Atlantic Ocean, Caribbean, Gulf of Mexico large pelagic longline fishery (Appendix III). Pelagic swordfish, tunas and billfish are the targets of the longline fishery operating in the northern Gulf of Mexico. There were no reports of mortality or serious injury to killer whales by this fishery during 1998–2010 (Yeung 1999; 2001; Garrison 2003; Garrison and Richards 2004; Garrison 2005; Fairfield Walsh and Garrison 2006; Fairfield Walsh and Garrison 2007; Fairfield and Garrison 2008; Garrison et al. 2009; Garrison and Stokes 2010; 2011). However, on 17 May 2008 there was 1 killer whale released alive with no serious injury after an entanglement interaction with the pelagic longline fishery (Garrison et al. 2009). This was the second observed interaction between a killer whale and this fishery and the first observed interaction within the Gulf of Mexico. During 15 April–15 June 2008 observer coverage in the Gulf of Mexico was greatly enhanced to collect more robust information on the interactions between pelagic longline vessels and spawning bluefin tuna. Resulting observer coverage for this time and area was dramatically higher than typical for previous years (Garrison et al. 2009).

**Other Mortality**

There were no reported strandings of killer whales in the Gulf of Mexico during 2006–2018-2010 (NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 21 May 2019; 4 November 2014). Stranding data probably underestimate the extent of human and fishery-related mortality and serious injury because not all of the whales that die or are seriously injured in human interactions wash ashore, or, if they do, they are not all recovered (Peltier et al. 2012; Wells et al. 2015). In particular, oceanic stocks in the Gulf of Mexico are less likely to strand than nearshore coastal stocks or shelf stocks (Williams et al. 2011). Additionally, not all carcasses will show evidence of human interaction, entanglement or other fishery-related interaction due to
decomposition, scavenger damage, etc. (Byrd et al. 2014). Finally, the level of technical expertise among stranding network personnel varies widely as does the ability to recognize signs of human interaction. Stranding data probably underestimate the extent of fishery-related mortality and serious injury because not all of the marine mammals which die or are seriously injured in fishery interactions wash ashore, not all that wash ashore are discovered, reported or investigated, 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 interactions.

An Unusual Mortality Event (UME) was declared for cetaceans in the northern Gulf of Mexico beginning 1 February/March 2010 and ending 31 July 2014 (Litz et al. 2014; https://www.fisheries.noaa.gov/national/marine-life-distress/2010-2014-cetacean-unusual-mortality-event-northern-gulf-mexico) and, as of early 2012, the event is still ongoing. It includes cetaceans that stranded prior to the Deepwater Horizon (DWH) oil spill (see “Habitat Issues” below), during the spill, and after. Exposure to the DWH oil spill was determined to be the primary underlying cause of the elevated stranding numbers in the northern Gulf of Mexico after the spill (e.g., Schwacke et al. 2014; Venn-Watson et al. 2015; Colegrove et al. 2016; DWH NRDAT 2016; see Habitat Issues section). However, there were no killer whale strandings recovered within the spatial and temporal boundaries of this UME. During 2010, no animals from this stock were considered to be part of the UME.

HABITAT ISSUES

The Deepwater Horizon (DWH) MC252 drilling platform, located approximately 80 miles80 km southeast of the Mississippi River Delta in waters about 1,500 m deep, exploded on 20 April 2010. The rig sank, and forever 87 days ~3.2 million barrels of oil and gas were discharged from the wellhead until it was capped on 15 July 2010 (DWH NRDAT 2016). During the response effort dispersants were applied extensively at the seafloor and at the sea surface (Lehr et al. 2010; OSAT 2010). In situ burning, or controlled burning of oil at the surface, was also used extensively as a response tool (Lehr et al. 2010). The oil, dispersant and burn residue compounds present ecological concerns. The magnitude of this oil spill was unprecedented in U.S. history, causing impacts to wildlife, natural habitats and human communities along coastal areas from western Louisiana to the Florida Panhandle (NOAA 2011). It could be years before the entire scope of damage is ascertained (NOAA 2011).

Shortly after the oil spill, the Natural Resource Damage Assessment (NRDA) process was initiated under the Oil Pollution Act of 1990. A variety of NRDA research studies are being conducted to determine potential impacts of the spill on marine mammals. These studies did not include killer whales regarding impacts of the spill due to insufficient data to determine the overlap of the DWH oil spill footprint and the range of killer whales (DWH MMIQT 2015). The impact of the spill on killer whales is unknown.

Anthropogenic sound in the world’s oceans has been shown to affect marine mammals, with vessel traffic, seismic surveys, and active naval sonars being the main anthropogenic contributors to low- and mid-frequency noise in oceanic waters (e.g., Nowacek et al. 2015; Gomez et al. 2016; NMFS 2018). The long-term and population consequences of these impacts are less well-documented and likely vary by species and other factors. Impacts on marine mammal prey from sound are also possible (Carroll et al. 2017), but the duration and severity of any such prey effects on marine mammals are unknown. These studies have focused on identifying the type, magnitude, severity, length and impact of oil exposure to oceanic, coastal and estuarine marine mammals. The research is ongoing and likely will continue for some time. For continental shelf and oceanic cetaceans, the NOAA-led efforts include: aerial surveys to document the distribution, abundance, species and exposure of marine mammals and turtles relative to oil from DWH spill; and ship surveys to evaluate exposure to oil and other chemicals and to access changes in animal behavior and distribution relative to oil exposure through visual and acoustic surveys, deployment of passive acoustic monitoring systems, collection of tissue samples, and deployment of satellite tags on sperm and Bryde’s whales.

Aerial surveys have observed Risso’s dolphins, spinner dolphins, pantropical spotted dolphins, striped dolphins, bottlenose dolphins and sperm whales swimming in oil in offshore waters (NOAA 2010a). The effects of oil exposure on marine mammals depend on a number of factors including the type and mixture of chemicals involved, the amount, frequency and duration of exposure, the route of exposure (inhaled, ingested, absorbed, or external) and biomedical risk factors of the particular animal (Geraci 1990; NOAA 2010b). In general, direct external contact with petroleum compounds or dispersants with skin may cause skin irritation, chemical burns and infections. Inhalation of volatile petroleum compounds or dispersants may irritate or injure the respiratory tract, which could lead to pneumonia or inflammation. Ingestion of petroleum compounds may cause injury to the gastrointestinal tract, which could affect an animal’s ability to digest or absorb food. Absorption of petroleum compounds or dispersants may damage kidney,
liver and brain function in addition to causing immune suppression and anemia. Long term chronic effects such as lowered reproductive success and decreased survival may occur (Geraci 1990; NOAA 2010b).

**STATUS OF STOCK**

The northern Gulf of Mexico stock of killer whales is not listed as threatened or endangered under the Endangered Species Act, nor is it considered strategic under the Marine Mammal Protection Act. No fishery-related mortality or serious injury has been observed in recent years; therefore, total fishery-related mortality and serious injury can be considered insignificant and approaching the zero mortality and serious injury rate. The status of killer whales in the northern Gulf of Mexico, relative to OSP, is unknown. The species is not listed as threatened or endangered under the Endangered Species Act. There was no statistically significant trend in population size for this stock in the northern Gulf of Mexico. There are insufficient data to determine the population trends for this species. Total human caused mortality and serious injury for this stock is not known but none has been documented. There is insufficient information available to determine whether the total fishery related mortality and serious injury for this stock is insignificant and approaching zero mortality and serious injury rate. This is not a strategic stock because it is assumed that the average annual human-related mortality and serious injury does not exceed PBR.

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Garrison, L.P. and L. Stokes. in review. Estimated bycatch of marine mammals and sea turtles in the U.S. Atlantic pelagic longline fleet during 2018. Southeast Fisheries Science Center, Protected Resources and Biodiversity Division, 75 Virginia Beach Dr., Miami, Florida 33140. PRD Contribution # PRD-2020-08. 55 pp.


FALSE KILLER WHALE (*Pseudorca crassidens*): Northern Gulf of Mexico Stock

**STOCK DEFINITION AND GEOGRAPHIC RANGE**

The false killer whale is distributed worldwide throughout warm temperate and tropical oceans (Leatherwood and Reeves 1983). Sightings of this species in the northern Gulf of Mexico (i.e., U.S. Gulf of Mexico) occur in oceanic waters, primarily in the eastern Gulf (Figure 1; Mullin and Fulling 2004; Maze-Foley and Mullin 2006; Garrison and Aichinger Dias 2020). They are sporadically seen during vessel and False killer whales were seen only in the spring and summer during GulfCet aerial surveys of the northern Gulf of Mexico between 1992 and 1998 (Hansen et al. 1996; Mullin and Hoggard 2000; Mullin and Fulling 2004; Maze-Foley and Mullin 2006; Garrison and Aichinger Dias 2020). Because there are many confirmed records from Gulf of Mexico waters beyond U.S. boundaries (e.g., Jefferson and Schiro 1997, Ortega Ortiz 2002), false killer whales almost certainly occur throughout the oceanic Gulf of Mexico (Jefferson et al. 2008), which is also composed of waters belonging to Mexico and Cuba where there is currently little information on cetacean species abundance and distribution. U.S. waters only comprise about 40% of the entire Gulf of Mexico, and 65% of oceanic waters are south of the U.S. Exclusive Economic Zone (EEZ). All the cetacean species found in the oceanic northern Gulf of Mexico almost certainly occur in similar habitat beyond U.S. boundaries in the southern Gulf. There are fewer cetacean sighting and stranding records in the southern Gulf due to more limited effort. Nevertheless there are records for most oceanic species in the southern Gulf (e.g., Jefferson and Schiro 1997; Ortega Ortiz 2002; Ortega-Argueta et al. 2005; Jefferson et al. 2008; Vázquez Castán et al. 2009; Whitt et al. 2011). This is therefore likely a transboundary stock with Cuba and/or Mexico. Because U.S. waters only comprise about 40% of the entire Gulf of Mexico and 35% of the oceanic (i.e., >200 m) Gulf of Mexico (Mullin and Fulling 2004), abundance and stock boundaries of oceanic species are poorly known.

Genetic analyses (Chivers et al. 2007; Martien et al. 2014) indicate false killer whales exhibit significant population structuring in the Pacific, with restricted gene flow among whales sampled near the main Hawaiian Islands, the Northwestern Hawaiian Islands, and pelagic waters of the eastern and the central North Pacific. Martien et al. (2014) also found their two Atlantic samples to be genetically divergent from those in the Pacific. False killer whales

![Figure 1. Distribution of false killer whale on-effort sightings from SEFSC vessel surveys during spring 1996-2001, summer 2003, spring 2004, summer 2009, summer 2017, and summer/fall 2018. Isobaths are the 200-m, 1,000-m, and 2,000-m depth contours. The darker line indicates the U.S. EEZ.](image-url)
in the northern Gulf of Mexico are managed separately from those in the western North Atlantic. Although there is currently no information to differentiate the two stocks, such separation is consistent with evidence for strong population structuring in other areas (Martien et al. 2014) and further supported because the two stocks occupy distinct marine ecoregions (Spalding et al. 2007; Moore and Merrick 2011). There are insufficient data to determine whether the northern Gulf of Mexico stock comprises multiple demographically independent populations. The Gulf of Mexico population is provisionally being considered 1 stock for management purposes, although there is currently no information to differentiate this stock from the Atlantic Ocean stock(s). Additional morphological, acoustic, genetic and/or behavioral data are needed to provide further delineate population structure within the Gulf of Mexico and across the broader geographic area. Information on stock delineation.

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

**POPULATION SIZE**

The best abundance estimate (Nbest) for the northern Gulf of Mexico false killer whale is 494 (CV=0.79; Table 1). This estimate is from summer 2017 and summer/fall 2018 oceanic surveys covering waters from the 200-m isobath to the seaward extent of the U.S. EEZ (Garrison et al. 2020). The current population size for the false killer whale in the northern Gulf of Mexico is unknown because the survey data are more than 8 years old (Wade and Angliss 1997).

**Earlier abundance estimates**

Please see Appendix IV for a summary of abundance estimates, including earlier estimates and survey descriptions. All estimates of abundance were derived through the application of distance sampling analysis (Buckland et al. 2001) and the computer program DISTANCE (Thomas et al. 1998) to line transect survey data collected from ships in the oceanic northern Gulf of Mexico (i.e., 200m isobath to seaward extent of the U.S. EEZ) and are summarized in Appendix IV.

—— From 1991 through 1994, and from 1996 through 2001 (excluding 1998), annual surveys were conducted during spring along a fixed plankton-sampling trackline. Due to limited survey effort in any given year, the survey effort-weighted estimated average abundance of false killer whales for all surveys combined was estimated. For 1991 to 1994, the estimate was 381 (CV=0.62) (Hansen et al. 1995), and for 1996 to 2001, 1038 (CV=0.71) (Mullin and Fulling 2004; Table 1).

—— During summer 2003 and spring 2004, surveys dedicated to estimating cetacean abundance were conducted along a grid of uniformly-spaced transect lines from a random start. The most recent abundance estimate for false killer whales, pooled from 2003 to 2004, was 777 (CV=0.56) (Mullin 2007; Table 1). Because these data are more than 8 years old, the current best population estimate is unknown.

**Recent surveys and abundance estimates**

An abundance estimate for false killer whales was generated from vessel surveys conducted in the northern Gulf of Mexico from the continental shelf edge (~200-m isobath) to the seaward extent of the U.S. EEZ (Garrison et al. 2020). One survey was conducted from 2 July to 25 August 2017 and consisted of 7,302 km of on-effort trackline, and the second survey was conducted from 11 August to 6 October 2018 and consisted of 6,473 km of on-effort trackline within the surveyed strata. Both surveys used a double-platform data-collection procedure, which allowed estimation of the detection probability on the trackline using the independent observer approach assuming point independence (Laake and Borchers 2004). Abundance was calculated using mark-recapture distance sampling implemented in package mrds (version 2.21, Laake et al. 2020) in the R statistical programming language. This approach accounted for the effects of covariates (e.g., sea state, glare) on detection probability within the surveyed strip. The surveys were conducted in "passing mode" (e.g., Schwarz et al. 2010) while all prior surveys in the Gulf of Mexico have been conducted in "closing mode." The abundance estimate for this stock included sightings of unidentified small whales that were apportioned among identified species based on their relative density within the survey strata (Garrison et al. 2020). There were sightings of false killer whales in 2017 but there were none in 2018.
Unidentified small whales observed during the 2018 survey were apportioned by the relative density from the summer 2017 survey to develop an abundance estimate for false killer whales in 2018. The inverse variance weighted mean abundance estimate for false killer whales in oceanic waters during 2017 and 2018 was 494 (CV=0.79; Table 1; Garrison et al. 2020). Unlike previous abundance estimates, this estimate was corrected for the probability of detection on the trackline. During summer 2009, a line-transect survey dedicated to estimating the abundance of oceanic cetaceans was conducted in the northern Gulf of Mexico. Survey lines were stratified in relation to depth and the location of the Loop Current. One group of false killer whales was sighted during the 2009 survey on an effort segment that was not included in the line-transect analysis. Therefore, false killer whale abundance could not be estimated from the 2009 survey.

Table 1. Most recent abundance estimate (Nbest) and coefficient of variation (CV) of northern Gulf of Mexico false killer whales in oceanic waters (200 m to the offshore extent of the EEZ) based on an average from summer 2017 and summer/fall 2018 vessel surveys.

<table>
<thead>
<tr>
<th>Years</th>
<th>Area</th>
<th>Nbest</th>
<th>CV</th>
</tr>
</thead>
<tbody>
<tr>
<td>2017, 2018</td>
<td>Gulf of Mexico</td>
<td>494</td>
<td>0.79</td>
</tr>
</tbody>
</table>

Table 1. Summary of abundance estimates for northern Gulf of Mexico false killer whales. Month, year and area covered during each abundance survey, and resulting abundance estimate (Nbest) and coefficient of variation (CV).

<table>
<thead>
<tr>
<th>Month/Year</th>
<th>Area</th>
<th>Nbest</th>
<th>CV</th>
</tr>
</thead>
<tbody>
<tr>
<td>Apr-Jun 1991–1994</td>
<td>Oceanic waters</td>
<td>381</td>
<td>0.62</td>
</tr>
<tr>
<td>Apr-Jun 1996–2001 (excluding 1998)</td>
<td>Oceanic waters</td>
<td>1,038</td>
<td>0.71</td>
</tr>
<tr>
<td>Jun-Aug 2003, Apr-Jun 2004</td>
<td>Oceanic waters</td>
<td>777</td>
<td>0.56</td>
</tr>
</tbody>
</table>

Minimum Population Estimate
The minimum population estimate is unknown. The minimum population estimate (Nmin) is the lower limit of the two-tailed 60% confidence interval of the log-normal distributed abundance estimate. This is equivalent to the 20th percentile of the log-normal distributed abundance estimate as specified by Wade and Angliss (1997). The best estimate of abundance for false killer whales is 494 (CV=0.79) unknown. The minimum population estimate for the northern Gulf of Mexico false killer whale is 276 (Table 2) unknown.

Current Population Trend
The statistical power to detect a trend in abundance for this stock is poor due to the relatively imprecise abundance estimates and long intervals between surveys. For example, the power to detect a precipitous decline in abundance (i.e., 50% decrease in 15 years) with estimates of low precision (e.g., CV>0.30) remains below 80% (alpha=0.30) unless surveys are conducted on an annual basis (Taylor et al. 2007). In addition, because these surveys are restricted to U.S. waters, it is not possible to distinguish between changes in population size and Gulf-wide shifts in spatial distribution. Five point estimates of false killer whale abundance have been made based on data from surveys during: 2003 (June–August), 2004 (April–June), 2009 (July–August), 2017 (July–August), and 2018 (August–October). Each of these surveys had a similar design and was conducted using the same vessel or a vessel with a similar observation platform. Prior year surveys only employed a single survey team. The estimates for 2017 and 2018 were conducted in “passing” mode which resulted in increased numbers of unidentified sightings and may have affected group size...
estimation for distant groups of dolphins and small whales. Apportioning unidentified species among identified taxa addresses the first issue, and there were no apparent relationships between sighting distance and estimated group size. The model for detection probability on the trackline from the summer 2017 survey was applied to the abundance estimates from the 2003, 2004, and 2009 surveys. The resulting abundance estimates were: 2003, N=1,293 (CV=0.63); 2004, N=0 (CV=NA); 2009, N=0 (CV=NA); 2017, N=1,069 (CV=0.97); and 2018, N=162 (CV=0.74). Pairwise comparisons of the non-zero log-transformed means were conducted between years, and significant differences were assessed at alpha=0.10. P-values were adjusted for multiple comparisons. There were significant differences between the 2003 and 2018 estimates (p.adjusted=0.027) and the 2017 and 2018 estimates (p.adjusted=0.072) suggesting a possible decline in abundance during recent years. Three point estimates of false killer whale abundance have been made based on data from surveys covering 1991-2004. The estimates vary by a maximum factor of nearly three. To determine whether changes in abundance have occurred over this period, an analysis of all the survey data needs to be conducted which incorporates covariates (e.g., survey conditions, season) that could potentially aect estimates. Nevertheless, differences in temporal abundance estimates will still be diicult to interpret without a Gulf of Mexico-wide understanding of false killer whale abundance. The oceanography of the Gulf of Mexico is quite dynamic, and the spatial scale of the Gulf is small relative to the ability of most cetacean species to travel. Studies based on abundance and distribution surveys restricted to U.S. waters are unable to detect temporal shifts in distribution beyond U.S. waters that might account for any changes in abundance.

CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

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

POTENTIAL BIOLOGICAL REMOVAL

Potential Biological Removal (PBR) is undetermined. PBR is the product of the minimum population size, one half the maximum net productivity rate and a recovery factor (MMPA Sec. 3.16 U.S.C. 1362; Wade and Angliss 1997). The minimum population size is unknown. The maximum productivity rate is 0.04, the default value for cetaceans. The recovery factor, which accounts for endangered, depleted, threatened stocks, or stocks of unknown status relative to optimum sustainable population (OSP), is assumed to be 0.5 because the stock is of unknown status. PBR for the northern Gulf of Mexico false killer whale is 2.8 (Table 2).

### Table 2. Best and minimum abundance estimates for northern Gulf of Mexico false killer whale with Maximum Productivity Rate (Rmax), Recovery Factor (Fr) and PBR.

<table>
<thead>
<tr>
<th>Nbest</th>
<th>CV</th>
<th>Nmin</th>
<th>Fr</th>
<th>Rmax</th>
<th>PBR</th>
</tr>
</thead>
<tbody>
<tr>
<td>494</td>
<td>0.79</td>
<td>276</td>
<td>0.5</td>
<td>0.04</td>
<td>2.8</td>
</tr>
</tbody>
</table>

ANNUAL HUMAN- CAUSED MORTALITY AND SERIOUS INJURY

Total annual estimated fishery-related mortality and serious injury to this stock during 2014–2018 was presumed to be zero, as there were no reports of mortalities or serious injuries to false killer whales in the Gulf of Mexico (Table 3). Mean annual mortality and serious injury during 2014–2018 due to other human-caused actions (the Deepwater Horizon oil spill) was predicted to be 2.2. The minimum total mean annual human-caused mortality and serious injury for this stock during 2014–2018 was, therefore, 2.2. There has been no reported fishing-related mortality or serious injury of a false killer whale during 1998–2010 (Yeung 1999; Yeung 2001; Garrison 2003; Garrison and Richards 2004; Garrison 2005; Fairfield Walsh and Garrison 2006; Fairfield-Walsh and Garrison 2007; Fairfield and Garrison 2008; Garrison et al. 2009; Garrison and Stokes 2010; 2011).

### Table 3. Total annual estimated fishery-related mortality and serious injury for northern Gulf of Mexico false killer whales.

<table>
<thead>
<tr>
<th>Years</th>
<th>Source</th>
<th>Annual Avg.</th>
<th>CV</th>
</tr>
</thead>
<tbody>
<tr>
<td>2014–2018</td>
<td>U.S. fisheries using observer data</td>
<td>0</td>
<td></td>
</tr>
</tbody>
</table>
Fisheries Information

There are two commercial fisheries that interact, or that could potentially interact, with this stock in the Gulf of Mexico. These are the Category I Atlantic Highly Migratory Species (high seas) longline fishery and the Category I Atlantic Ocean, Caribbean, Gulf of Mexico large pelagics longline fishery (Appendix III). Percent observer coverage (percentage of sets observed) for these longline fisheries for each year during 2014–2018 was 18, 19, 23, 13 and 20, respectively. There is very little effort within the Gulf of Mexico by the Atlantic Highly Migratory Species (high seas) longline fishery, and no takes of false killer whales within high seas waters of the Gulf of Mexico have been observed or reported thus far. Pelagic swordfish, tunas and billfish are the targets of the large pelagics longline fishery operating in the northern Gulf of Mexico. During 2014–2018 there were no observed mortalities or serious injuries to false killer whales by this fishery (Garrison and Stokes 2016; 2017; 2019; 2020; in review). The commercial fishery which potentially could interact with this stock in the Gulf of Mexico is the Atlantic Ocean, Caribbean, Gulf of Mexico large pelagic longline fishery (Appendix III). Pelagic swordfish, tunas and billfish are the targets of the longline fishery operating in the northern Gulf of Mexico. There were no reports of mortality or serious injury to false killer whales by this fishery during 1998–2010 (Yeung 1999; Yeung 2001; Garrison 2003; Garrison and Richards 2004; Garrison 2005; Fairfield Walsh and Garrison 2006; Fairfield Walsh and Garrison 2007; Fairfield and Garrison 2008; Garrison et al. 2009; Garrison and Stokes 2010; 2011).

Other Mortality

There was one mass stranding of 99 false killer whales in the Gulf of Mexico during 2014–2018 (NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 21 May 2019; 16 November 2014). The mass stranding occurred within the Everglades National Park in Florida in 2017. Two animals mass stranded in Florida during 2009. Evidence of human interaction was detected for one of the stranded whales (ingested plastic debris), and for the remaining strandings, it could not be determined if there was evidence of human interactions. Stranding data probably underestimate the extent of human and fishery-related mortality and serious injury because not all of the whales that die or are seriously injured in human interactions wash ashore, or, if they do, they are not all recovered (Peltier et al. 2012; Wells et al. 2015). In particular, oceanic stocks in the Gulf of Mexico are less likely to strand than nearshore coastal stocks or shelf stocks (Williams et al. 2011). Additionally, not all carcasses will show evidence of human interaction, entanglement or other fishery-related interaction due to decomposition, scavenger damage, etc. (Byrd et al. 2014). Finally, the level of technical expertise among stranding network personnel varies widely as does the ability to recognize signs of human interaction. Stranding data probably underestimate the extent of fishery-related mortality and serious injury because not all of the marine mammals which die or are seriously injured in fishery interactions wash ashore, not all that wash ashore are discovered, reported or investigated, 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 interactions.

An Unusual Mortality Event (UME) was declared for cetaceans in the northern Gulf of Mexico beginning 1 February/March 2010 and ending 31 July 2014 (Litz et al. 2014; https://www.fisheries.noaa.gov/national/marine-life-distress/2010-2014-cetacean-unusual-mortality-event-northern-gulf-mexico) and, as of early 2012, the event is still ongoing. It includes cetaceans that stranded prior to the Deepwater Horizon (DWH) oil spill (see “Habitat Issues” below), during the spill, and after. Exposure to the DWH oil spill was determined to be the primary underlying cause of the elevated stranding numbers in the northern Gulf of Mexico after the spill (e.g., Schwacke et al. 2014; Venn-Watson et al. 2015; Colegrove et al. 2016; DWH NRDAT 2016; see Habitat Issues section). However, there were no false killer whale strandings recovered within the spatial and temporal boundaries of this UME during 2010, no animals from this stock were considered to be part of the UME.

A population model was developed to estimate the injury and time to recovery for stocks affected by the DWH oil spill, taking into account long-term effects resulting from mortality, reproductive failure, reduced survival rates, and the proportion of the stock exposed to DWH oil (DWH MIMIQT 2015). Overall, the model estimated that this stock experienced a 9% maximum reduction in population size due to the oil spill (DWH MIMIQT 2015). The mortality projected for the years 2010–2013 due to the spill has not been reported previously. Based on the population model, it was projected that 21 false killer whales died during 2010–2013 (four year annual average of 5.3) due to elevated mortality associated with oil exposure (see Appendix VI). For the 2014–2018 reporting period of this SAR, the population model estimated 11 false killer whales died due to elevated mortality associated with oil exposure. The population model used to predict false killer whale mortality due to the DWH event has a number of sources of uncertainty. Model parameters (e.g., survival rates, reproductive rates, and life-history parameters) were derived from

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literature sources for false killer whales occupying waters outside of the Gulf of Mexico. In addition, proxy values for the effects of DWH oil exposure on both survival rates and reproductive success were applied based upon estimated values for common bottlenose dolphins in Barataria Bay. Finally, there was no estimation of uncertainty in model parameters or outputs.

HABITAT ISSUES

The Deepwater Horizon (DWH) MC252 drilling platform, located approximately 50 miles south of the Mississippi River Delta in waters about 1,500 m deep, exploded on 20 April 2010. The rig sank, and for over 87 days ~3.2 millions of barrels of oil and gas were discharged from the wellhead until it was capped on 15 July 2010 (DWH NRDAT 2016). During the response effort dispersants were applied extensively at the seafloor and at the sea surface (Lehr et al. 2010; OSAT 2010). In-situ burning, or controlled burning of oil at the surface, was also used extensively as a response tool (Lehr et al. 2010). The oil, dispersant and burn residue compounds present ecological concerns. The magnitude of this oil spill was unprecedented in U.S. history, causing impacts to wildlife, natural habitats and human communities along coastal areas from western Louisiana to the Florida Panhandle (NOAA 2011). It could be years before the entire scope of damage is ascertained (NOAA 2011).

Shortly after the oil spill, the Natural Resource Damage Assessment (NRDA) process was initiated under the Oil Pollution Act of 1990. A variety of NRDA research studies are being conducted to determine potential impacts of the spill on marine mammals. These studies estimated that 18% (95% CI: 7–48) of false whales in the Gulf were exposed to oil, that 8% (95% CI: 4–12) of females suffered from reproductive failure, and 7% (95% CI: 3–11) of false killer whales suffered adverse health effects (DWH MMIQT 2015). A population model estimated the stock experienced a maximum 9% reduction in population size (see Other Mortality section above).

Anthropogenic sound in the world’s oceans has been shown to affect marine mammals, with vessel traffic, seismic surveys, and active naval sonars being the main anthropogenic contributors to low- and mid-frequency noise in oceanic waters (e.g., Nowacek et al. 2015; Gomez et al. 2016; NMFS 2018). The long-term and population consequences of these impacts are less well-documented and likely vary by species and other factors. Impacts on marine mammal prey from sound are also possible (Carroll et al. 2017), but the duration and severity of any such prey effects on marine mammals are unknown. These studies have focused on identifying the type, magnitude, severity, length and impact of oil exposure to oceanic, coastal and estuarine marine mammals. The research is ongoing and likely will continue for some time. For continental shelf and oceanic cetaceans, the NOAA-led efforts include: aerial surveys to document the distribution, abundance, species and exposure of marine mammals and turtles relative to oil from DWH spill; and ship surveys to evaluate exposure to oil and other chemicals and to assess changes in animal behavior and distribution relative to oil exposure through visual and acoustic surveys, deployment of passive acoustic monitoring systems, collection of tissue samples, and deployment of satellite tags on sperm and Bryde’s whales.

Aerial surveys have observed Risso’s dolphins, spinner dolphins, pantropical spotted dolphins, striped dolphins, bottlenose dolphins and sperm whales swimming in oil in offshore waters (NOAA 2010a). The effects of oil exposure on marine mammals depend on a number of factors including the type and mixture of chemicals involved, the amount, frequency and duration of exposure, the route of exposure (inhaled, ingested, absorbed, or external) and biomedical risk factors of the particular animal (Geraci 1990; NOAA 2010b). In general, direct external contact with petroleum compounds or dispersants with skin may cause skin irritation, chemical burns and infections. Inhalation of volatile petroleum compounds or dispersants may irritate or injure the respiratory tract, which could lead to pneumonia or inflammation. Ingestion of petroleum compounds may cause injury to the gastrointestinal tract, which could affect an animal’s ability to digest or absorb food. Absorption of petroleum compounds or dispersants may damage kidney, liver and brain function in addition to causing immune suppression and anemia. Long term chronic effects such as lowered reproductive success and decreased survival may occur (Geraci 1990; NOAA 2010b).

STATUS OF STOCK

False killer whales are not listed as threatened or endangered under the Endangered Species Act, and the northern Gulf of Mexico stock is not considered strategic under the MMPA. No fishery-related mortality or serious injury has been observed in recent years; therefore, total fishery-related mortality and serious injury can be considered insignificant and approaching the zero mortality and serious injury rate. The status of false killer whales in the northern Gulf of Mexico, relative to OSP, is unknown. There were statistically significant decreases in population size in recent years for this stock in the northern Gulf of Mexico. The status of false killer whales in the northern Gulf of Mexico, relative to OSP, is unknown. The species is not listed as threatened or endangered under the Endangered Species Act.
There are insufficient data to determine population trends for this stock. Total human caused mortality and serious injury for this stock is not known but none has been documented. There is insufficient information available to determine whether the total fishery-related mortality and serious injury for this stock is insignificant and approaching zero mortality and serious injury rate. Despite an undetermined PBR, this is not a strategic stock because there is no documented human-related mortality and serious injury.

REFERENCES


DWH MMIQT. 2015. Models and analyses for the quantification of injury to Gulf of Mexico cetaceans from the Deepwater Horizon Oil Spill. MM TR.01 Schwacke Quantification.of.Injury.to.GOM.Cetaceans. Southeast Fisheries Science Center, Protected Resources and Biodiversity Division, 75 Virginia Beach Dr., Miami, Florida 33140. PRBD Contribution #: PRBD-2020-02. Available from: https://repository.library.noaa.gov/view/noaa/25568


PYGMY KILLER WHALE (*Feresa attenuata*): Northern Gulf of Mexico Stock

STOCK DEFINITION AND GEOGRAPHIC RANGE

The pygmy killer whale is distributed worldwide in tropical and subtropical waters (Ross and Leatherwood 1994). Sightings of these animals in the northern Gulf of Mexico (i.e., U.S. Gulf of Mexico) occur primarily in waters >1000 m in oceanic waters (Figure 1; Mullin and Fulling 2004; Maze-Foley and Mullin 2006; Garrison and Aichinger Dias 2020). Sightings of pygmy killer whales have been documented in all seasons during GulfCet aerial surveys of the northern Gulf of Mexico between 1992 and 1998 (Hansen et al. 1996; Mullin and Hoggard 2000).

All the cetacean species found in the oceanic northern Gulf of Mexico almost certainly occur in similar habitat beyond U.S. boundaries in the southern Gulf. There are fewer cetacean sighting and stranding records in the southern Gulf due to more limited effort. Nevertheless, there are records for most oceanic species in the southern Gulf (e.g., Jefferson and Schiro 1997; Ortega Ortiz 2002; Ortega-Argueta et al. 2005; Jefferson et al. 2008; Vázquez Castán et al. 2009; Whitt et al. 2011). This is therefore likely a transboundary stock with Cuba and/or Mexico. Because U.S. waters only comprise about 40% of the entire Gulf of Mexico and 35% of the oceanic (i.e., >200 m) Gulf of Mexico (Mullin and Fulling 2004), abundance and stock boundaries of oceanic species are poorly known. Although there are only a few records from Gulf of Mexico waters beyond U.S. boundaries (e.g., Ortega Ortiz 2002), pygmy killer whales almost certainly occur throughout the oceanic Gulf of Mexico (Jefferson et al. 2008), which is also composed of waters belonging to Mexico and Cuba where there is currently little information on cetacean species abundance and distribution. U.S. waters only comprise about 40% of the entire Gulf of Mexico, and 65% of oceanic waters are south of the U.S. Exclusive Economic Zone (EEZ).
Pygmy killer whales in the northern Gulf of Mexico are managed separately from those in the western North Atlantic. Although there have been no directed studies of the degree of demographic independence between the two areas, such separation is consistent with evidence for population structure in other areas (Baird 2018) and is further supported because the two stocks occupy distinct marine ecoregions (Spalding et al. 2007; Moore and Merrick 2011). In addition, two pygmy killer whales that stranded in Mississippi were rehabilitated, tagged with a satellite-linked transmitter, released, and tracked for 15 and 88 days (Pulis et al. 2018). Nearly all the tracked locations occurred over continental slope waters ranging from 200 to 1,200 m in depth in the northern Gulf of Mexico. As Wells et al. (2009) note, it is difficult to determine the effects of stranding and rehabilitation on post-release behavior, so it is unknown whether these movements were representative of pygmy killer whale ranging patterns in the northern Gulf of Mexico. Due to the paucity of sightings, there are insufficient data to determine whether the northern Gulf of Mexico stock comprises multiple demographically independent populations. Additional morphological, acoustic, genetic, and/or behavioral data are needed to further delineate population structure within the Gulf of Mexico and across the broader geographic area. The Gulf of Mexico population is provisionally being considered a separate stock for management purposes, although there is currently no information to differentiate this stock from the Atlantic Ocean stock(s). Additional morphological, genetic and/or behavioral data are needed to provide further information on stock delineation.

**POPULATION SIZE**

The best abundance estimate (Nbest) for the northern Gulf of Mexico pygmy killer whale is 613 (CV=1.15; Table 1). This estimate is from summer 2017 and summer/fall 2018 oceanic surveys covering waters from the 200-m isobath to the seaward extent of the U.S. EEZ (Garrison et al. 2020). The best abundance estimate available for northern Gulf of Mexico pygmy killer whales is 152 (CV=1.02; Table 1). This estimate is from a summer 2009 oceanic survey covering waters from the 200m isobath to the seaward extent of the U.S. EEZ.

**Earlier abundance estimates**

Please see Appendix IV for a summary of abundance estimates, including earlier estimates and survey descriptions. All estimates of abundance were derived through the application of distance sampling analysis (Buckland et al. 2001) and the computer program DISTANCE (Thomas et al. 1998) to line transect survey data collected from ships in the oceanic northern Gulf of Mexico (i.e., 200m isobath to seaward extent of the U.S. EEZ) and are summarized in Appendix IV.

- From 1991 through 1994, and from 1996 through 2001 (excluding 1998), annual surveys were conducted during spring along a fixed plankton-sampling trackline. Due to limited survey effort in any given year, the survey effort-weighted estimated average abundance of pygmy killer whales for all surveys combined was estimated. For 1991 to 1994, the estimate was 518 (CV=0.81) (Hansen et al. 1995), and for 1996 to 2001, 408 (CV=0.60) (Mullin and Fulling 2004; Table 1).
- During summer 2003 and spring 2004, surveys dedicated to estimating cetacean abundance were conducted along a grid of uniformly-spaced transect lines from a random start. The abundance estimate for pygmy killer whales, pooled from 2003 to 2004, was 323 (CV=0.60) (Mullin 2007; Table 1).

**Recent surveys and abundance estimates**

An abundance estimate for pygmy killer whales was generated from vessel surveys conducted in the northern Gulf of Mexico from the continental shelf edge (~200-m isobath) to the seaward extent of the U.S. EEZ (Garrison et al. 2020). One survey was conducted from 2 July to 25 August 2017 and consisted of 7,302 km of on-effort trackline, and the second survey was conducted from 11 August to 6 October 2018 and consisted of 6,473 km of on-effort trackline within the surveyed strata. Both surveys used a double-platform data-collection procedure, to allow...
estimation of the detection probability on the trackline using the independent observer approach assuming point independence (Laake and Borchers 2004). However, this species was observed during tracklines when only one survey team was on effort. Therefore, abundance estimates were derived using MCDS distance sampling methods that accounted for the effects of covariates (e.g., sea state, glare) on detection probability within the surveyed strip (Thomas et al. 2010) implemented in package mrds (version 2.21, Laake et al. 2020) in the R statistical programming language. The estimated detection probability on the trackline for similar species was then applied to develop the final abundance estimate. The abundance estimate for this stock included sightings of unidentified small whales that were apportioned among species based on their relative density within the survey strata (Garrison et al. 2020). The surveys were conducted in "passing mode" (e.g., Schwarz et al. 2010) while all prior surveys in the Gulf of Mexico have been conducted in "closing mode." There were sightings of pygmy killer whales in 2017 but there were none in 2018. The inverse variance weighted mean abundance estimate for pygmy killer whales in oceanic waters during 2017 and 2018 is 613 (CV=1.15; Table 1; Garrison et al. 2020). Unlike previous abundance estimates, this estimate was corrected for the probability of detection on the trackline. During summer 2009, a line-transect survey dedicated to estimating the abundance of oceanic cetaceans was conducted in the northern Gulf of Mexico. Survey lines were stratified in relation to depth and the location of the Loop Current. The abundance estimate for pygmy killer whales in oceanic waters during 2009 was 152 (CV=1.02; Table 1).

Table 1. Most recent abundance estimate (N_{est}) and coefficient of variation (CV) of northern Gulf of Mexico pygmy killer whales in oceanic waters (200 m to the offshore extent of the EEZ) based on an average from summer 2017 and summer/fall 2018 vessel surveys.

<table>
<thead>
<tr>
<th>Years</th>
<th>Area</th>
<th>N_{est}</th>
<th>CV</th>
</tr>
</thead>
<tbody>
<tr>
<td>2017, 2018</td>
<td>Gulf of Mexico</td>
<td>613</td>
<td>1.15</td>
</tr>
</tbody>
</table>

Table 1. Summary of abundance estimates for northern Gulf of Mexico pygmy killer whales. Month, year and area covered during each abundance survey, and resulting abundance estimate (N_{est}) and coefficient of variation (CV).

<table>
<thead>
<tr>
<th>Month/Year</th>
<th>Area</th>
<th>N_{est}</th>
<th>CV</th>
</tr>
</thead>
<tbody>
<tr>
<td>Apr-Jun 1991-1994</td>
<td>Oceanic waters</td>
<td>518</td>
<td>0.81</td>
</tr>
<tr>
<td>Apr-Jun 1996-2001 (excluding 1998)</td>
<td>Oceanic waters</td>
<td>408</td>
<td>0.60</td>
</tr>
<tr>
<td>Jun-Aug 2003, Apr-Jun 2004</td>
<td>Oceanic waters</td>
<td>323</td>
<td>0.60</td>
</tr>
<tr>
<td>Jun-Aug 2009</td>
<td>Oceanic waters</td>
<td>152</td>
<td>1.02</td>
</tr>
</tbody>
</table>

Minimum Population Estimate

The minimum population estimate (N_{min}) is the lower limit of the two-tailed 60% confidence interval of the log-normal distributed abundance estimate. This is equivalent to the 20th percentile of the log-normal distributed abundance estimate as specified by Wade and Angliss (1997). The best estimate of abundance for pygmy killer whales is 613\,452 (CV=1.1502). The minimum population estimate for the northern Gulf of Mexico pygmy killer whale is 283 (Table 2) pygmy killer whales.

Current Population Trend
The statistical power to detect a trend in abundance for this stock is poor due to the relatively imprecise abundance estimates and long intervals between surveys. For example, the power to detect a precipitous decline in abundance (i.e., 50% decrease in 15 years) with estimates of low precision (e.g., CV>0.30) remains below 80% (alpha=0.30) unless surveys are conducted on an annual basis (Taylor et al. 2007). In addition, because these surveys are restricted to U.S. waters, it is not possible to distinguish between changes in population size and Gulf-wide shifts in spatial distribution. Five point estimates of pygmy killer whale abundance have been made based on data from surveys during: 2003 (June–August), 2004 (April–June), 2009 (July–August), 2017 (July–August), and 2018 (August–October). Each of these surveys had a similar design and was conducted using the same vessel or a vessel with a similar observation platform. Prior year surveys only employed a single survey team. The estimates for 2017 and 2018 were conducted in “passing” mode which resulted in increased numbers of unidentified sightings and may have affected group size estimation for distant groups of dolphins and small whales. Apportioning unidentified species among identified taxa addresses the first issue, and there were no apparent relationships between sighting distance and estimated group size. The model for detection probability on the trackline from the summer 2017 survey was applied to the abundance estimates from the 2003, 2004, and 2009 surveys. The resulting abundance estimates were: 2003, N=501 (CV=0.74); 2004, N=490 (CV=0.87); 2009, N=359 (CV=0.95); 2017, N=1,226 (CV=1.15); and 2018, N=0 (CV=NA). Pairwise comparisons of the non-zero log-transformed means were conducted between years, and significant differences were assessed at alpha=0.10. P-values were adjusted for multiple comparisons. There were no significant differences between survey years.

Four point estimates of pygmy killer whale abundance have been made based on data from surveys covering 1991–2009. The estimates vary by a maximum factor of more than three. To determine whether changes in abundance have occurred over this period, an analysis of all the survey data needs to be conducted which incorporates covariates (e.g., survey conditions, season) that could potentially affect estimates. Nevertheless, differences in temporal abundance estimates will still be difficult to interpret without a Gulf of Mexico-wide understanding of pygmy killer whale abundance. The oceanography of the Gulf of Mexico is quite dynamic, and the spatial scale of the Gulf is small relative to the ability of most cetacean species to travel. Studies based on abundance and distribution surveys restricted to U.S. waters are unable to detect temporal shifts in distribution beyond U.S. waters that might account for any changes in abundance.

**CURRENT AND MAXIMUM NET PRODUCTIVITY RATES**

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

**POTENTIAL BIOLOGICAL REMOVAL**

Potential Biological Removal (PBR) is the product of the minimum population size, one half the maximum net productivity rate and a recovery factor (MMPA Sec. 3.16 U.S.C. 1362; Wade and Angliss 1997). The minimum population size is 283. The maximum productivity rate is 0.04, the default value for cetaceans. The recovery factor, which accounts for endangered, depleted, threatened stocks, or stocks of unknown status relative to optimum sustainable population (OSP), is assumed to be 0.5 because the stock is of unknown status. PBR for the northern Gulf of Mexico pygmy killer whale is 2.8 (Table 2).

**Table 2. Best and minimum abundance estimates for the northern Gulf of Mexico pygmy killer whale with Maximum Productivity Rate (Rmax), Recovery Factor (Fr) and PBR.**

<table>
<thead>
<tr>
<th>Nbest</th>
<th>CV</th>
<th>Nmin</th>
<th>Fr</th>
<th>Rmax</th>
<th>PBR</th>
</tr>
</thead>
<tbody>
<tr>
<td>613</td>
<td>1.15</td>
<td>283</td>
<td>0.5</td>
<td>0.04</td>
<td>2.8</td>
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</table>

**ANNUAL HUMAN-CAUSED MORTALITY AND SERIOUS INJURY**

Total annual estimated fishery-related mortality and serious injury to this stock during 2014–2018 was presumed to be zero, as there were no reports of mortalities or serious injuries to pygmy killer whales in the Gulf of Mexico (Table 3). Mean annual mortality and serious injury during 2014–2018 due to other human-caused actions (the Deepwater Horizon oil spill) was predicted to be 1.6. The minimum total mean annual human-caused mortality and serious injury for this stock during 2014–2018 was, therefore, 1.6. There has been no reported fishing-related mortality of a pygmy killer whale during 1998-2010 (Yeung 1999; Yeung 2001; Garrison 2003; Garrison and Richards 2004;
Table 3. Total annual estimated fishery-related mortality and serious injury for the northern Gulf of Mexico pygmy killer whale.

<table>
<thead>
<tr>
<th>Years</th>
<th>Source</th>
<th>Annual Avg.</th>
<th>CV</th>
</tr>
</thead>
<tbody>
<tr>
<td>2014–2018</td>
<td>U.S. fisheries using observer data</td>
<td>0</td>
<td></td>
</tr>
</tbody>
</table>

Fisheries Information

There are two commercial fisheries that interact, or that potentially could interact, with this stock in the Gulf of Mexico. These are the Category I Atlantic Highly Migratory Species (high seas) longline fishery and the Category I Atlantic Ocean, Caribbean, Gulf of Mexico large pelagics longline fishery (Appendix III). Percent observer coverage (percentage of sets observed) for these longline fisheries for each year during 2014–2018 was 18, 19, 23, 13 and 20, respectively. There is very little effort within the Gulf of Mexico by the high seas longline fishery, and no takes of pygmy killer whales within high seas waters of the Gulf of Mexico have been observed or reported thus far. Pelagic swordfish, tunas and billfish are the targets of the large pelagic longline fishery operating in the northern Gulf of Mexico. During 2014–2018 there were no observed mortalities or serious injuries to pygmy killer whales by this fishery (Garrison and Stokes 2016; 2017; 2019; 2020; in review). There has historically been some take of this species in small cetacean fisheries in the Caribbean (Caldwell and Caldwell 1971). The commercial fishery which potentially could interact with this stock in the Gulf of Mexico is the Atlantic Ocean, Caribbean, Gulf of Mexico large pelagic longline fishery (Appendix III). Pelagic swordfish, tunas and billfish are the targets of the large pelagic longline fishery operating in the northern Gulf of Mexico. There were no reports of mortality or serious injury to pygmy killer whales by this fishery during 1998–2010 (Yeung 1999; Yeung 2001; Garrison 2003; Garrison and Richards 2004; Garrison 2005; Fairfield Walsh and Garrison 2006; Fairfield Walsh and Garrison 2007; Fairfield and Garrison 2008; Garrison et al. 2009; Garrison and Stokes 2010; 2011).

Other Mortality

There were 4seven reported strandings of pygmy killer whales in the Gulf of Mexico during 2006–2014–2018 (Table 4; NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 21 May 2019 46 November 2014). Four pygmy killer whales stranded in 2008 (2 animals mass stranded in Florida, 1 in Mississippi, 1 in Texas). Evidence of human interaction was detected for 1 of these stranded animals. A plastic, office sheet protector was crumpled up and lodged in the esophagus of the animal. No evidence of human interaction was detected for three stranded animals, and for the remaining 2 four stranded animals, it could not be determined if there was evidence of human interactions and for the remaining stranded animal, 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 whales that die or are seriously injured in human interactions wash ashore, or, if they do, they are not all recovered (Peltier et al. 2012; Wells et al. 2015). In particular, oceanic stocks in the Gulf of Mexico are less likely to strand than nearshore coastal stocks or shelf stocks (Williams et al. 2011). Additionally, not all carcasses will show evidence of human interaction, entanglement or other fishery-related interaction due to decomposition, scavenger damage, etc. (Byrd et al. 2014). Finally, the level of technical expertise among stranding network personnel varies widely as does the ability to recognize signs of human interaction. Stranding data probably underestimate the extent of fishery-related mortality and serious injury because not all of the marine mammals which die or are seriously injured in fishery interactions wash ashore, not all that wash ashore are discovered, reported or investigated, 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 interactions.

An Unusual Mortality Event (UME) was declared for cetaceans in the northern Gulf of Mexico beginning 1 February 2010 and ending 31 July 2014 (Litz et al. 2014; https://www.fisheries.noaa.gov/national/marine-life-distress/2010-2014-cetacean-unusual-mortality事件-northern-gulf-mexico). As of early 2012, the event is still ongoing. It includes cetaceans that stranded prior to the Deepwater Horizon (DWH) oil spill (see “Habitat Issues” below), during the spill, and after. Exposure to the DWH oil spill was determined to be the primary underlying cause of the elevated stranding numbers in the northern Gulf of Mexico after the spill (e.g., Schwacke et al. 2014; Venn-Watson et al. 2015; Colegrove et al. 2016; DWH NRDAT 2016; see Habitat Issues section). However, there were no
pygmy killer whale strandings recovered within the spatial and temporal boundaries of this UME. During 2010, no animals from this stock were considered to be part of the UME.

A population model was developed to estimate the injury and time to recovery for stocks affected by the DWH oil spill, taking into account long-term effects resulting from mortality, reproductive failure, reduced survival rates, and the proportion of the stock exposed to DWH oil (DWH MMIQT 2015). Overall, the model estimated that this stock experienced a 7% maximum reduction in population size due to the oil spill (DWH MMIQT 2015). The mortality projected for the years 2010–2013 due to the spill has not been reported previously. Based on the population model, it was projected that 16 pygmy killer whales died during 2010–2013 (four year annual average of 3.9) due to elevated mortality associated with oil exposure (see Appendix VI). For the 2014–2018 reporting period of this SAR, the population model estimated 8.1 pygmy killer whales died due to elevated mortality associated with oil exposure. The population model used to predict pygmy killer whale mortality due to the DWH event has a number of sources of uncertainty. Model parameters (e.g., survival rates, reproductive rates, and life-history parameters) were derived from literature sources for pygmy killer whales occupying waters outside of the Gulf of Mexico. In addition, proxy values for the effects of DWH oil exposure on both survival rates and reproductive success were applied based upon estimated values for common bottlenose dolphins in Barataria Bay. Finally, there was no estimation of uncertainty in model parameters or outputs.

Table 4. Pygmy killer whale strandings along the northern Gulf of Mexico coast, 2014–2018. Data are from the NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 21 May 2019. There were no strandings of pygmy killer whales in Alabama, Louisiana, or Texas.

<table>
<thead>
<tr>
<th>Area</th>
<th>2014</th>
<th>2015</th>
<th>2016</th>
<th>2017</th>
<th>2018</th>
<th>Total</th>
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<td>0</td>
<td>1</td>
<td>1</td>
<td>0</td>
<td>3</td>
<td>5</td>
</tr>
<tr>
<td>Mississippi</td>
<td>0</td>
<td>2</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>2</td>
</tr>
<tr>
<td>TOTAL</td>
<td>0</td>
<td>3</td>
<td>1</td>
<td>0</td>
<td>3</td>
<td>7</td>
</tr>
</tbody>
</table>

HABITAT ISSUES

The Deepwater Horizon (DWH) MC252 drilling platform, located approximately 50 miles 80 km southeast of the Mississippi River Delta in waters about 1,500 m deep, exploded on 20 April 2010. The rig sank, and for over 87 days ~3.2 million barrels of oil and gas were discharged from the wellhead until it was capped on 15 July 2010 (DWH NRDAT 2016). During the response effort dispersants were applied extensively at the seafloor and at the sea surface (Lehr et al. 2010; OSAT 2010). In-situ burning, or controlled burning of oil at the surface, was also used extensively as a response tool (Lehr et al. 2010). The oil, dispersant and burn residue compounds present ecological concerns. The magnitude of this oil spill was unprecedented in U.S. history, causing impacts to wildlife, natural habitats and human communities along coastal areas from western Louisiana to the Florida Panhandle (NOAA 2011). It could be years before the entire scope of damage is ascertained (NOAA 2011).

Shortly after the oil spill, the Natural Resource Damage Assessment (NRDA) process was initiated under the Oil Pollution Act of 1990. A variety of NRDA research studies were conducted to determine potential impacts of the spill on marine mammals. These studies estimated that 15% (95% CI: 7–33) of pygmy killer whales in the Gulf were exposed to oil, that 7% (95% CI: 3–10) of females suffered from reproductive failure, and 6% (95% CI: 2–9) of pygmy killer whales suffered adverse health effects (DWH MMIQT 2015). A population model estimated the stock experienced a maximum 7% reduction in population size (see Other Mortality section above).

Anthropogenic sound in the world’s oceans has been shown to affect marine mammals, with vessel traffic, seismic surveys, and active naval sonars being the main anthropogenic contributors to low- and mid-frequency noise in oceanic waters (e.g., Nowacek et al. 2015; Gomez et al. 2016; NMFS 2018). The long-term and population consequences of these impacts are less well-documented and likely vary by species and other factors. Impacts on marine mammal prey from sound are also possible (Carroll et al. 2017), but the duration and severity of any such prey effects on marine mammals are unknown. These studies have focused on identifying the type, magnitude, severity, length and impact of oil exposure to oceanic, coastal and estuarine marine mammals. The research is ongoing and likely will continue for some time. For continental shelf and oceanic cetaceans, the NOAA led efforts include: aerial surveys to document the distribution, abundance, species and exposure of marine mammals and turtles relative to oil from DWH spill; and ship surveys to evaluate
exposure to oil and other chemicals and to assess changes in animal behavior and distribution relative to oil exposure through visual and acoustic surveys, deployment of passive acoustic monitoring systems, collection of tissue samples, and deployment of satellite tags on sperm and Bryde’s whales.

Aerial surveys have observed Risso’s dolphins, spinner dolphins, pantropical spotted dolphins, striped dolphins, bottlenose dolphins and sperm whales swimming in oil in offshore waters (NOAA 2010a). The effects of oil exposure on marine mammals depend on a number of factors including the type and mixture of chemicals involved, the amount, frequency and duration of exposure, the route of exposure (inhaled, ingested, absorbed, or external) and biomedical risk factors of the particular animal (Geraci 1990; NOAA 2010b). In general, direct external contact with petroleum compounds or dispersants with skin may cause skin irritation, chemical burns and infections. Inhalation of volatile petroleum compounds or dispersants may irritate or injure the respiratory tract, which could lead to pneumonia or inflammation. Ingestion of petroleum compounds may cause injury to the gastrointestinal tract, which could affect an animal’s ability to digest or absorb food. Absorption of petroleum compounds or dispersants may damage kidney, liver and brain function in addition to causing immune suppression and anemia. Long term chronic effects such as lowered reproductive success and decreased survival may occur (Geraci 1990; NOAA 2010b).

STATUS OF STOCK

Pygmy killer whales are not listed as threatened or endangered under the Endangered Species Act, and the northern Gulf of Mexico stock is not considered strategic under the Marine Mammal Protection Act. No fishery-related mortality or serious injury has been observed in recent years; therefore, total fishery-related mortality and serious injury can be considered insignificant and approaching the zero mortality and serious injury rate. The status of pygmy killer whales in the northern Gulf of Mexico, relative to OSP, is unknown. There was no statistically significant trend in population size for this stock in the northern Gulf of Mexico. The species is not listed as threatened or endangered under the Endangered Species Act. There are insufficient data to determine the population trends for this species. Total human-caused mortality and serious injury for this stock is not known but none has been documented. There is insufficient information available to determine whether the total fishery-related mortality and serious injury for this stock is insignificant and approaching zero mortality and serious injury rate. This is not a strategic stock because it is assumed that the average annual human-related mortality and serious injury does not exceed PBR.

REFERENCES


DWH MMIQT. 2015. Models and analyses for the quantification of injury to Gulf of Mexico cetaceans from the Deepwater Horizon Oil Spill, MM TR.01 Schwacke Quantification of Injury to GOM.Cetaceans. Southeast Fisheries Science Center, Protected Resources and Biodiversity Division, 75 Virginia Beach Dr., Miami, Florida 33140. PRBD Contribution #: PRBD-2020-02. Available from:


Garrison, L.P. and L. Stokes. in review. Estimated bycatch of marine mammals and sea turtles in the U.S. Atlantic pelagic longline fleet during 2018. Southeast Fisheries Science Center, Protected Resources and Biodiversity Division, 75 Virginia Beach Dr., Miami, Florida 33140. PRD Contribution # PRD-2020-08. 55 pp.


DWARF SPERM WHALE (*Kogia sima*): Northern Gulf of Mexico Stock

**STOCK DEFINITION AND GEOGRAPHIC RANGE**

The dwarf sperm whale *appears to be distributed worldwide in temperate to tropical waters* (Caldwell and Caldwell 1989). Sightings of these animals in the northern Gulf of Mexico (i.e., U.S. Gulf of Mexico) occur primarily in oceanic waters (Figure 1; Mullin *et al.* 1991; Mullin and Fulling 2004; Maze-Foley and Mullin 2006). Dwarf sperm whales and pygmy sperm whales (*Kogia breviceps*) are often difficult to differentiate at sea (Caldwell and Caldwell 1989; Bloodworth and Odell 2008; McAlpine 2009) unless sighting conditions are ideal, and sightings of either species are usually categorized as *Kogia* spp. In addition, the acoustic signals of dwarf and pygmy sperm whales also cannot be distinguished from each other at this time (Merkens *et al.* 2018) adding to the difficulties of identification at sea.

In the northern Gulf of Mexico, *Kogia* spp. are sighted in waters >200 m, over the continental slope and deep basin. They have been seen in all seasons during aerial and vessel surveys of the northern Gulf of Mexico (Hansen *et al.* 1996; Mullin and Hoggard 2000; Maze-Foley and Mullin 2006; Garrison and Aichinger Dias 2020; Figure 1). All the cetacean species found in the oceanic northern Gulf of Mexico almost certainly occur in similar habitat beyond U.S. boundaries in the southern Gulf. There are fewer cetacean sighting and stranding records in the southern Gulf due to more limited effort. Nevertheless there are records for most oceanic species in the southern Gulf (e.g., Jefferson and Schiro 1997; Ortega Ortiz 2002; Ortega-Argueta *et al.* 2005; Jefferson *et al.* 2008; Vázquez Castán *et al.* 2009; Whitt *et al.* 2011). This is therefore likely a transboundary stock with Cuba and/or Mexico. Because U.S. waters only comprise about 40% of the entire Gulf of Mexico and 35% of the oceanic (i.e., >200 m) Gulf of Mexico (Mullin and Fulling 2004), abundance and stock boundaries of oceanic species are poorly known.

Sightings of this category were documented in all seasons during GulfCet aerial surveys of the northern Gulf of Mexico from 1992 to 1998 (Hansen *et al.* 1996; Mullin and Hoggard 2000). The difficulty in sighting dwarf and pygmy sperm whales may be exacerbated by their avoidance reaction towards ships, and change in behavior towards approaching survey aircraft (Würsig *et al.* 1998).

In a study using hematological and stable isotope data, Barros *et al.* (1998) speculated that dwarf sperm whales may have a more pelagic distribution than pygmy sperm whales and/or dive deeper during feeding bouts. Diagnostic
Morphological characters have also been useful in distinguishing the two *Kogia* species (Barros and Duffield 2003), thus enabling researchers to use stranding data in distributional and ecological studies. Specifically, the distance from the snout to the center of the blowhole in proportion to the animal’s total length, as well as the height of the dorsal fin, in proportion to the animal’s total length, can be used to differentiate between the two *Kogia* species when such measurements are obtainable (Barros and Duffield 2003).

Although there are only a few records from Gulf of Mexico waters beyond U.S. boundaries (e.g., Ortega Ortiz 2002), dwarf sperm whales almost certainly occur throughout the oceanic Gulf of Mexico (Jefferson *et al.* 2008), which is also composed of waters belonging to Mexico and Cuba where there is currently little information on cetacean species abundance and distribution. U.S. waters only comprise about 40% of the entire Gulf of Mexico, and 65% of oceanic waters are south of the U.S. Exclusive Economic Zone (EEZ).

Dwarf sperm whales in the northern Gulf of Mexico are managed separately from those in the western North Atlantic. Although there have been no directed studies of the degree of demographic independence between the two areas, this management structure is consistent with the fact that the Gulf of Mexico and western North Atlantic belong to distinct marine ecoregions (Spalding *et al.* 2007; Moore and Merrick 2011). There are insufficient data to determine whether the northern Gulf of Mexico stock comprises multiple demographically independent populations. Additional morphological, acoustic, genetic, and/or behavioral data are needed to further delineate population structure within the Gulf of Mexico and across the broader geographic area. The Gulf of Mexico population is provisionally being considered a separate stock for management purposes, although there is currently no information to differentiate this stock from the Atlantic Ocean stock(s). Additional morphological, genetic and/or behavioral data are needed to provide further information on stock delineation.

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

**Population Size**

The best abundance estimate (Nbest) available for northern Gulf of Mexico dwarf and pygmy sperm whales combined is 336 (CV=0.35; Table 1) 186 (CV=1.04; Table 1). This estimate is from summer 2017 and summer/fall 2018 oceanic surveys covering waters from the 200-m isobath to the seaward extent of the U.S. EEZ (Garrison *et al.* 2020). This estimate was not corrected for the probability of detection on the trackline, and is likely a severe underestimate because *Kogia* spp. are often difficult to see, present little of themselves at the surface, do not fluke when they dive, and have long dive times. This estimate is from a summer 2009 oceanic surveys covering waters from the 200-m isobath to the seaward extent of the U.S. EEZ. In addition, they exhibit avoidance behavior towards ships and changes in behavior towards approaching survey aircraft (Würsig *et al.* 1998).

Earlier abundance estimates

Please see Appendix IV for a summary of abundance estimates, including earlier estimates and survey descriptions. All estimates of abundance were derived through the application of distance sampling analysis (Buckland *et al.* 2001) and the computer program DISTANCE (Thomas *et al.* 1998) to line transect survey data collected from ships in the oceanic northern Gulf of Mexico (i.e., 200m isobath to seaward extent of the U.S. EEZ) and are summarized in Appendix IV.

From 1991 through 1994, and from 1996 through 2001 (excluding 1998), annual surveys were conducted during spring along a fixed plankton-sampling trackline. Due to limited survey effort in any given year, the survey effort-weighted estimated average abundance of dwarf and pygmy sperm whales for all surveys combined was estimated. For 1991 to 1994, the estimate was 547 (CV=0.28) (Hansen *et al.* 1995), and for 1996 to 2001, 742 (CV=0.29) (Mullin and Fulling 2004; Table 1). A separate estimate of abundance for dwarf sperm whales could not be estimated due to uncertainty of species identification at sea.
During summer 2003 and spring 2004, surveys dedicated to estimating cetacean abundance were conducted along a grid of uniformly spaced transect lines from a random start. The estimate of abundance for dwarf and pygmy sperm whales in oceanic waters, pooled from 2003 to 2004, was 453 (CV=0.35) (Mullin 2007; Table 1).

**Recent surveys and abundance estimates**

An abundance estimate for dwarf and pygmy sperm whales combined was generated from vessel surveys conducted in the northern Gulf of Mexico from the continental shelf edge (~200-m isobath) to the seaward extent of the U.S. EEZ (Garrison et al. 2020). One survey was conducted from 2 July to 25 August 2017 and consisted of 7,302 km of on-effort trackline, and the second survey was conducted from 11 August to 6 October 2018 and consisted of 6,473 km of on-effort trackline within the surveyed strata. Both surveys used a double-platform data-collection procedure to allow estimation of the detection probability on the trackline using the independent observer approach assuming point independence (Laake and Borchers 2004). However, there were too few sightings and too few resightings of these species to allow estimation of detection probability on the trackline. Therefore, abundance estimates were derived using MCDS distance sampling methods that accounted for the effects of covariates (e.g., sea state, glare) on detection probability within the surveyed strip (Thomas et al. 2010) implemented in package mrds (version 2.21, Laake et al. 2020) in the R statistical programming language. The surveys were conducted in "passing mode" (e.g., Schwarz et al. 2010) while all prior surveys in the Gulf of Mexico have been conducted in "closing mode." The inverse variance weighted mean abundance estimate for dwarf and pygmy sperm whales in oceanic waters during 2017 and 2018 was 336 (CV=0.35; Table 1; Garrison et al. 2020). This estimate was not corrected for the probability of detection on the trackline, and is likely a severe underestimate due to the long dive times of these species. During summer 2009, a line-transect survey dedicated to estimating the abundance of oceanic cetaceans was conducted in the northern Gulf of Mexico. Survey lines were stratified in relation to depth and the location of the Loop Current. The abundance estimate for dwarf and pygmy sperm whales in oceanic waters during 2009 was 186 (CV=1.04; Table 1).

<table>
<thead>
<tr>
<th>Month/Year</th>
<th>Area</th>
<th>Nbest</th>
<th>CV</th>
</tr>
</thead>
<tbody>
<tr>
<td>Apr-Jun 1991–1994</td>
<td>Oceanic waters</td>
<td>547</td>
<td>0.28</td>
</tr>
<tr>
<td>Apr-Jun 1996-2001 (excluding 1998)</td>
<td>Oceanic waters</td>
<td>742</td>
<td>0.29</td>
</tr>
<tr>
<td>Jun-Aug 2003, Apr-Jun 2004</td>
<td>Oceanic waters</td>
<td>453</td>
<td>0.35</td>
</tr>
<tr>
<td>Jun-Aug 2009</td>
<td>Oceanic waters</td>
<td>186</td>
<td>1.04</td>
</tr>
</tbody>
</table>

Table 1. Summary of combined abundance estimates for northern Gulf of Mexico dwarf and pygmy sperm whales. Month, year and area covered during each abundance survey, and resulting abundance estimate (Nbest) and coefficient of variation (CV).
**Minimum Population Estimate**

The minimum population estimate \(N_{\text{min}}\) is the lower limit of the two-tailed 60% confidence interval of the log-normal distributed abundance estimate. This is equivalent to the 20th percentile of the log-normal distributed abundance estimate as specified by Wade and Angliss (1997). The best estimate of abundance for dwarf and pygmy sperm whales is \(336,486\) (CV=0.354.04). It is not possible to determine the minimum population estimate for only dwarf sperm whales. The minimum population estimate for the northern Gulf of Mexico dwarf and pygmy sperm whales is \(253,900\) (Table 2) dwarf and pygmy sperm whales.

**Current Population Trend**

The statistical power to detect a trend in abundance for this stock is poor due to the relatively imprecise abundance estimates and long intervals between surveys. For example, the power to detect a precipitous decline in abundance (i.e., 50% decrease in 15 years) with estimates of low precision (e.g., CV=0.30) remains below 80% (alpha = 0.30) unless surveys are conducted on an annual basis (Taylor et al. 2007). In addition, because these surveys are restricted to U.S. waters, it is not possible to distinguish between changes in population size and Gulf-wide shifts in spatial distribution. Five point estimates of Kogia spp. abundance have been made based on data from surveys during 2003 (June–August), 2004 (April–June), 2009 (July–August), 2017 (July–August), and 2018 (August–October). Each of these surveys had a similar design and was conducted using the same vessel or a vessel with a similar observation platform. Prior year surveys only employed a single survey team. The estimates for 2017 and 2018 were conducted in “passing” mode which resulted in increased numbers of unidentified sightings and may have affected group size estimation for distant groups of dolphins and small whales. Apportioning unidentified species among identified taxa addresses the first issue, and there were no apparent relationships between sighting distance and estimated group size. The model for detection probability on the trackline from the summer 2017 survey was applied to the abundance estimates from the 2003, 2004, and 2009 surveys. The resulting abundance estimates were: 2003, \(N=441\) (CV=0.42); 2004, \(N=38\) (CV=0.71); 2009, \(N=124\) (CV=0.60); 2017, \(N=293\) (CV=0.59); and 2018, \(N=358\) (CV=0.42). Pairwise comparisons of the log-transformed means were conducted between years, and significant differences were assessed at alpha=0.10. P-values were adjusted for multiple comparisons. There was a significant decrease between the 2003 and 2004 estimates (p.adjusted=0.006), and a significant increase between the 2004 estimates and both the 2017 (p.adjusted=0.063) and 2018 (p.adjusted=0.014) estimates; however there is no clear pattern in the overall trend. There are insufficient data to determine the population trends for this species due to uncertainty in species identification at sea. Four point estimates of Kogia spp. abundance have been made based on data from surveys covering 1991-2009. The estimates vary by a maximum factor of nearly four. To determine whether changes in abundance have occurred over this period, an analysis of all the survey data needs to be conducted which incorporates covariates (e.g., survey conditions, season) that could potentially affect estimates. Nevertheless, differences in temporal abundance estimates will still be difficult to interpret without a Gulf of Mexico wide understanding of Kogia abundance. The oceanography of the Gulf of Mexico is quite dynamic, and the spatial scale of the Gulf is small relative to the ability of most cetacean species to travel. Studies based on abundance and distribution surveys restricted to U.S. waters are unable to detect temporal shifts in distribution beyond U.S. waters that might account for any changes in abundance.

**CURRENT AND MAXIMUM NET PRODUCTIVITY RATES**

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

**POTENTIAL BIOLOGICAL REMOVAL**

Potential Biological Removal (PBR) is the product of the minimum population size, one half the maximum net productivity rate and a recovery factor (MMPA Sec. 3.16 U.S.C. 1362; Wade and Angliss 1997). The minimum population size for dwarf and pygmy sperm whales is \(253,900\). The maximum productivity rate is 0.04, the default value for cetaceans. The recovery factor, which accounts for endangered, depleted, threatened stocks, or stocks of unknown status relative to optimum sustainable population (OSP), is assumed to be 0.5 because the stock is of unknown status. PBR for the northern Gulf of Mexico dwarf and pygmy sperm whales is \(2.5\) (Table 2)0.9. It is not possible to determine the PBR for only dwarf sperm whales.
Table 2. Best and minimum abundance estimates for northern Gulf of Mexico dwarf and pygmy sperm whales with Maximum Productivity Rate (Rmax), Recovery Factor (Fr) and PBR.

<table>
<thead>
<tr>
<th>Nbest</th>
<th>CV</th>
<th>Nmin</th>
<th>Fr</th>
<th>Rmax</th>
<th>PBR</th>
</tr>
</thead>
<tbody>
<tr>
<td>336</td>
<td>0.35</td>
<td>253</td>
<td>0.5</td>
<td>0.04</td>
<td>2.5</td>
</tr>
</tbody>
</table>

**ANNUAL HUMAN-CAUSED MORTALITY AND SERIOUS INJURY**

Total annual estimated fishery-related mortality and serious injury to this stock during 2014–2018 was presumed to be zero, as there were no reports of mortalities or serious injuries to dwarf or pygmy sperm whales in the Gulf of Mexico (Table 3). Mean annual mortality and serious injury during 2014–2018 for dwarf and pygmy sperm whales due to other human-caused actions (the Deepwater Horizon oil spill) was predicted to be 31. The minimum total mean annual human-caused mortality and serious injury for dwarf and pygmy sperm whales during 2014–2018 was, therefore, 31. The minimum total mean annual human-caused mortality and serious injury for dwarf sperm whales is unknown. There has been no reported fishing-related mortality of dwarf sperm whales during 1998-2010 (Yeung 1999; Yeung 2001; Garrison 2003; Garrison and Richards 2004; Garrison 2005; Fairfield-Walsh and Garrison 2006; Fairfield-Walsh and Garrison 2007; Fairfield and Garrison 2008; Garrison et al. 2009; Garrison and Stokes 2010; 2011).

Table 3. Total annual estimated fishery-related mortality and serious injury for northern Gulf of Mexico dwarf and pygmy sperm whales.

<table>
<thead>
<tr>
<th>Years</th>
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<tbody>
<tr>
<td>2014–2018</td>
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<td>0</td>
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**Fisheries Information**

There are two commercial fisheries that interact, or that could potentially interact, with this stock in the Gulf of Mexico. These are the Category I Atlantic Highly Migratory Species (high seas) longline fishery and the Category I Atlantic Ocean, Caribbean, Gulf of Mexico large pelagics longline fishery (Appendix III). Percent observer coverage (percentage of sets observed) for these longline fisheries for each year during 2014–2018 was 18, 19, 23, 13 and 20, respectively. There is very little effort within the Gulf of Mexico by the Atlantic Highly Migratory Species (high seas) longline fishery, and no takes of pygmy or dwarf sperm whales within high seas waters of the Gulf of Mexico have been observed or reported thus far. Pelagic swordfish, tunas and billfish are the targets of the large pelagics longline fishery operating in the northern Gulf of Mexico. During 2014–2018 there were no observed mortalities or serious injuries to dwarf or pygmy sperm whales by this fishery (Garrison and Stokes 2016; 2017; 2019; 2020; in review). The commercial fishery which potentially could interact with this stock in the Gulf of Mexico is the Atlantic Ocean, Caribbean, Gulf of Mexico large pelagic longline fishery (Appendix III). Pelagic swordfish, tunas and billfish are the targets of the longline fishery operating in the northern Gulf of Mexico. There were no reports of mortality or serious injury to dwarf sperm whales by this fishery during 1998-2010 (Yeung 1999; Yeung 2001; Garrison 2003; Garrison and Richards 2004; Garrison 2005; Fairfield-Walsh and Garrison 2006; Fairfield-Walsh and Garrison 2007; Fairfield and Garrison 2008; Garrison et al. 2009; Garrison and Stokes 2010; 2011).

**Other Mortality**

At least nine dwarf sperm whale strandings were documented in the northern Gulf of Mexico during 2006–2018 (Table 4; NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 21 May 2019; 16 November 2019). Evidence of human interactions was detected for 1 of these stranded animals; no evidence of human interactions was detected for 6 one animals; for the remaining eight animals, it could not be determined if there was evidence of human interactions. An additional 10 Kogia spp. stranded during 2006–2014–2018. No evidence of human interactions was detected for 4 one of the Kogia spp. strandings; it could not be determined if there was evidence of human interactions for the remaining 2 nine Kogia spp. strandings. Stranding data probably underestimate the extent of human and fishery-related mortality and serious injury because not all of the whales that die or are seriously injured in human interactions wash ashore, or, if they do, they are not all recovered (Peltier et al. 2012; Wells et al. 2015). In particular, oceanic stocks in the Gulf of Mexico are less likely to strand than nearshore coastal stocks or shelf stocks (Williams et al. 2011). Additionally, not all carcasses will show evidence of human interaction, entanglement or other fishery-related interaction due to decomposition, scavenger
damage, etc. (Byrd et al. 2014). Finally, the level of technical expertise among stranding network personnel varies widely as does the ability to recognize signs of human interaction. Stranding data probably underestimate the extent of fishery-related mortality and serious injury because not all of the marine mammals which die or are seriously injured in fishery interactions wash ashore, not all that wash ashore are discovered, reported or investigated, 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 interactions.

An Unusual Mortality Event (UME) was declared for cetaceans in the northern Gulf of Mexico beginning 1 February March 2010 and ending 31 July 2014 (Litz et al. 2014; https://www.fisheries.noaa.gov/national/marine-life-distress/2010-2014-cetacean-unusual-mortality-event-northern-gulf-mexico) and, as of early 2012, the event is still ongoing. It included cetaceans that stranded prior to the Deepwater Horizon (DWH) oil spill (see “Habitat Issues” below), during the spill, and after. Exposure to the DWH oil spill was determined to be the primary underlying cause of the elevated stranding numbers in the northern Gulf of Mexico after the spill (e.g., Schwacke et al. 2014; Venn-Watson et al. 2015; Colegrove et al. 2016; DWH NRDAT 2016; see Habitat Issues section). Four dwarf sperm whale strandings were considered to be part of this UME, one of which occurred during 2014. During 2010, no animals from this stock were considered to be part of the UME.

A population model was developed to estimate the injury and time to recovery for stocks affected by the DWH oil spill, taking into account long-term effects resulting from mortality, reproductive failure, reduced survival rates, and the proportion of the stock exposed to DWH oil (DWH MMIQT 2015). Overall, the model estimated that dwarf and pygmy sperm whale stocks experienced a 6% maximum reduction in population size due to the oil spill (DWH MMIQT 2015). The mortality projected for the years 2010–2013 due to the spill has not been reported previously. Based on the population model, it was projected that 340 dwarf and pygmy sperm whale died during 2010–2013 (four year annual average of 85) due to elevated mortality associated with oil exposure (see Appendix VI). For the 2014–2018 reporting period of this SAR, the population model estimated 154 dwarf and pygmy sperm whales died due to elevated mortality associated with oil exposure. However, this mortality estimate is not comparable to the current abundance estimate derived from visual surveys because the population model included a correction factor for detection probability derived from acoustic density estimates (DWH MMIQT 2015). The population model used to predict dwarf/pygmy sperm whale mortality due to the DWH event has a number of sources of uncertainty. Model parameters (e.g., survival rates, reproductive rates, and life-history parameters) were derived from literature sources for dwarf/pygmy sperm whales occupying waters outside of the Gulf of Mexico. In addition, proxy values for the effects of DWH oil exposure on both survival rates and reproductive success were applied based upon estimated values for common bottlenose dolphins in Barataria Bay. Finally, there was no estimation of uncertainty in model parameters or outputs.

Table 4. Dwarf and pygmy sperm whale (Kogia sima (Ks), Kogia breviceps (Kb) and Kogia sp. (Sp)) strandings along the northern Gulf of Mexico coast, 2014–2018. Strandings that were not reported to species have been reported as Kogia spp. The level of technical expertise among stranding network personnel varies, and given the potential difficulty in correctly identifying stranded Kogia whales to species, reports to specific species should be viewed with caution.

<table>
<thead>
<tr>
<th>State</th>
<th>2014</th>
<th>2015</th>
<th>2016</th>
<th>2017</th>
<th>2018</th>
<th>Total</th>
</tr>
</thead>
<tbody>
<tr>
<td>Alabama</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
</tr>
<tr>
<td>Florida</td>
<td>2</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>2</td>
</tr>
<tr>
<td>Louisiana</td>
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<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>1</td>
</tr>
<tr>
<td>Mississippi</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
</tr>
<tr>
<td>Texas</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
</tr>
<tr>
<td>TOTAL</td>
<td>3</td>
<td>2</td>
<td>4</td>
<td>2</td>
<td>2</td>
<td>10</td>
</tr>
</tbody>
</table>
Table 2. Dwarf sperm whale (*Kogia sima*) strandings along the northern Gulf of Mexico coast, 2006–2010.

<table>
<thead>
<tr>
<th>State</th>
<th>2006</th>
<th>2007</th>
<th>2008</th>
<th>2009</th>
<th>2010</th>
<th>Total</th>
</tr>
</thead>
<tbody>
<tr>
<td>Alabama</td>
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<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
</tr>
<tr>
<td>Florida</td>
<td>2&lt;sup&gt;a,b&lt;/sup&gt;</td>
<td>2</td>
<td>0</td>
<td>1&lt;sup&gt;f&lt;/sup&gt;</td>
<td>0</td>
<td>5</td>
</tr>
<tr>
<td>Louisiana</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
</tr>
<tr>
<td>Mississippi</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
</tr>
<tr>
<td>Texas</td>
<td>0</td>
<td>2&lt;sup&gt;+&lt;/sup&gt;</td>
<td>2&lt;sup&gt;d&lt;/sup&gt;</td>
<td>0</td>
<td>0</td>
<td>4</td>
</tr>
<tr>
<td>TOTAL</td>
<td>2</td>
<td>4</td>
<td>2</td>
<td>1</td>
<td>0</td>
<td>9</td>
</tr>
</tbody>
</table>

<sup>a</sup> 1 additional *Kogia* sp. stranded  
<sup>b</sup> Previously reported incorrectly as 1 stranded animal  
<sup>c</sup> Mass stranding of 2 animals in August 2007  
<sup>d</sup> A mom/calf pair stranding together  
<sup>e</sup> 1 *Kogia* sp. stranded  
<sup>f</sup> 1 additional *Kogia* sp. stranded

**HABITAT ISSUES**

The Deepwater Horizon (DWH) MC252 drilling platform, located approximately 80 km southeast of the Mississippi River Delta in waters about 1,500 m deep, exploded on 20 April 2010. The rig sank, and for over 87 days ~3.2 millions of barrels of oil and gas were discharged from the wellhead until it was capped on 15 July 2010 (DWH NRDAT 2016). During the response effort dispersants were applied extensively at the seafloor and at the sea surface (Lehr et al. 2010; OSAT 2010). In-situ burning, or controlled burning of oil at the surface, was also used extensively as a response tool (Lehr et al. 2010). The oil, dispersant and burn residue compounds present ecological concerns. The magnitude of this oil spill was unprecedented in U.S. history, causing impacts to wildlife, natural habitats and human communities along coastal areas from western Louisiana to the Florida Panhandle (NOAA 2011). It could be years before the entire scope of damage is ascertained (NOAA 2011).
Shortly after the oil spill, the Natural Resource Damage Assessment (NRDA) process was initiated under the Oil Pollution Act of 1990. A variety of NRDA research studies were conducted to determine potential impacts of the spill on marine mammals. These studies estimated that 15% (95% CI: 8–29) of dwarf/pygmy sperm whales in the Gulf were exposed to oil, that 7% (95% CI: 3–10) of females suffered from reproductive failure, and 6% (95% CI: 2–9) of dwarf/pygmy sperm whales suffered adverse health effects (DWH MMIQT 2015). A population model estimated the stocks experienced a maximum 6% reduction in population size (see Other Mortality section above).

Anthropogenic sound in the world’s oceans has been shown to affect marine mammals, with vessel traffic, seismic surveys, and active naval sonars being the main anthropogenic contributors to low- and mid-frequency noise in oceanic waters (e.g., Nowacek et al. 2015; Gomez et al. 2016; NMFS 2018). The long-term and population consequences of these impacts are less well-documented and likely vary by species and other factors. Impacts on marine mammal prey from sound are also possible (Carroll et al. 2017), but the duration and severity of any such prey effects on marine mammals are unknown. These studies have focused on identifying the type, magnitude, severity, length and impact of oil exposure to oceanic, coastal and estuarine marine mammals. The research is ongoing and will continue for some time. For continental shelf and oceanic cetaceans, the NOAA-led efforts include: aerial surveys to document the distribution, abundance, species and exposure of marine mammals and turtles relative to oil from DWH spill; and ship surveys to evaluate exposure to oil and other chemicals and to assess changes in animal behavior and distribution relative to oil exposure through visual and acoustic surveys, deployment of passive acoustic monitoring systems, collection of tissue samples, and deployment of satellite tags on sperm and Bryde’s whales.

Aerial surveys have observed Risso’s dolphins, spinner dolphins, pantropical spotted dolphins, striped dolphins, bottlenose dolphins and sperm whales swimming in oil in offshore waters (NOAA 2010a). The effects of oil exposure on marine mammals depend on a number of factors including the type and mixture of chemicals involved, the amount, frequency and duration of exposure, the route of exposure (inhaled, ingested, absorbed, or external) and biomedical risk factors of the particular animal (Geraci 1990; NOAA 2010b). In general, direct external contact with petroleum compounds or dispersants with skin may cause skin irritation, chemical burns and infections. Inhalation of volatile petroleum compounds or dispersants may irritate or injure the respiratory tract, which could lead to pneumonia or inflammation. Ingestion of petroleum compounds may cause injury to the gastrointestinal tract, which could affect an animal’s ability to digest or absorb food. Absorption of petroleum compounds or dispersants may damage kidney, liver and brain function in addition to causing immune suppression and anemia. Long term chronic effects such as lowered reproductive success and decreased survival may occur (Geraci 1990; NOAA 2010b).

STATUS OF STOCK

Dwarf sperm whales are not listed as threatened or endangered under the Endangered Species Act, and the northern Gulf of Mexico stock is not considered strategic under the MMPA because PBR is likely a severe underestimate due to the long dive times of this species and because the mean modeled annual human-caused mortality and serious injury due to the DWH oil spill is based on all dwarf and pygmy sperm whales combined and cannot be apportioned to individual species. The status of dwarf sperm whales in the northern Gulf of Mexico, relative to OSP, is unknown. The species is not listed as threatened or endangered under the Endangered Species Act. No fishery-related mortality or serious injury has been observed in recent years; therefore, total fishery-related mortality and serious injury can be considered insignificant and approaching the zero mortality and serious injury rate. The status of dwarf sperm whales in the northern Gulf of Mexico, relative to OSP, is unknown. There are insufficient data to determine the population trends for this stock. There are insufficient data to determine the population trends for this species. Total human-caused mortality and serious injury for this stock is not known. There is insufficient information available to determine whether the total fishery-related mortality and serious injury for this stock is insignificant and approaching zero mortality and serious injury rate. Despite an unknown PBR for this species, this is not a strategic stock because it is assumed that average annual human-related mortality and serious injury does not exceed combined PBR for dwarf and pygmy sperm whales. However, the continuing inability to distinguish between species of Kogia raises concerns about the possibility of mortalities of one stock or the other exceeding PBR.

REFERENCES


PYGMY SPERM WHALE (Kogia breviceps): Northern Gulf of Mexico Stock

STOCK DEFINITION AND GEOGRAPHIC RANGE

The pygmy sperm whale is appears to be distributed worldwide in temperate to tropical waters (Caldwell and Caldwell 1989; Bloodworth and Odell 2008). Sightings of these animals in the northern Gulf of Mexico (i.e., U.S. Gulf of Mexico) occur primarily in oceanic waters (Figure 1; Mullin et al. 1991; Mullin and Fulling 2004; Maze-Foley and Mullin 2006). Pygmy sperm whales and dwarf sperm whales (Kogia sima) are often difficult to differentiate at sea (Caldwell and Caldwell 1989; Bloodworth and Odell 2008; McAlpine 2009) unless sighting conditions are ideal, and sightings of either species are often categorized as Kogia spp. In addition, the acoustic signals of dwarf and pygmy sperm whales also cannot be distinguished from each other at this time (Merkens et al. 2018) adding to the difficulties of identification at sea.

In the northern Gulf of Mexico, Kogia spp. are sighted in waters >200 m, over the continental slope and deep basin. They have been seen in all seasons during aerial and vessel surveys of the northern Gulf of Mexico (Hansen et al. 1996; Mullin and Hoggard 2000; Maze-Foley and Mullin 2006; Garrison and Aichinger Dias 2020; Figure 1). All the cetacean species found in the oceanic northern Gulf of Mexico almost certainly occur in similar habitat beyond U.S. boundaries in the southern Gulf. There are fewer cetacean sighting and stranding records in the southern Gulf due to more limited effort. Nevertheless, there are records for most oceanic species in the southern Gulf (e.g., Jefferson and Schiro 1997; Ortega Ortiz 2002; Ortega-Argueta et al. 2005; Jefferson et al. 2008; Vázquez Castán et al. 2009; Whitt et al. 2011). This is therefore likely a transboundary stock with Cuba and/or Mexico. Because U.S. waters only comprise about 40% of the entire Gulf of Mexico and 35% of the oceanic (i.e., >200 m) Gulf of Mexico (Mullin and Fulling 2004), abundance and stock boundaries of oceanic species are poorly known. Sightings of this category were documented in all seasons during GulfCet aerial surveys of the northern Gulf of Mexico from 1992 to 1998 (Hansen et al. 1996; Mullin and Hoggard 2000). The difficulty in sighting pygmy and dwarf sperm whales may be exacerbated by their avoidance reaction towards ships, and change in behavior towards approaching survey aircraft (Würsig et al. 1998). In a study using hematological and stable isotope data, Barros et al. (1998) speculated that dwarf sperm whales may have a more pelagic distribution than pygmy sperm whales, and/or dive deeper during feeding bouts. Diagnostic morphological characters have also been useful in distinguishing the 2 Kogia species (Barros and Duffield...
2003), thus enabling researchers to use stranding data in distributional and ecological studies. Specifically, the distance from the snout to the center of the blowhole in proportion to the animal’s total length, as well as the height of the dorsal fin, in proportion to the animal’s total length, can be used to differentiate between the 2 Kogia species when such measurements are obtainable (Barros and Duffield 2003).

Although there are only a few records from Gulf of Mexico waters beyond U.S. boundaries (e.g., Ortega Ortiz 2002), pygmy sperm whales almost certainly occur throughout the oceanic Gulf of Mexico (Jefferson et al. 2008), which is also composed of waters belonging to Mexico and Cuba where there is currently little information on cetacean species abundance and distribution. U.S. waters only comprise about 40% of the entire Gulf of Mexico, and 65% of oceanic waters are south of the U.S. Exclusive Economic Zone (EEZ).

Pygmy sperm whales in the northern Gulf of Mexico are managed separately from those in the western North Atlantic. Although there have been no directed studies of the degree of demographic independence between the two areas, this management structure is consistent with the fact that the Gulf of Mexico and western North Atlantic belong to distinct marine ecoregions (Spalding et al. 2007; Moore and Merrick 2011). There are insufficient data to determine whether the northern Gulf of Mexico stock comprises multiple demographically independent populations. Additional morphological, acoustic, genetic, and/or behavioral data are needed to further delineate population structure within the Gulf of Mexico and across the broader geographic area. The Gulf of Mexico population is provisionally being considered a separate stock for management purposes, although there is currently no information to differentiate this stock from the Atlantic Ocean stock(s). Additional morphological, genetic and/or behavioral data are needed to provide further information on stock delineation (Figure 1). Distribution of pygmy and dwarf sperm whale sightings from SEFSC vessel surveys during spring 1996-2001, summer 2003 and spring 2004, and summer 2009. All the on-effort sightings are shown, though not all were used to estimate abundance. Solid lines indicate the 100m and 1,000m isobaths and the offshore extent of the U.S. EEZ.

Population Size

The best abundance estimate (Nbest) available for northern Gulf of Mexico pygmy and dwarf sperm whales combined is 336 (CV=0.35; Table 1) (Hansen et al. 1995), and for 1996 to 2001, 742 (CV=0.29) (Mullin and Fulling 2004; Table 1). A separate estimate of abundance for pygmy sperm whales could not be estimated due to uncertainty of species identification at sea.

Earlier abundance estimates

Please see Appendix IV for a summary of abundance estimates, including earlier estimates and survey descriptions. All estimates of abundance were derived through the application of distance sampling analysis (Buckland et al. 2001) and the computer program DISTANCE (Thomas et al. 1998) to line transect survey data collected from ships in the oceanic northern Gulf of Mexico (i.e., 200-m isobath to seaward extent of the U.S. EEZ) and are summarized in Appendix IV.

From 1991 through 1994, and from 1996 through 2001 (excluding 1998), annual surveys were conducted during spring along a fixed plankton-sampling trackline. Due to limited survey effort in any given year, the survey effort-weighted estimated average abundance of dwarf and pygmy sperm whales for all surveys combined was estimated. For 1991 to 1994, the estimate was 517 (CV=0.28) (Hansen et al. 1995), and for 1996 to 2001, 742 (CV=0.29) (Mullin and Fulling 2004; Table 1). A separate estimate of abundance for pygmy sperm whales could not be estimated due to uncertainty of species identification at sea.

During summer 2003 and spring 2004, surveys dedicated to estimating cetacean abundance were conducted along a grid of uniformly spaced transect lines from a random start. The estimate of abundance for dwarf and pygmy sperm whales in oceanic waters, pooled from 2003 to 2004, was 453 (CV=0.35) (Mullin 2007; Table 1).

Recent surveys and abundance estimates

An abundance estimate for dwarf and pygmy sperm whales combined was generated from vessel surveys conducted in the northern Gulf of Mexico from the continental shelf edge (~200-m isobath) to the seaward extent of the U.S. EEZ (Garrison et al. 2020). One survey was conducted from 2 July to 25 August 2017 and consisted of 7,302
km of on-effort trackline, and the second survey was conducted from 11 August to 6 October 2018 and consisted of 6,473 km of on-effort trackline within the surveyed strata. Both surveys used a double-platform data-collection procedure to allow estimation of the detection probability on the trackline using the independent observer approach assuming point independence (Laake and Borchers 2004). However, there were too few sightings and too few resightings of these species to allow estimation of detection probability on the trackline. Therefore, abundance estimates were derived using MCDS distance sampling methods that accounted for the effects of covariates (e.g., sea state, glare) on detection probability within the surveyed strip (Thomas et al. 2010) implemented in package mrds (version 2.21, Laake et al. 2020) in the R statistical programming language. The surveys were conducted in “passing mode” (e.g., Schwarz et al. 2010) while all prior surveys in the Gulf of Mexico have been conducted in "closing mode.” The inverse variance weighted mean abundance estimate for dwarf and pygmy sperm whales in oceanic waters during 2017 and 2018 was 336 (CV=0.35; Table 1; Garrison et al. 2020). This estimate was not corrected for the probability of detection on the trackline, and is likely a severe underestimate due to the long dive times of these species. During summer 2009, a line-transect survey dedicated to estimating the abundance of oceanic cetaceans was conducted in the northern Gulf of Mexico. Survey lines were stratified in relation to depth and the location of the Loop Current. The estimate of abundance for dwarf and pygmy sperm whales in oceanic waters during 2009 was 186 (CV=1.04; Table 1).

Table 1. Most recent abundance estimate (Nbest) and coefficient of variation (CV) of northern Gulf of Mexico pygmy and dwarf sperm whales in oceanic waters (200m to the offshore extent of the EEZ) based on an average from summer 2017 and summer/fall 2018 vessel surveys.

<table>
<thead>
<tr>
<th>Years</th>
<th>Area</th>
<th>Nbest</th>
<th>CV</th>
</tr>
</thead>
<tbody>
<tr>
<td>2017, 2018</td>
<td>Gulf of Mexico</td>
<td>336</td>
<td>0.35</td>
</tr>
</tbody>
</table>

Table 1. Summary of combined abundance estimates for northern Gulf of Mexico pygmy and dwarf sperm whales. Month, year and area covered during each abundance survey, and resulting abundance estimate (Nbest) and coefficient of variation (CV).

<table>
<thead>
<tr>
<th>Month/Year</th>
<th>Area</th>
<th>Nbest</th>
<th>CV</th>
</tr>
</thead>
<tbody>
<tr>
<td>Apr-Jun 1991–1994</td>
<td>Oceanic waters</td>
<td>547</td>
<td>0.28</td>
</tr>
<tr>
<td>Apr-Jun 1996–2001 (excluding 1998)</td>
<td>Oceanic waters</td>
<td>742</td>
<td>0.29</td>
</tr>
<tr>
<td>Jun-Aug 2003, Apr-Jun 2004</td>
<td>Oceanic waters</td>
<td>453</td>
<td>0.35</td>
</tr>
<tr>
<td>Jun-Aug 2009</td>
<td>Oceanic waters</td>
<td>186</td>
<td>1.04</td>
</tr>
</tbody>
</table>

Minimum Population Estimate

The minimum population estimate (Nmin) is the lower limit of the two-tailed 60% confidence interval of the log-normal distributed abundance estimate. This is equivalent to the 20th percentile of the log-normal distributed abundance estimate as specified by Wade and Angliss (1997). The best estimate of abundance for pygmy and dwarf sperm whales is 336 (CV=0.35, Table 1). It is not possible to determine the minimum population estimate for only pygmy sperm whales. The minimum population estimate for the northern Gulf of Mexico is 90 pygmy and dwarf
sperm whales is 253 (Table 2).

**Current Population Trend**

The statistical power to detect a trend in abundance for this stock is poor due to the relatively imprecise abundance estimates and long intervals between surveys. For example, the power to detect a precipitous decline in abundance (i.e., 50% decrease in 15 years) with estimates of low precision (e.g., CV>0.30) remains below 80% (alpha = 0.30) unless surveys are conducted on an annual basis (Taylor et al. 2007). In addition, because these surveys are restricted to U.S. waters, it is not possible to distinguish between changes in population size and Gulf-wide shifts in spatial distribution. Five point estimates of Kogia spp. abundance have been made based on data from surveys during 2003 (June–August), 2004 (April–June), 2009 (July–August), 2017 (July–August), and 2018 (August–October). Each of these surveys had a similar design and was conducted using the same vessel or a vessel with a similar observation platform. Prior year surveys only employed a single survey team. The estimates for 2017 and 2018 were conducted in "passing" mode which resulted in increased numbers of unidentified sightings and may have affected group size estimation for distant groups of dolphins and small whales. Apportioning unidentified species among identified taxa addresses the first issue, and there were no apparent relationships between sighting distance and estimated group size. The model for detection probability on the trackline from the summer 2017 survey was applied to the abundance estimates from the 2003, 2004, and 2009 surveys. The resulting abundance estimates were: 2003, N=441 (CV=0.42); 2004, N=38 (CV=0.71); 2009, N=124 (CV=0.60); 2017, N=293 (CV=0.59); and 2018, N=358 (CV=0.42). Pairwise comparisons of the log-transformed means were conducted between years, and significant differences were assessed at alpha=0.10. P-values were adjusted for multiple comparisons. There was a significant decrease between the 2003 and 2004 estimates (p.adjusted=0.006), and a significant increase between the 2004 estimates and both the 2017 (p.adjusted=0.063) and 2018 (p.adjusted=0.014) estimates; however there is no clear pattern in the overall trend. There are insufficient data to determine the population trends for this species due to uncertainty in species identification at sea. Four point estimates of Kogia spp. abundance have been made based on data from surveys covering 1991-2009. The estimates vary by a maximum factor of nearly four. To determine whether changes in abundance have occurred over this period, an analysis of all the survey data needs to be conducted which incorporates covariates (e.g., survey conditions, season) that could potentially affect estimates. Nevertheless, differences in temporal abundance estimates will still be difficult to interpret without a Gulf of Mexico-wide understanding of Kogia abundance. The oceanography of the Gulf of Mexico is quite dynamic, and the spatial scale of the Gulf is small relative to the ability of most cetacean species to travel. Studies based on abundance and distribution surveys restricted to U.S. waters are unable to detect temporal shifts in distribution beyond U.S. waters that might account for any changes in abundance.

**CURRENT AND MAXIMUM NET PRODUCTIVITY RATES**

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

**POTENTIAL BIOLOGICAL REMOVAL**

Potential Biological Removal (PBR) is the product of the minimum population size, one half the maximum net productivity rate and a recovery factor (MMPA Sec. 3.16 U.S.C. 1362; Wade and Angliss 1997). The minimum population size for pygmy and dwarf sperm whales is 253. The maximum productivity rate is 0.04, the default value for cetaceans. The recovery factor, which accounts for endangered, depleted, threatened stocks, or stocks of unknown status relative to optimum sustainable population (OSP), is assumed to be 0.5 because the stock is of unknown status. PBR for the northern Gulf of Mexico pygmy and dwarf sperm whales is 2.5 (Table 2). It is not possible to determine the PBR for only pygmy sperm whales.

*Table 2. Best and minimum abundance estimates for the northern Gulf of Mexico pygmy and dwarf sperm whales with Maximum Productivity Rate (Rmax), Recovery Factor (Fr) and PBR.*

<table>
<thead>
<tr>
<th>Nbest</th>
<th>CV</th>
<th>Nmin</th>
<th>Fr</th>
<th>Rmax</th>
<th>PBR</th>
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<tr>
<td>336</td>
<td>0.35</td>
<td>253</td>
<td>0.5</td>
<td>0.04</td>
<td>2.5</td>
</tr>
</tbody>
</table>

**ANNUAL HUMAN- CAUSED MORTALITY AND SERIOUS INJURY**

Total annual estimated fishery-related mortality and serious injury to this stock during 2014–2018 was presumed...
to be zero, as there were no reports of mortalities or serious injuries to pygmy or dwarf sperm whales in the Gulf of Mexico (Table 3). Mean annual mortality and serious injury during 2014–2018 for pygmy and dwarf sperm whales due to other human-caused actions (the Deepwater Horizon oil spill) was predicted to be 31. The minimum total mean annual human-caused mortality and serious injury for pygmy and dwarf sperm whales during 2014–2018 was, therefore, 31. The minimum total mean annual human-caused mortality and serious injury for pygmy sperm whales is unknown. The estimated annual average fishery-related mortality or serious injury to this stock during 2006–2010 was 0.3 pygmy sperm whales (CV=1.0; Table 2).

Table 3. Total annual estimated fishery-related mortality and serious injury for the northern Gulf of Mexico pygmy and dwarf sperm whales.

<table>
<thead>
<tr>
<th>Years</th>
<th>Source</th>
<th>Annual Avg.</th>
<th>CV</th>
</tr>
</thead>
<tbody>
<tr>
<td>2014–2018</td>
<td>U.S. fisheries using observer data</td>
<td>0</td>
<td></td>
</tr>
</tbody>
</table>

Fisheries Information

There are two commercial fisheries that interact, or that could potentially interact, with this stock in the Gulf of Mexico. These are the Category I Atlantic Highly Migratory Species (high seas) longline fishery, and the Category I Atlantic Ocean, Caribbean, Gulf of Mexico large pelagics longline fishery (Appendix III). Percent observer coverage (percentage of sets observed) for these longline fisheries for each year during 2014–2018 was 18, 19, 23, 13 and X, respectively. There is very little effort within the Gulf of Mexico by the Atlantic Highly Migratory Species (high seas) longline fishery, and no takes of pygmy or dwarf sperm whales within high seas waters of the Gulf of Mexico have been observed or reported thus far. Pelagic swordfish, tunas and billfish are the targets of the large pelagics longline fishery operating in the northern Gulf of Mexico. During 2014–2018 there were no observed mortalities or serious injuries to dwarf or pygmy sperm whales by this fishery (Garrison and Stokes 2016; 2017; 2019; 2020; in review). The commercial fishery which potentially could interact with this stock in the Gulf of Mexico is the Atlantic Ocean, Caribbean, Gulf of Mexico large pelagic longline fishery (Appendix III). Pelagic swordfish, tunas and billfish are the targets of the longline fishery operating in the northern Gulf of Mexico. There were no reports of mortality or serious injury to pygmy sperm whales by this fishery during 1998–2009 (Yeung 1999; Yeung 2001; Garrison 2003; Garrison and Richards 2004; Garrison 2005; Fairfield Walsh and Garrison 2006; Fairfield Walsh and Garrison 2007; Fairfield and Garrison 2008; Garrison et al. 2009; Garrison and Stokes 2010); however, during 2010, 1 mortality of a pygmy sperm whale (a portion of the carcass was retrieved and species identification was confirmed through genetic analyses) was observed during quarter 2 and estimated mortalities attributable to the pelagic longline fishery in the Gulf of Mexico region during quarter 2 were 1.2 (CV=1.00; Garrison and Stokes 2011). The total estimated mortality for 2010 was 1.2 animals (CV=1.0). The annual average serious injury and mortality attributable to the Gulf of Mexico pelagic longline fishery for the 5-year period from 2006 to 2010 was 0.3 animals (CV=1.0; Table 2).

Table 2. Summary of the incidental mortality and serious injury of northern Gulf of Mexico pygmy sperm whale (Kogia breviceps) by commercial fishery including the years sampled (Years), the number of vessels active within the fishery (Vessels), the type of data used (Data Type), the annual observer coverage (Observer Coverage), the observed mortalities and serious injuries recorded by on-board observers, the estimated annual mortality and serious injury, the combined annual estimates of mortality and serious injury (Estimated Combined Mortality), the estimated CV of the combined estimates (Estimated CVs) and the mean of the combined estimates (CV in parentheses).
Other Mortality

At least 1,419 pygmy sperm whale strandings were documented in the northern Gulf of Mexico during 2006–2014–2018 (Table 43; NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 21 May 2019–16 November 2011). Evidence of human interactions was detected for one of the stranded animals, and this involved the ingestion of plastic debris. For four of the strandings, no evidence of human interactions was detected, and for the remaining 14, it could not be determined if there was evidence of human interactions. An additional 310 *Kogia* spp. stranded during 2006–2014–2018–2010. No evidence of human interactions was detected for one of the *Kogia* sp. strandings; it could not be determined if there was evidence of human interactions for the remaining 29 *Kogia* sp. strandings. Stranding data probably underestimate the extent of human and fishery-related mortality and serious injury because not all of the whales that die or are seriously injured in human interactions wash ashore, or, if they do, they are not all recovered (Peltier et al. 2012; Wells et al. 2015). In particular, oceanic stocks in the Gulf of Mexico are less likely to strand than nearshore coastal stocks or shelf stocks (Williams et al. 2011). Additionally, not all carcasses will show evidence of human interaction, entanglement or other fishery-related interaction due to decomposition, scavenger damage, etc. (Byrd et al. 2014). Finally, the level of technical expertise among stranding network personnel varies widely as does the ability to recognize signs of human interaction. Stranding data probably underestimate the extent of fishery-related mortality and serious injury because not all of the marine mammals which die or are seriously injured in fishery interactions wash ashore, not all that wash ashore are discovered, reported or investigated, 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 interactions.

An Unusual Mortality Event (UME) was declared for cetaceans in the northern Gulf of Mexico beginning 1 February–March 2010 and ending 31 July 2014 (Litz et al. 2014; https://www.fisheries.noaa.gov/national/marine-life-distress/2010-2014-cetacean-unusual-mortality-event-northern-gulf-mexico); and, as of early 2012, the event is still...
ongoing. It included cetaceans that stranded prior to the Deepwater Horizon (DWH) oil spill (see “Habitat Issues” below), during the spill, and after. Exposure to the DWH oil spill was determined to be the primary underlying cause of the elevated stranding numbers in the northern Gulf of Mexico after the spill (e.g., Schwacke et al. 2014; Venn-Watson et al. 2015; Colegrove et al. 2016; DWH NRDA-T 2016; see Habitat Issues section). Four pygmy sperm whale strandings (from 2011, 2012 and 2013) were considered to be part of this UME. During 2010, 1 animal from this stock was considered to be part of the UME.

A population model was developed to estimate the injury and time to recovery for stocks affected by the DWH oil spill, taking into account long-term effects resulting from mortality, reproductive failure, reduced survival rates, and the proportion of the stock exposed to DWH oil (DWH MMIQT 2015). Overall, the model estimated that 340 dwarf and pygmy sperm whale died during 2010–2013 (four year annual average of 85) due to elevated mortality associated with oil exposure (see Appendix VI). For the 2014–2018 reporting period of this SAR, the population model estimated 154 dwarf and pygmy sperm whales died due to elevated mortality associated with oil exposure. However, this mortality estimate is not comparable to the current abundance estimate derived from visual surveys because the population model included a correction factor for detection probability derived from acoustic density estimates (DWH MMIQT 2015). The population model used to predict dwarf/pygmy sperm whale mortality due to the DWH event has a number of sources of uncertainty. Model parameters (e.g., survival rates, reproductive rates, and life-history parameters) were derived from literature sources for dwarf/pygmy sperm whales occupying waters outside of the Gulf of Mexico. In addition, proxy values for the effects of DWH oil exposure on both survival rates and reproductive success were applied based upon estimated values for common bottlenose dolphins in Barataria Bay. Finally, there was no estimation of uncertainty in model parameters or outputs.

Table 4. Dwarf and pygmy sperm whale (Kogia sima (Ks), Kogia breviceps (Kb) and Kogia sp. (Sp)) strandings along the northern Gulf of Mexico coast, 2014–2018. Strandings that were not reported to species have been reported as Kogia spp. The level of technical expertise among stranding network personnel varies, and given the potential difficulty in correctly identifying stranded Kogia whales to species, reports to specific species should be viewed with caution.

<table>
<thead>
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<td>3</td>
<td>2</td>
<td>7</td>
<td>9</td>
</tr>
</tbody>
</table>

Table 3. Pygmy sperm whale (Kogia breviceps) strandings along the northern Gulf of Mexico coast, 2006–2010.

<table>
<thead>
<tr>
<th>STATE</th>
<th>2006</th>
<th>2007</th>
<th>2008</th>
<th>2009</th>
<th>2010</th>
<th>TOTAL</th>
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</table>
HABITAT ISSUES

The Deepwater Horizon (DWH) MC252 drilling platform, located approximately 50 miles (80 km) southeast of the Mississippi River Delta in waters about 1,500 m deep, exploded on 20 April 2010. The rig sank, and for over 87 days, ~3.2 million barrels of oil and gas were discharged from the wellhead until it was capped on 15 July 2010 (DWH NRDAT 2016). During the response effort dispersants were applied extensively at the seafloor and at the sea surface (Lehr et al. 2010; OSAT 2010). In-situ burning, or controlled burning of oil at the surface, was also used extensively as a response tool (Lehr et al. 2010). The oil, dispersant and burn residue compounds present ecological concerns. The magnitude of this oil spill was unprecedented in U.S. history, causing impacts to wildlife, natural habitats and human communities along coastal areas from western Louisiana to the Florida Panhandle (NOAA 2011). It could be years before the entire scope of damage is ascertained (NOAA 2011).

Shortly after the oil spill, the Natural Resource Damage Assessment (NRDA) process was initiated under the Oil Pollution Act of 1990. A variety of NRDA research studies were conducted to determine potential impacts of the spill on marine mammals. These studies estimated that 15% (95% CI: 8–29) of dwarf/pygmy sperm whales in the Gulf were exposed to oil, that 7% (95% CI: 3–10) of females suffered from reproductive failure, and 6% (95% CI: 2–9) of dwarf/pygmy sperm whales suffered adverse health effects (DWH MMIQT 2015). A population model estimated the stocks experienced a maximum 6% reduction in population size (see Other Mortality section above).

Anthropogenic sound in the world’s oceans has been shown to affect marine mammals, with vessel traffic, seismic surveys, and active naval sonars being the main anthropogenic contributors to low- and mid-frequency noise in oceanic waters (e.g., Nowacek et al. 2015; Gomez et al. 2016; NMFS 2018). The long-term and population consequences of these impacts are less well-documented and likely vary by species and other factors. Impacts on marine mammal prey from sound are also possible (Carroll et al. 2017), but the duration and severity of any such prey effects on marine mammals are unknown. These studies have focused on identifying the type, magnitude, severity, length and impact of oil exposure to oceanic, coastal and estuarine marine mammals. The research is ongoing and likely will continue for some time. For continental shelf and oceanic cetaceans, the NOAA-led efforts include: aerial surveys to document the distribution, abundance, species and exposure of marine mammals and turtles relative to oil from DWH spill; and ship surveys to evaluate exposure to oil and other chemicals and to assess changes in animal behavior and distribution.
relative to oil exposure through visual and acoustic surveys, deployment of passive acoustic monitoring systems, collection of tissue samples, and deployment of satellite tags on sperm and Bryde’s whales.

Aerial surveys have observed Risso’s dolphins, spinner dolphins, pantropical spotted dolphins, striped dolphins, bottlenose dolphins and sperm whales swimming in oil in offshore waters (NOAA 2010a). The effects of oil exposure on marine mammals depend on a number of factors including the type and mixture of chemicals involved, the amount, frequency and duration of exposure, the route of exposure (inhaled, ingested, absorbed, or external) and biomedical risk factors of the particular animal (Geraci 1990; NOAA 2010b). In general, direct external contact with petroleum compounds or dispersants with skin may cause skin irritation, chemical burns and infections. Inhalation of volatile petroleum compounds or dispersants may irritate or injure the respiratory tract, which could lead to pneumonia or inflammation. Ingestion of petroleum compounds may cause injury to the gastrointestinal tract, which could affect an animal’s ability to digest or absorb food. Absorption of petroleum compounds or dispersants may damage kidney, liver and brain function in addition to causing immune suppression and anemia. Long term chronic effects such as lowered reproductive success and decreased survival may occur (Geraci 1990; NOAA 2010b).

STATUS OF STOCK

Pygmy sperm whales are not listed as threatened or endangered under the Endangered Species Act, and the northern Gulf of Mexico stock is not considered strategic under the MMPA because PBR is likely a severe underestimate due to the long dive times of this species and because the mean modeled annual human-caused mortality and serious injury due to the DWH oil spill is based on all dwarf and pygmy sperm whales combined and cannot be apportioned to individual species. No fishery-related mortality or serious injury has been observed in recent years; therefore, total fishery-related mortality and serious injury can be considered insignificant and approaching the zero mortality and serious injury rate. The status of pygmy sperm whales in the northern Gulf of Mexico, relative to OSP, is unknown. There are insufficient data to determine the population trends for this stock. The status of pygmy sperm whales in the northern Gulf of Mexico, relative to OSP, is unknown. The species is not listed as threatened or endangered under the Endangered Species Act. There are insufficient data to determine the population trends for this species. Total human-caused mortality and serious injury for this stock is not known. There is insufficient information available to determine whether the total fishery-related mortality and serious injury for this stock is insignificant and approaching zero mortality and serious injury rate. Despite an unknown PBR for this species, this is not a strategic stock because it is assumed that average annual human-related mortality and serious injury does not exceed combined PBR for dwarf and pygmy sperm whales. However, the continuing inability to distinguish between species of Kogia raises concerns about the possibility of mortalities of 1 stock or the other exceeding PBR.

REFERENCES


DWH MMIQT. 2015. Models and analyses for the quantification of injury to Gulf of Mexico cetaceans from the Deepwater Horizon Oil Spill. MM TR.01 Schwacke Quantification.of.Injury.to.GOM.Cetaceans. Southeast Fisheries Science Center, Protected Resources and Biodiversity Division, 75 Virginia Beach Dr., Miami, Florida 33140. PRBD Contribution #: PRBD-2020-02. Available from:


pp. Available from: https://repository.library.noaa.gov/view/noaa/17892


MELON-HEADED WHALE (*Peponocephala electra*):  
Northern Gulf of Mexico Stock

STOCK DEFINITION AND GEOGRAPHIC RANGE

The melon-headed whale is distributed worldwide in tropical to sub-tropical waters (Jefferson *et al.* 2008). Sightings in the northern Gulf of Mexico (i.e., U.S. Gulf of Mexico) have generally occurred in water depths >800 m and west of Mobile Bay, Alabama (Figure 1; Mullin *et al.* 1994; Mullin and Fulling 2004; Maze-Foley and Mullin 2006; Garrison and Aichinger Dias 2020) and have been documented in all seasons during GulfCet aerial surveys of the northern Gulf of Mexico between 1992 and 1998 (Hansen *et al.* 1996; Mullin and Hoggard 2000).

![Figure 1. Distribution of melon-headed whale on-effort sightings from SEFSC vessel surveys during spring 1996–2001, summer 2003, spring 2004, summer 2009, summer 2017, and summer/fall 2018. Isobaths are the 200-m, 1,000-m, and 2,000-m depth contours. The darker line indicates the U.S. EEZ.](image)

All the cetacean species found in the oceanic northern Gulf of Mexico almost certainly occur in similar habitat beyond U.S. boundaries in the southern Gulf. There are fewer cetacean sighting and stranding records in the southern Gulf due to more limited effort. Nevertheless, there are records for most oceanic species in the southern Gulf (e.g., Ortega Ortiz 2002; Ortega-Argueta *et al.* 2005; Jefferson *et al.* 2008; Vázquez Castán *et al.* 2009; Whitt *et al.* 2011). Although there are only a few records from Gulf of Mexico waters beyond U.S. boundaries (e.g., Ortega Ortiz 2002), melon-headed whales almost certainly occur throughout the oceanic Gulf of Mexico (Jefferson *et al.* 2008). This is therefore likely a transboundary stock with Cuba and/or Mexico. Because U.S. waters only comprise about 40% of the entire Gulf of Mexico and 35% of the oceanic (i.e., >200 m) Gulf of Mexico (Mullin and Fulling 2004), abundance and stock boundaries of oceanic species are poorly known, which is also composed of waters belonging to Mexico and Cuba where there is currently little information on cetacean species abundance and distribution. U.S. waters only comprise about 40% of the entire Gulf of Mexico, and 65% of oceanic waters are south of the U.S. Exclusive Economic Zone (EEZ).

Melon-headed whales in the Gulf of Mexico are managed separately from those in the western North Atlantic. Although there have been no directed studies of the degree of demographic independence between the two areas, this management structure is consistent with evidence for population structuring in other areas (Martien *et al.* 2017) and is further supported because the two stocks occupy distinct marine ecoregions (Spalding *et al.* 2007; Moore and...
There are insufficient data to determine whether the northern Gulf of Mexico stock comprises multiple demographically independent populations. Additional morphological, acoustic, genetic, and/or behavioral data are needed to further delineate population structure within the Gulf of Mexico and across the broader geographic area. The Gulf of Mexico population is provisionally being considered 1 stock for management purposes, although there is currently no information to differentiate this stock from the Atlantic Ocean stock(s). Additional morphological, genetic and/or behavioral data are needed to provide further information on stock delineation.

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

**POPULATION SIZE**

The best abundance estimate (Nbest) for the northern Gulf of Mexico melon-headed whale is 1,749 (CV=0.68; Table 1). This estimate is from summer 2017 and summer/fall 2018 oceanic surveys covering waters from the 200-m isobath to the seaward extent of the U.S. EEZ (Garrison et al. 2020). The best abundance estimate available for northern Gulf of Mexico melon-headed whales is 2,235 (CV=0.75; Table 1). This estimate is from a summer 2009 oceanic survey covering waters from the 200m isobath to the seaward extent of the U.S. EEZ.

**Earlier abundance estimates**

Please see Appendix IV for a summary of abundance estimates, including earlier estimates and survey descriptions. All estimates of abundance were derived through the application of distance sampling analysis (Buckland et al. 2001) and the computer program DISTANCE (Thomas et al. 1998) to line transect survey data collected from ships in the oceanic northern Gulf of Mexico (i.e., 200m isobath to seaward extent of the U.S. EEZ) and are summarized in Appendix IV. From 1991 through 1994, and from 1996 through 2001 (excluding 1998), annual surveys were conducted during spring along a fixed plankton-sampling trackline. Due to limited survey effort in any given year, the survey effort-weighted estimated average abundance of melon-headed whales for all surveys combined was estimated. For 1991 to 1994, the estimate was 3,965 (CV=0.39) (Hansen et al. 1995), and for 1996 to 2001, 3,451 (CV=0.55) (Mullin and Fulling 2004; Table 1).

During summer 2003 and spring 2004, surveys dedicated to estimating cetacean abundance were conducted along a grid of uniformly spaced transect lines from a random start. The abundance estimate for melon-headed whales, pooled from 2003 to 2004, was 2,283 (CV=0.76) (Mullin 2007; Table 1).

**Recent surveys and abundance estimates**

An abundance estimate for melon-headed whales was generated from vessel surveys conducted in the northern Gulf of Mexico from the continental shelf edge (~200-m isobath) to the seaward extent of the U.S. EEZ (Garrison et al. 2020). One survey was conducted from 2 July to 25 August 2017 and consisted of 7,302 km of on-effort trackline, and the second survey was conducted from 11 August to 6 October 2018 and consisted of 6,473 km of on-effort trackline within the surveyed strata. Both surveys used a double-platform data-collection procedure, which allowed estimation of the detection probability on the trackline using the independent observer approach assuming point independence (Laake and Borchers 2004). Abundance was calculated using mark-recapture distance sampling implemented in package mrds (version 2.21, Laake et al. 2020) in the R statistical programming language. This approach accounted for the effects of covariates (e.g., sea state, glare) on detection probability within the surveyed strip. The surveys were conducted in "passing mode" (e.g., Schwarz et al. 2010) while all prior surveys in the Gulf of Mexico have been conducted in "closing mode." The abundance estimate for this stock included sightings of unidentified small whales that were apportioned among identified species based on their relative density within the survey strata (Garrison et al. 2020). The inverse variance weighted mean abundance estimate for melon-headed whales in oceanic waters during 2017 and 2018 was 1,749 (CV=0.68; Table 1; Garrison et al. 2020). Unlike previous abundance estimates, this estimate was corrected for the probability of detection on the trackline. During summer 2009, a line transect survey dedicated to estimating the abundance of oceanic cetaceans was conducted in the northern
Gulf of Mexico. Survey lines were stratified in relation to depth and the location of the Loop Current. The abundance estimate for melon-headed whales in oceanic waters during 2009 was 2,235 (CV=0.75; Table 1).

Table 1. Summary of abundance estimates for northern Gulf of Mexico melon-headed whales. Month, year and area covered during each abundance survey, and resulting abundance estimate (Nbest) and coefficient of variation (CV).

<table>
<thead>
<tr>
<th>Month/Year</th>
<th>Area</th>
<th>Nbest</th>
<th>CV</th>
</tr>
</thead>
<tbody>
<tr>
<td>Apr-Jun 1991–1994</td>
<td>Oceanic waters</td>
<td>3,965</td>
<td>0.39</td>
</tr>
<tr>
<td>Jun-Aug 2003, Apr-Jun 2004</td>
<td>Oceanic waters</td>
<td>2,283</td>
<td>0.76</td>
</tr>
<tr>
<td>Jun-Aug 2009</td>
<td>Oceanic waters</td>
<td>2,235</td>
<td>0.75</td>
</tr>
</tbody>
</table>

Table 1. Most recent abundance estimate (Nbest) and coefficient of variation (CV) of northern Gulf of Mexico melon-headed whales in oceanic waters (200 m to the offshore extent of the EEZ) based on an average from summer 2017 and summer/fall 2018 vessel surveys.

<table>
<thead>
<tr>
<th>Years</th>
<th>Area</th>
<th>Nbest</th>
<th>CV</th>
</tr>
</thead>
<tbody>
<tr>
<td>2017, 2018</td>
<td>Gulf of Mexico</td>
<td>1,749</td>
<td>0.68</td>
</tr>
</tbody>
</table>

Minimum Population Estimate

The minimum population estimate (Nmin) is the lower limit of the two-tailed 60% confidence interval of the log-normal distributed abundance estimate. This is equivalent to the 20th percentile of the log-normal distributed abundance estimate as specified by Wade and Angliss (1997). The best estimate of abundance for melon-headed whales is 1,749±2,235 (CV=0.68±0.75). The minimum population estimate for the northern Gulf of Mexico melon-headed whale is 1,039 (Table 2).

Current Population Trend

The statistical power to detect a trend in abundance for this stock is poor due to the relatively imprecise abundance estimates and long intervals between surveys. For example, the power to detect a precipitous decline in abundance (i.e., 50% decrease in 15 years) with estimates of low precision (e.g., CV>0.30) remains below 80% (alpha=0.30) unless surveys are conducted on an annual basis (Taylor et al. 2007). In addition, because these surveys are restricted to U.S. waters, it is not possible to distinguish between changes in population size and Gulf-wide shifts in spatial distribution. Five point estimates of melon-headed whale abundance have been made based on data from surveys during: 2003 (June−August), 2004 (April–June), 2009 (July–August), 2017 (July–August), and 2018 (August–October). Each of these surveys had a similar design and was conducted using the same vessel or a vessel with a similar observation platform. Prior year surveys only employed a single survey team. The estimates for 2017 and 2018 were conducted in "passing" mode which resulted in increased numbers of unidentified sightings and may have affected group size estimation for distant groups of dolphins and small whales. Apportioning unidentified species among identified taxa addresses the first issue, and there were no apparent relationships between sighting distance and estimated group size. The model for detection probability on the trackline from the summer 2017 survey was applied to the abundance estimates from the 2003, 2004, and 2009 surveys. The resulting abundance estimates were: 2003, N=1,502 (CV=0.96); 2004, N=7,351 (CV=0.87); 2009, N=4,188 (CV=0.76); 2017, N=2,694 (CV=0.76); and 2018,
N=454 (CV=0.89). Pairwise comparisons of the log-transformed means were conducted between years, and significant differences were assessed at alpha=0.10. P-values were adjusted for multiple comparisons. There was a significant difference between the 2004 and 2018 estimates (p.adjusted=0.047) suggesting a possible decline in abundance during recent years. Four point estimates of melon-headed whale abundance have been made based on data from surveys covering 1991-2009. The estimates vary by a maximum factor of nearly two. To determine whether changes in abundance have occurred over this period, an analysis of all the survey data needs to be conducted which incorporates covariates (e.g., survey conditions, season) that could potentially affect estimates. Nevertheless, differences in temporal abundance estimates will still be difficult to interpret without a Gulf of Mexico-wide understanding of melon-headed whale abundance. The oceanography of the Gulf of Mexico is quite dynamic, and the spatial scale of the Gulf is small relative to the ability of most cetacean species to travel. Studies based on abundance and distribution surveys restricted to U.S. waters are unable to detect temporal shifts in distribution beyond U.S. waters that might account for any changes in abundance.

**CURRENT AND MAXIMUM NET PRODUCTIVITY RATES**

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

**POTENTIAL BIOLOGICAL REMOVAL**

Potential Biological Removal (PBR) is the product of the minimum population size, one half the maximum net productivity rate and a recovery factor (MMPA Sec. 3.16 U.S.C. 1362; Wade and Angliss 1997). The minimum population size is 1,039\[1.274\]. The maximum productivity rate is 0.04, the default value for cetaceans. The recovery factor, which accounts for endangered, depleted, threatened stocks, or stocks of unknown status relative to optimum sustainable population (OSP), is assumed to be 0.5 because the stock is of unknown status. PBR for the northern Gulf of Mexico melon-headed whale is 10 (Table 2).

<table>
<thead>
<tr>
<th>N_{best}</th>
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<th>N_{min}</th>
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<th>R_{max}</th>
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</table>

**ANNUAL HUMAN-CAUSED MORTALITY AND SERIOUS INJURY**

Total annual estimated fishery-related mortality and serious injury to this stock during 2014–2018 was presumed to be zero, as there were no reports of mortalities or serious injuries to melon-headed whales in the Gulf of Mexico (Table 3). Mean annual mortality and serious injury during 2014–2018 due to other human-caused actions (the Deepwater Horizon oil spill) was predicted to be 9.5. The minimum total mean annual human-caused mortality and serious injury for this stock during 2014–2018 was, therefore, 9.5. There has been no reported fishing-related mortality of a melon-headed whale during 1998-2010 (Yeung 1999; Yeung 2001; Garrison 2003; Garrison and Richards 2004; Garrison 2005; Fairfield Walsh and Garrison 2006; Fairfield and Garrison 2007; Fairfield and Garrison 2008; Garrison et al. 2009; Garrison and Stokes 2010; 2011).

<table>
<thead>
<tr>
<th>Years</th>
<th>Source</th>
<th>Annual Avg.</th>
<th>CV</th>
</tr>
</thead>
<tbody>
<tr>
<td>2014–2018</td>
<td>U.S. fisheries using observer data</td>
<td>0</td>
<td>-</td>
</tr>
</tbody>
</table>

**Fisheries Information**

There are two commercial fisheries that interact, or that could potentially interact, with this stock in the Gulf of Mexico. These are the Category I Atlantic Highly Migratory Species longline fishery and the Category I Atlantic Ocean, Caribbean, Gulf of Mexico large pelagics longline fishery (Appendix III). Percent observer coverage (percentage of sets observed) for these longline fisheries for each year during 2014–2018 was 18, 19, 23, 13 and 20, respectively. There is very little effort within the Gulf of Mexico by the Atlantic Highly Migratory Species (high seas...
longline fishery, and no takes of melon-headed whales within high seas waters of the Gulf of Mexico have been observed or reported thus far. Pelagic swordfish, tunas and billfish are the targets of the large pelagics longline fishery operating in the northern Gulf of Mexico. During 2014–2018 there were no observed mortalities or serious injuries to melon-headed whales by this fishery (Garrison and Stokes 2016; 2017; 2019; 2020; in review). There has historically been some take of this species in small cetacean fisheries in the Caribbean (Caldwell et al. 1976). The commercial fishery which potentially could interact with this stock in the Gulf of Mexico is the Atlantic Ocean, Caribbean, Gulf of Mexico large pelagic longline fishery (Appendix III). Pelagic swordfish, tunas and billfish are the targets of the longline fishery operating in the northern Gulf of Mexico. There were no reports of mortality or serious injury to melon-headed whales by this fishery during 1998–2010 (Yeung 1999; Yeung 2001; Garrison 2003; Garrison and Richards 2004; Garrison 2005; Fairfield Walsh and Garrison 2006; Fairfield-Walsh and Garrison 2007; Fairfield and Garrison 2008; Garrison et al. 2009; Garrison and Stokes 2010; 2011).

Other Mortality

There were 4112 reported strandings of melon-headed whales in the Gulf of Mexico during 2006–2018 (Table 42; NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 16 November 2014–21 May 2019). Evidence of human interaction was detected for one stranding (the interaction being the animal was pushed back into the water by the public). No evidence of human interaction was detected for three strandings, and for the remaining eight strandings, it could not be determined whether there was evidence of human interaction. Stranding data probably underestimate the extent of human and fishery-related mortality and serious injury because not all of the whales that die or are seriously injured in human interactions wash ashore, or, if they do, they are not all recovered (Peltier et al. 2012; Wells et al. 2015). In particular, oceanic stocks in the Gulf of Mexico are less likely to strand than nearshore coastal stocks or shelf stocks (Williams et al. 2011). Additionally, not all carcasses will show evidence of human interaction, entanglement or other fishery-related interaction due to decomposition, scavenger damage, etc. (Byrd et al. 2014). Finally, the level of technical expertise among stranding network personnel varies widely as does the ability to recognize signs of human interaction. No evidence of human interactions was detected for 6 of these stranded animals, and for the remaining 5, it could not be determined if there was evidence of human interactions. Stranding data probably underestimate the extent of fishery-related mortality and serious injury because not all of the marine mammals which die or are seriously injured in fishery interactions wash ashore, not all that wash ashore are discovered, reported or investigated, 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 interactions.

An Unusual Mortality Event (UME) was declared for cetaceans in the northern Gulf of Mexico beginning 1 February 2010, and, as of early 2012, the event is still ongoing and ending 31 July 2014 (Litz et al. 2014; https://www.fisheries.noaa.gov/national/marine-life-distress/2010-2014-cetacean-unusual-mortality-event-northern-gulf-mexico). It included cetaceans that stranded prior to the Deepwater Horizon (DWH) oil spill (see “Habitat Issues” below), during the spill, and after. Exposure to the DWH oil spill was determined to be the primary underlying cause of the elevated stranding numbers in the northern Gulf of Mexico after the spill (e.g., Schwacke et al. 2014; Venn-Watson et al. 2015; Colegrove et al. 2016; DWH NRDAT 2016; see Habitat Issues section). Ten melon-headed whale strandings were considered to be part of this UME, one of which occurred during 2014. During 2010, 2 animals from this stock were considered to be part of the UME.

A population model was developed to estimate the injury and time to recovery for stocks affected by the DWH oil spill, taking into account long-term effects resulting from mortality, reproductive failure, reduced survival rates, and the proportion of the stock exposed to DWH oil (DWH MM IQT 2015). Overall, the model estimated that this stock experienced a 7% maximum reduction in population size due to the oil spill (DWH MM IQT 2015). The mortality projected for the years 2010–2013 due to the spill has not been reported previously. Based on the population model, it was projected that 93 melon-headed whales died during 2010–2013 (four year annual average of 23) due to elevated mortality associated with oil exposure (see Appendix VI). For the 2014–2018 reporting period of this SAR, the population model estimated 47 melon-headed whales died due to elevated mortality associated with oil exposure. The population model used to predict melon-headed whale mortality due to the DWH event has a number of sources of uncertainty. Model parameters (e.g., survival rates, reproductive rates, and life-history parameters) were derived from literature sources for melon-headed whales occupying waters outside of the Gulf of Mexico. In addition, proxy values for the effects of DWH oil exposure on both survival rates and reproductive success were applied based upon estimated values for common bottlenose dolphins in Barataria Bay. Finally, there was no estimation of uncertainty in model parameters or outputs.
Table 2. Melon-headed whale (*Peponocephala electra*) strandings along the northern Gulf of Mexico coast, 2006–2010.

<table>
<thead>
<tr>
<th>STATE</th>
<th>2006</th>
<th>2007</th>
<th>2008</th>
<th>2009</th>
<th>2010</th>
<th>TOTAL</th>
</tr>
</thead>
<tbody>
<tr>
<td>Alabama</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>1*</td>
<td>1</td>
</tr>
<tr>
<td>Florida</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
</tr>
<tr>
<td>Louisiana</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>1*</td>
<td>1</td>
</tr>
<tr>
<td>Mississippi</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
</tr>
<tr>
<td>Texas</td>
<td>1</td>
<td>2</td>
<td>1</td>
<td>0</td>
<td>5</td>
<td>12</td>
</tr>
<tr>
<td>TOTAL</td>
<td>1</td>
<td>2</td>
<td>1</td>
<td>0</td>
<td>7</td>
<td>11</td>
</tr>
</tbody>
</table>

* This stranding is included in the Northern Gulf of Mexico UME.


<table>
<thead>
<tr>
<th>Area</th>
<th>2014</th>
<th>2015</th>
<th>2016</th>
<th>2017</th>
<th>2018</th>
<th>Total</th>
</tr>
</thead>
<tbody>
<tr>
<td>Alabama</td>
<td>0</td>
<td>1</td>
<td>0</td>
<td>2</td>
<td>0</td>
<td>3</td>
</tr>
<tr>
<td>Florida</td>
<td>1*</td>
<td>0</td>
<td>0</td>
<td>2</td>
<td>3</td>
<td>6</td>
</tr>
<tr>
<td>Louisiana</td>
<td>0</td>
<td>0</td>
<td>1</td>
<td>0</td>
<td>0</td>
<td>1</td>
</tr>
<tr>
<td>Mississippi</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
</tr>
<tr>
<td>Texas</td>
<td>1</td>
<td>0</td>
<td>1</td>
<td>0</td>
<td>0</td>
<td>2</td>
</tr>
<tr>
<td>TOTAL</td>
<td>2</td>
<td>1</td>
<td>2</td>
<td>4</td>
<td>3</td>
<td>12</td>
</tr>
</tbody>
</table>

* This stranding was part of the Northern Gulf of Mexico UME.

**HABITAT ISSUES**

The Deepwater Horizon (DWH) MC252 drilling platform, located approximately 50 miles (80 km) southeast of the Mississippi River Delta in waters about 1,500 m deep, exploded on 20 April 2010. The rig sank, and over 87 days (~3.2 millions of barrels of oil and gas were discharged from the wellhead until it was capped on 15 July 2010 [DWH NRDAT 2016]. During the response effort dispersants were applied extensively at the seafloor and at the sea surface (Lehr et al. 2010; OSAT 2010). In-situ burning, or controlled burning of oil at the surface, was also used extensively as a response tool (Lehr et al. 2010). The oil, dispersant and burn residue compounds present ecological concerns.
The magnitude of this oil spill was unprecedented in U.S. history, causing impacts to wildlife, natural habitats and human communities along coastal areas from western Louisiana to the Florida Panhandle (NOAA 2011). It could be years before the entire scope of damage is ascertained (NOAA 2011).

Shortly after the oil spill, the Natural Resource Damage Assessment (NRDA) process was initiated under the Oil Pollution Act of 1990. A variety of NRDA research studies were conducted to determine potential impacts of the spill on marine mammals. These studies estimated that 15% (95% CI: 6–36) of melon-headed whales in the Gulf were exposed to oil, that 7% (95% CI: 3–10) of females suffered from reproductive failure, and 6% (95% CI: 2–9) of melon-headed whales suffered adverse health effects (DWH MMIQT 2015). A population model estimated the stock experienced a maximum 7% reduction in population size (see Other Mortality section above).

Anthropogenic sound in the world’s oceans has been shown to affect marine mammals, with vessel traffic, seismic surveys, and active naval sonars being the main anthropogenic contributors to low- and mid-frequency noise in oceanic waters (e.g., Nowacek et al. 2015; Gomez et al. 2016; NMFS 2018). The long-term and population consequences of these impacts are less well-documented and likely vary by species and other factors. Impacts on marine mammal prey from sound are also possible (Carroll et al. 2017), but the duration and severity of any such prey effects on marine mammals are unknown. These studies have focused on identifying the type, magnitude, severity, length and impact of oil exposure to oceanic, coastal and estuarine marine mammals. The research is ongoing and likely will continue for some time. For continental shelf and oceanic cetaceans, the NOAA led efforts include: aerial surveys to document the distribution, abundance, species and exposure of marine mammals and turtles relative to oil from DWH spill; and ship surveys to evaluate exposure to oil and other chemicals and to assess changes in animal behavior and distribution relative to oil exposure through visual and acoustic surveys, deployment of passive acoustic monitoring systems, collection of tissue samples, and deployment of satellite tags on sperm and Bryde’s whales.

Aerial surveys have observed Risso’s dolphins, spinner dolphins, pantropical spotted dolphins, striped dolphins, bottlenose dolphins and sperm whales swimming in oil in offshore waters (NOAA 2010a). The effects of oil exposure on marine mammals depend on a number of factors including the type and mixture of chemicals involved, the amount, frequency and duration of exposure, the route of exposure (inhaled, ingested, absorbed, or external) and biomedical risk factors of the particular animal (Geraci 1990; NOAA 2010b). In general, direct external contact with petroleum compounds or dispersants with skin may cause skin irritation, chemical burns and infections. Inhalation of volatile petroleum compounds or dispersants may irritate or injure the respiratory tract, which could lead to pneumonia or inflammation. Ingestion of petroleum compounds may cause injury to the gastrointestinal tract, which could affect an animal’s ability to digest or absorb food. Absorption of petroleum compounds or dispersants may damage kidney, liver and brain function in addition to causing immune suppression and anemia. Long term chronic effects such as lowered reproductive success and decreased survival may occur (Geraci 1990; NOAA 2010b).

**STATUS OF STOCK**

Melon-headed whales are not listed as threatened or endangered under the Endangered Species Act, and the northern Gulf of Mexico stock is not considered strategic under the Marine Mammal Protection Act. No fishery-related mortality or serious injury has been observed in recent years; therefore, total fishery-related mortality and serious injury can be considered insignificant and approaching the zero mortality and serious injury rate. The status of melon-headed whales in the northern Gulf of Mexico, relative to OSP, is unknown. There was a statistically significant decrease in population size in recent years for this stock in the northern Gulf of Mexico. The species is not listed as threatened or endangered under the Endangered Species Act. There are insufficient data to determine the population trends for this species. Total human caused mortality and serious injury for this stock is not known but none has been documented. There is insufficient information available to determine whether the total fishery-related mortality and serious injury for this stock is insignificant and approaching zero mortality and serious injury rate. This is not a strategic stock because it is assumed that the average annual human-related mortality and serious injury does not exceed PBR.

**REFERENCES**


DWH MMIQT. 2015. Models and analyses for the quantification of injury to Gulf of Mexico cetaceans from the Deepwater Horizon Oil Spill, MM TR.01 Schwacke Quantification of Injury to GOM Cetaceans, Southeast Fisheries Science Center, Protected Resources and Biodiversity Division, 75 Virginia Beach Dr., Miami, Florida 33140. PRBD Contribution #: PRBD-2020-02. Available from:


RISSO'S DOLPHIN (*Grampus griseus*): Northern Gulf of Mexico Stock

STOCK DEFINITION AND GEOGRAPHIC RANGE

Risso’s dolphins are distributed worldwide in tropical to warm temperate waters (Leatherwood and Reeves 1983; Jefferson *et al.* 2014). Risso’s dolphins in the northern Gulf of Mexico (i.e., U.S. Gulf of Mexico) occur throughout oceanic waters but are concentrated in continental slope waters (Figure 1; Baumgartner 1997; Maze-Foley and Mullin 2006; Garrison and Aichinger Dias 2020). This species has been observed Risso's dolphins were seen in all seasons in the northern Gulf of Mexico during GulfCet aerial surveys of the northern Gulf of Mexico between 1992 and 1998.

![Figure 1. Distribution of Risso's dolphin on-effort sightings from SEFSC vessel surveys during spring 1996-2001, summer 2003, spring 2004, summer 2009, summer 2017, and summer/fall 2018. Isobaths are the 200-m, 1,000-m, and 2,000-m depth contours. The darker line indicates the U.S. EEZ.](image)

All the cetacean species found in the oceanic northern Gulf of Mexico almost certainly occur in similar habitat beyond U.S. boundaries in the southern Gulf. There are fewer cetacean sighting and stranding records in the southern Gulf due to more limited effort. Nevertheless there are records for most oceanic species in the southern Gulf (e.g., Jefferson and Schiro 1997; Ortega Ortiz 2002; Ortega-Argueta *et al.* 2005; Jefferson *et al.* 2008; Vázquez Castán *et al.* 2009; Whitt *et al.* 2011). This is therefore likely a transboundary stock with Cuba and/or Mexico. Because U.S. waters only comprise about 40% of the entire Gulf of Mexico and 35% of the oceanic (i.e., >200 m) Gulf of Mexico (Mullin and Fulling 2004), abundance and stock boundaries of oceanic species are poorly known. Although there are only a few records from Gulf of Mexico waters beyond U.S. boundaries (e.g., Jefferson and Schiro 1997, Ortega Ortiz 2002), Risso’s dolphins almost certainly occur throughout the oceanic Gulf of Mexico (Jefferson *et al.* 2008), including waters belonging to Mexico and Cuba where there is currently little information on cetacean species abundance and distribution. U.S. waters comprise about 40% of the entire Gulf of Mexico and 35% of the oceanic (i.e., >200 m) Gulf of Mexico.
Figure 1. Distribution of Risso’s dolphin sightings from SEFSC vessel surveys during summer 2003 and spring 2004, and during summer 2009. All the on-effort sightings are shown, though not all were used to estimate abundance. Solid lines indicate the 20-m and 200-m isobaths and the offshore extent of the U.S. EEZ.

The Gulf of Mexico population is being considered a separate stock for management purposes, although there is currently little information to differentiate this stock from the Atlantic Ocean stock. In 2006, a Risso’s dolphin that stranded on the Florida Gulf Coast was rehabilitated, tagged with a satellite-linked transmitter and released into the Gulf southwest of Tampa Bay. Over a 23-day period the Risso’s dolphin moved from the Gulf release site into the Atlantic Ocean and north to just off of Delaware (Wells et al. 2009). During September 2007 – January 2008, tracking of an adult female Risso’s dolphin that had been rehabilitated and released by Mote Marine Laboratory after stranding on the southwest coast of Florida documented movements throughout the northern Gulf of Mexico. The dolphin, released with its young calf, traveled as far as Bahia de Campeche, Mexico, and waters off Texas and Louisiana before returning to the shelf edge southwest of its stranding site off Florida (Wells et al. 2008). As Wells et al. (2009) note, it is difficult to determine the effects of stranding and rehabilitation on post-release behavior, so it is unknown whether these movements were representative of Risso’s dolphin ranging patterns in either the Gulf of Mexico or Atlantic Ocean. Additional morphological, genetic and/or behavioral data are needed to provide further information on stock delineation.

Risso’s dolphins in the northern Gulf of Mexico are managed separately from those in the western North Atlantic. Although there have been no directed studies of the degree of demographic independence between the two areas, this management structure is consistent with acoustic evidence. The frequency values of spectral peaks in Risso’s dolphin echolocation clicks differ between the western North Atlantic and Gulf of Mexico stocks (Soldevilla et al. 2017). In addition, these two stocks occupy distinct marine ecoregions (Spalding et al. 2007; Moore and Merrick 2011) and biogeographic endemism has been identified for Risso’s dolphins in the North Pacific (Chen et al. 2018). However, a stranded, rehabilitated Risso’s dolphin that was released and tagged with a satellite-linked transmitter moved from the Gulf release site near Tampa, Florida, into the Atlantic Ocean and north to just off of Delaware over a 23 day period (Wells et al. 2009), suggesting the possibility of connectivity between the two basins. As Wells et al. (2009) note, it is difficult to determine the effects of stranding and rehabilitation on post-release behavior, so it is unknown whether these movements were representative of Risso’s dolphin ranging patterns in either the Gulf of Mexico or Atlantic Ocean. There are insufficient data to determine whether the northern Gulf of Mexico stock comprises multiple demographically independent populations. Additional morphological, acoustic, genetic, and/or behavioral data are needed to further delineate population structure within the Gulf of Mexico and across the broader geographic area.

**POPULATION SIZE**

The best abundance estimate (Nbest) for the northern Gulf of Mexico Risso’s dolphins is 1,974 (CV=0.46; Table 1). This estimate is from summer 2017 and summer/fall 2018 oceanic surveys covering waters from the 200-m isobath to the seaward extent of the U.S. EEZ (Garrison et al. 2020). The best abundance estimate available for northern Gulf of Mexico Risso’s dolphins is 2,442 (CV=0.57; Table 1). This estimate is from a summer 2009 oceanic survey covering waters from the 200-m isobath to the seaward extent of the U.S. EEZ.

**Earlier abundance estimates**

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

**Recent surveys and abundance estimates**

An abundance estimate for Risso’s dolphins was generated from vessel surveys conducted in the northern Gulf of Mexico from the continental shelf edge (~200-m isobath) to the seaward extent of the U.S. EEZ (Table 1; Garrison et al. 2020). One survey was conducted from 2 July to 25 August 2017 and consisted of 7,302 km of on-effort trackline, and the second survey was conducted from 11 August to 6 October 2018 and consisted of 6,473 km of on-effort trackline within the surveyed strata. Both surveys used a double-platform data-collection procedure, which allowed
estimation of the detection probability on the trackline using the independent observer approach assuming point independence (Laake and Borchers 2004). Abundance was calculated using mark-recapture distance sampling implemented in package mrds (version 2.21, Laake et al. 2020) in the R statistical programming language. This approach accounted for the effects of covariates (e.g., sea state, glare) on detection probability within the surveyed strip. The surveys were conducted in "passing mode" (e.g., Schwarz et al. 2010) while all prior surveys in the Gulf of Mexico have been conducted in "closing mode." The abundance estimate for this stock included sightings of unidentified small whales that were apportioned among identified species based on their relative density within the survey strata (Garrison et al. 2020). The inverse variance weighted mean abundance estimate for Risso’s dolphins in oceanic waters during 2017 and 2018 was 1,974 (CV=0.46; Table 1; Garrison et al. 2020). Unlike previous abundance estimates, this estimate was corrected for the probability of detection on the trackline. During summer 2009, a vessel-based line-transect survey dedicated to estimating the abundance of oceanic cetaceans was conducted in the northern Gulf of Mexico. Survey lines were stratified in relation to depth and the location of the Loop Current. The abundance estimate for Risso’s dolphins in oceanic waters during 2009 was 2,442 (CV=0.57; Table 1).

Table 1. Most recent abundance estimate (N_best) and coefficient of variation (CV) of northern Gulf of Mexico Risso’s dolphins in oceanic waters (200 m to the offshore extent of the EEZ) based on an average from summer 2017 and summer/fall 2018 vessel surveys.

<table>
<thead>
<tr>
<th>Years</th>
<th>Area</th>
<th>N_best</th>
<th>CV</th>
</tr>
</thead>
<tbody>
<tr>
<td>2017, 2018</td>
<td>Gulf of Mexico</td>
<td>1,974</td>
<td>0.46</td>
</tr>
</tbody>
</table>

Table 1. Summary of abundance estimates for northern Gulf of Mexico Risso’s dolphins. Month, year and area covered during each abundance survey, and resulting abundance estimate (N_best) and coefficient of variation (CV).

<table>
<thead>
<tr>
<th>Month/Year</th>
<th>Area</th>
<th>N_best</th>
<th>CV</th>
</tr>
</thead>
<tbody>
<tr>
<td>Apr-Jun 1991-1994</td>
<td>Oceanic waters</td>
<td>2,749</td>
<td>0.27</td>
</tr>
<tr>
<td>Apr-Jun 1996-2001 (excluding 1998)</td>
<td>Oceanic waters</td>
<td>2,169</td>
<td>0.32</td>
</tr>
<tr>
<td>Jun-Aug 2003, Apr-Jun 2004</td>
<td>Oceanic waters</td>
<td>1,589</td>
<td>0.27</td>
</tr>
<tr>
<td>Jun-Aug 2009</td>
<td>Oceanic waters</td>
<td>2,442</td>
<td>0.57</td>
</tr>
</tbody>
</table>

Minimum Population Estimate

The minimum population estimate (N_min) is the lower limit of the two-tailed 60% confidence interval of the log-normal distributed abundance estimate. This is equivalent to the 20th percentile of the log-normal distributed abundance estimate as specified by Wade and Angliss (1997). The best estimate of abundance for Risso’s dolphins is 1,974,2,442 (CV=0.46,0.57). The minimum population estimate for the northern Gulf of Mexico Risso’s dolphin is 1,368,1,563 Risso’s dolphins.

Current Population Trend

The statistical power to detect a trend in abundance for this stock is poor due to the relatively imprecise abundance
estimates and long intervals between surveys. For example, the power to detect a precipitous decline in abundance (i.e., 50% decrease in 15 years) with estimates of low precision (e.g., CV>0.30) remains below 80% (alpha=0.30) unless surveys are conducted on an annual basis (Taylor et al. 2007). In addition, because these surveys are restricted to U.S. waters, it is not possible to distinguish between changes in population size and Gulf-wide shifts in spatial distribution. Five point estimates of Risso’s dolphin abundance have been made based on data from surveys during: 2003 (June–August), 2004 (April–June), 2009 (July–August), 2017 (July–August), and 2018 (August–October). Each of these surveys had a similar design and was conducted using the same vessel or a vessel with a similar observation platform. Prior year surveys only employed a single survey team. The estimates for 2017 and 2018 were conducted in "passing" mode which resulted in increased numbers of unidentified sightings and may have affected group size estimation for distant groups of dolphins and small whales. Apportioning unidentified species among identified taxa addresses the first issue, and there were no apparent relationships between sighting distance and estimated group size. The model for detection probability on the trackline from the summer 2017 survey was applied to the abundance estimates from the 2003, 2004, and 2009 surveys. The resulting abundance estimates were: 2003, N=4,471 (CV=0.47); 2004, N=4,641 (CV=0.86); 2009, N=7,788 (CV=0.67); 2017, N=2,999 (CV=0.52); and 2018, N=632 (CV=0.60). Pairwise comparisons of the log-transformed mean abundances were conducted between years, and significant differences were assessed at alpha=0.10. P-values were adjusted for multiple comparisons. There were significant differences between the 2003 and 2018 estimates (p.adjusted=0.026) and the 2009 and 2018 estimates (p.adjusted=0.011) suggesting a possible decline in abundance during recent years. A trend analysis has not been conducted for this stock. Four point estimates of Risso’s dolphin abundance have been made based on data from surveys covering 1991-2009 (Table 1). The estimates vary by a maximum factor of nearly two. To determine whether changes in abundance have occurred over this period, an analysis of all the survey data needs to be conducted which incorporates covariates (e.g., survey conditions, season) that could potentially affect estimates. It should be noted that since this is a transboundary stock and the abundance estimates are for U.S. waters only, it will be difficult to interpret any detected trends.

CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

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

POTENTIAL BIOLOGICAL REMOVAL

Potential Biological Removal (PBR) is the product of the minimum population size, one half the maximum net productivity rate and a recovery factor (MMPA Sec. 3.16 U.S.C. 1362; Wade and Angliss 1997). The minimum population size is 1,3681,563. The maximum productivity rate is 0.04, the default value for cetaceans. The recovery factor is 0.5 because the stock is of unknown status. PBR for the northern Gulf of Mexico Risso’s dolphin is 14 (Table 2).

Table 2. Best and minimum abundance estimates for northern Gulf of Mexico Risso’s dolphins with Maximum Productivity Rate (Rmax), Recovery Factor (Fr) and PBR.

<table>
<thead>
<tr>
<th>Nbest</th>
<th>CV</th>
<th>Nmin</th>
<th>Fr</th>
<th>Rmax</th>
<th>PBR</th>
</tr>
</thead>
<tbody>
<tr>
<td>1,974</td>
<td>0.46</td>
<td>1,368</td>
<td>0.5</td>
<td>0.04</td>
<td>14</td>
</tr>
</tbody>
</table>

ANNUAL HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

Total annual estimated fishery-related mortality and serious injury to this stock during 2014–2018 was presumed to be zero, as there were no reports of mortalities or serious injuries to Risso’s dolphins in the Gulf of Mexico (Table 3). Mean annual mortality and serious injury during 2014–2018 due to other human-caused actions (the Deepwater Horizon oil spill) was predicted to be 5.3. The minimum total mean annual human-caused mortality and serious injury for this stock during 2014–2018 was, therefore, 5.3. The estimated mean annual fishery-related mortality and serious injury for this stock during 2009–2013 was 7.9 Risso’s dolphins (CV=0.85; Table 2) due to interactions with the pelagic longline fishery.

Table 3. Total annual estimated fishery-related mortality and serious injury for northern Gulf of Mexico Risso’s dolphins.
Fisheries Information

There are two commercial fisheries that interact, or that could potentially interact, with this stock in the Gulf of Mexico. These are the Category I Atlantic Highly Migratory Species (high seas) longline fishery and the Category I Atlantic Ocean, Caribbean, Gulf of Mexico large pelagics longline fishery (Appendix III). Percent observer coverage (percentage of sets observed) for these longline fisheries for each year during 2014–2018 was 18, 19, 23, 13 and 20, respectively. There is very little effort within the Gulf of Mexico by the high seas longline fishery, and no takes of Risso’s dolphins within high seas waters of the Gulf of Mexico have been observed or reported thus far. Pelagic swordfish, tunas and billfish are the targets of the large pelagics longline fishery operating in the northern Gulf of Mexico. During 2014–2018 there were no observed mortalities or serious injuries to Risso’s dolphins by this fishery (Garrison and Stokes 2016; 2017; 2019; 2020; in review).

During the second quarters (15 April – 15 June) of 2009–2013, observer coverage in the Gulf of Mexico pelagic longline fishery was greatly enhanced (approaching 55%) to collect more robust information on the interactions between pelagic longline vessels and spawning bluefin tuna. Therefore, the high annual observer coverage rates during 2009–2013 (Table 2) primarily reflects high coverage rates during the second quarter of each year. During the second quarter, this elevated coverage results in an increased probability that relatively rare interactions will be detected. Species within the oceanic Gulf of Mexico are presumed to be resident year-round; however, it is unknown if the bycatch rate observed during the second quarter is representative of that which occurs throughout the year.

For the 5-year period 2009–2013, the estimated annual combined serious injury and mortality attributable to the pelagic longline fishery in the northern Gulf of Mexico was 7.9 (CV=0.85) Risso’s dolphins. During 2009–2013, 3 serious injuries of Risso’s dolphins were observed during interactions with the pelagic longline fishery. These interactions occurred during the second quarter of 2011, during the fourth quarter of 2012 and during the third quarter of 2013 (Table 2; Garrison and Stokes 2010; 2012a,b; 2013; 2014). In addition, in the second quarter of 2011, 1 Risso’s dolphin was observed entangled and released alive without serious injury in the northern Gulf of Mexico (Garrison and Stokes 2012b).

Prior to 2009, 1 mortality and 2 serious injuries were observed in 2008, and in 2005 a Risso’s dolphin was observed entangled and released alive without serious injury in the northern Gulf of Mexico (Fairfield Walsh and Garrison 2006; Garrison et al. 2009).

Table 2. Summary of the incidental mortality and serious injury of northern Gulf of Mexico Risso’s dolphins in the pelagic longline commercial fishery including the years sampled (Years), the number of vessels active within the fishery (Vessels), the type of data used (Data Type), the annual observer coverage (Observer Coverage), the observed mortalities and serious injuries recorded by on-board observers, the estimated annual mortality and serious injury, the combined annual estimates of mortality and serious injury (Estimated Combined Mortality), the estimated CV of the combined estimates (Estimated CVs) and the mean of the combined estimates (CV in parentheses).
**Other Mortality**

There were 25 reported strandings of Risso’s dolphins in the Gulf of Mexico during 2009–2018 (Table 43; NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 21 May 2014). This includes one mass stranding of 2 animals in Florida during January 2009. No evidence of human interactions was detected for 2 of the stranded animals, and it could not be determined if there was evidence of human interactions for the remaining 4. Proportion of sets observed.

Since 1990, there have been 13 common bottlenose dolphin or cetacean die-offs or Unusual Mortality Events (UMEs) in the northern Gulf of Mexico, and 2 of these included a Risso’s dolphin. Between August 1999 and May 2000, 150 common bottlenose dolphins died coincident with *K. brevis* blooms and fish kills in the Florida Panhandle (additional strandings included three Atlantic spotted dolphins, *Stenella frontalis*, one Risso’s dolphin, two Blainville’s beaked whales, *Mesoplodon densirostris*, and four unidentified dolphins. Brevetoxin was determined to be the cause of this event (Twine et al. 2012; Litz et al. 2014). Between August 1999 and May 2000, 152 common bottlenose dolphins died coincident with *K. brevis* blooms and fish kills in the Florida Panhandle. Additional strandings included 3 Atlantic spotted dolphins, *Stenella frontalis*, 1 Risso’s dolphin, 2 Blainville’s beaked whales, *Mesoplodon densirostris*, and 4 unidentified dolphins. A UME was declared for cetaceans in the northern Gulf of Mexico beginning 1 February 2010 and ending 31 July 2014, and as of September 2014, the event is still ongoing (Litz et al. 2014; https://www.fisheries.noaa.gov/national/marine-life-distress/2010-2014-cetacean-unusual-mortality-event-northern-gulf-mexico). It included cetaceans that stranded prior to the Deepwater Horizon (DWH) oil spill (see “Habitat Issues” below), during the spill, and after. Exposure to the DWH oil spill was determined to be the

<table>
<thead>
<tr>
<th>Pelagic Longline</th>
<th>09-13</th>
<th>47, 46, 42, 47, 42</th>
<th>Obs. Data Logbook</th>
<th>0.0, 1, 1.1</th>
<th>0.0, 0.0, 1.5, 29.8, 15.2</th>
<th>0.0, 0.0, 0.0, 0.0, 29.8, 15.2</th>
<th>0.0, 0.0, 1.5, 29.8, 15.2</th>
<th>0.0, NA, NA, 1.0, 1.0, 0.0</th>
<th>7.9 (0.85)</th>
</tr>
</thead>
</table>

* Number of vessels in the fishery is based on vessels reporting effort to the pelagic longline logbook.

**Other Mortality**

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primary underlying cause of the elevated stranding numbers in the northern Gulf of Mexico after the spill (e.g., Schwacke et al. 2014; Venn-Watson et al. 2015; Colegrove et al. 2016; DWH NRDAT 2016; see Habitat Issues section). One Risso’s dolphin stranding from 2012 was considered to be part of this UME. During 2010, 2011, and 2013, no animals from this stock were considered to be part of the UME, but during 2012, 1 stranded Risso’s dolphin was included in the UME.

A population model was developed to estimate the injury and time to recovery for stocks affected by the DWH oil spill, taking into account long-term effects resulting from mortality, reproductive failure, reduced survival rates, and the proportion of the stock exposed to DWH oil (DWH MMIQT 2015). Overall, the model estimated that this stock experienced a 3% maximum reduction in population size due to the oil spill (DWH MMIQT 2015). The mortality projected for the years 2010–2013 due to the spill has not been reported previously. Based on the population model, it was projected that 52 Risso’s dolphins died during 2010–2013 (four year annual average of 13) due to elevated mortality associated with oil exposure (see Appendix VI). For the 2014–2018 reporting period of this SAR, the population model estimated 27 Risso’s dolphins died due to elevated mortality associated with oil exposure. The population model used to predict Risso’s dolphin mortality due to the DWH event has a number of sources of uncertainty. Model parameters (e.g., survival rates, reproductive rates, and life-history parameters) were derived from literature sources for Risso’s dolphins occupying waters outside of the Gulf of Mexico. In addition, proxy values for the effects of DWH oil exposure on both survival rates and reproductive success were applied based upon estimated values for common bottlenose dolphins in Barataria Bay. Finally, there was no estimation of uncertainty in model parameters or outputs.


<table>
<thead>
<tr>
<th>State</th>
<th>2014</th>
<th>2015</th>
<th>2016</th>
<th>2017</th>
<th>2018</th>
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<tbody>
<tr>
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<td>1</td>
<td>3</td>
<td>0</td>
<td>0</td>
<td>4</td>
</tr>
<tr>
<td>Louisiana</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>1</td>
<td>0</td>
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<tr>
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<td>0</td>
<td>1</td>
<td>3</td>
<td>1</td>
<td>0</td>
<td>5</td>
</tr>
</tbody>
</table>

Table 3. Risso’s dolphin (Grampus griseus) strandings along the northern Gulf of Mexico coast, 2009-2011 stranding - PRico about HI. Waiting to see. One Fl, have asked for more info. SER12-0636.e)research trawl ttted dolphin o-2013.
HABITAT ISSUES

The Deepwater Horizon (DWH) MC252 drilling platform, located approximately 50 miles/80 km southeast of the Mississippi River Delta in waters about 1,500 m deep, exploded on 20 April 2010. The rig sank, and over 87 days up to ~3.24.9 million barrels of oil were discharged from the wellhead until it was capped on 15 July 2010 (DWH NRDAT 2016McNutt et al. 2012). During the response effort dispersants were applied extensively at the seafloor and at the sea surface (Lehr et al. 2010; OSAT 2010). In situ burning, or controlled burning of oil at the surface, was also used extensively as a response tool (Lehr et al. 2010). The oil, dispersant and burn residue compounds present ecological concerns (Buist et al. 1999; NOAA 2011). The magnitude of this oil spill was unprecedented in U.S. history, causing impacts to wildlife, natural habitats and human communities along coastal areas from western Louisiana to the Florida Panhandle (NOAA 2011). It could be years before the entire scope of damage is ascertained (NOAA 2011).

Shortly after the oil spill, the Natural Resource Damage Assessment (NRDA) process was initiated under the Oil Pollution Act of 1990. A variety of NRDA research studies are being conducted to determine potential impacts of the spill on marine mammals. These studies estimated that 8% (95% CI: 5–13) of Risso’s dolphins in the Gulf were exposed to oil, that 3% (95% CI: 2–5) of females suffered from reproductive failure, and 3% (95% CI: 1–4) of Risso’s dolphins suffered adverse health effects (DWH MMIQT 2015). A population model estimated the stock experienced a maximum 3% reduction in population size (see Other Mortality section above).

Anthropogenic sound in the world’s oceans has been shown to affect marine mammals, with vessel traffic, seismic surveys, and active naval sonars being the main anthropogenic contributors to low- and mid-frequency noise in oceanic waters (e.g., Nowacek et al. 2015; Gomez et al. 2016; NMFS 2018). The long-term and population consequences of these impacts are less well-documented and likely vary by species and other factors. Impacts on marine mammal prey from sound are also possible (Carroll et al. 2017), but the duration and severity of such prey effects on marine mammals are unknown. These studies have focused on identifying the type, magnitude, severity, length and impact of oil exposure to oceanic, continental shelf, coastal and estuarine marine mammals. For continental shelf and oceanic cetaceans, the NOAA-led efforts include: aerial surveys to document the distribution, abundance, species and exposure relative to oil from the DWH spill; and ship surveys to evaluate exposure to oil and other chemicals and to assess changes in animal behavior and distribution relative to oil exposure through visual and acoustic surveys, deployment of passive acoustic monitoring systems, collection of tissue samples, and deployment of satellite-linked tags on sperm and Bryde’s whales.

Vessel and aerial surveys documented Risso’s dolphins, bottlenose dolphins, Atlantic spotted dolphins, rough-toothed dolphins, spinner dolphins, pantropical spotted dolphins, striped dolphins, sperm whales, dwarf pygmy sperm whales and a Cuvier’s beaked whale swimming in oil or potentially oil-derived substances (e.g., sheen, mousse) in offshore waters of the northern Gulf of Mexico following the DWH oil spill. The effects of oil exposure on marine mammals depend on a number of factors including the type and mixture of chemicals involved, the amount, frequency and duration of exposure, the route of exposure (inhaled, ingested, absorbed, or external) and biomedical risk factors of the particular animal (Geraci 1990). In general, direct external contact with petroleum compounds or dispersants with skin may cause skin irritation, chemical burns and infections. Inhalation of volatile petroleum compounds or dispersants may irritate or injure the respiratory tract, which could lead to pneumonia or inflammation. Ingestion of petroleum compounds may cause injury to the gastrointestinal tract, which could affect an animal’s ability to digest or absorb food. Absorption of petroleum compounds or dispersants may damage kidney, liver and brain function in addition to causing immune suppression and anemia. Long-term chronic effects such as lowered reproductive success and decreased survival may occur (Geraci 1990).

STATUS OF STOCK
Risso's dolphins are not listed as threatened or endangered under the Endangered Species Act, and the northern Gulf of Mexico stock is not considered strategic under the MMPA. No fishery-related mortality or serious injury has been observed in recent years; therefore, total fishery-related mortality and serious injury 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 mean annual fishery-related mortality and serious injury does not exceed PBR. The status of Risso’s dolphins in the northern Gulf of Mexico, relative to OSP, is unknown. There were statistically significant decreases in population size in recent years for this stock in the northern Gulf of Mexico. There are insufficient data to determine the population trends for this species.

REFERENCES CITED
DWH MMIQT. 2015. Models and analyses for the quantification of injury to Gulf of Mexico cetaceans from the Deepwater Horizon Oil Spill, MM TR.01 Schwacke Quantification.of.Injury.to.GOM.Cetaceans. Southeast Fisheries Science Center, Protected Resources and Biodiversity Division, 75 Virginia Beach Dr., Miami, Florida 33140. PRBD Contribution #: PRBD-2020-02. Available from:
Garrison, L.P. and L. Stokes. in review. Estimated bycatch of marine mammals and sea turtles in the U.S. Atlantic pelagic longline fleet during 2018. Southeast Fisheries Science Center, Protected Resources and Biodiversity Division, 75 Virginia Beach Dr., Miami, Florida 33140. PRD Contribution # PRD-2020-08. 55 pp.
Garrison, L.P., J. Ortega-Ortiz and G. Rappucci. 2020. Abundance of marine mammals in the waters of the U.S. Gulf of Mexico in the summer of 2017 and 2018. Southeast Fisheries Science Center, Protected Resources and


SHORT-FINNED PILOT WHALE (*Globicephala macrorhynchus*): Northern Gulf of Mexico Stock

STOCK DEFINITION AND GEOGRAPHIC RANGE

The short-finned pilot whale is distributed worldwide in tropical to temperate waters (Leatherwood and Reeves 1983). In the northern Gulf of Mexico (i.e., U.S. Gulf of Mexico), sightings of this species occur primarily on the continental slope west of 89°W (Figure 1; Mullin and Fulling 2004; Maze-Foley and Mullin 2006; Garrison and Aichinger Dias 2020). Short-finned pilot whales have been seen in all seasons during NMFS visual GulfCet aerial surveys of the northern Gulf of Mexico between 1992 and 1998 (Hansen et al. 1996; Mullin and Hoggard 2000).

![Figure 1. Distribution of short-finned pilot whale on-effort sightings from SEFSC vessel surveys during spring 1996–2001, summer 2003, spring 2004, summer 2009, summer 2017, and summer/fall 2018. Isobaths are the 200-m, 1,000-m, and 2,000-m depth contours. The darker line indicates the U.S. EEZ.](image)

All the cetacean species found in the oceanic northern Gulf of Mexico almost certainly occur in similar habitat beyond U.S. boundaries in the southern Gulf. There are fewer cetacean sighting and stranding records in the southern Gulf due to more limited effort. Nevertheless there are records for most oceanic species in the southern Gulf (e.g., Jefferson and Schiro 1997; Ortega Ortiz 2002; Ortega-Argueta *et al.* 2005; Jefferson *et al.* 2008; Vázquez Castán *et al.* 2009; Whitt *et al.* 2011). Because there are many confirmed records from Gulf of Mexico waters beyond U.S. boundaries (e.g., Jefferson and Schiro 1997, Ortega Ortiz 2002), short-finned pilot whales almost certainly occur throughout the oceanic Gulf of Mexico (Jefferson *et al.* 2008), which is therefore likely a transboundary stock with Cuba and/or Mexico. Because U.S. waters only comprise about 40% of the entire Gulf of Mexico and 35% of the oceanic (i.e., >200 m) Gulf of Mexico (Mullin and Fulling 2004), abundance and stock boundaries of oceanic species are poorly known. The oceanic Gulf of Mexico is also composed of waters belonging to Mexico and Cuba where there is currently little information on cetacean species abundance and distribution. U.S. waters comprise about 40% of the entire Gulf of Mexico and 35% of the oceanic (i.e., >200 m) Gulf of Mexico.

Short-finned pilot whales in the northern Gulf of Mexico are managed separately from those in the western North Atlantic. Although there have been no directed studies of the degree of demographic independence between the two areas, such separation is consistent with the fact that the two areas belong to distinct marine ecoregions (Spalding *et al.* 2007; Moore and Merrick 2011). However, there is some evidence to suggest there may be connectivity between...
the Gulf of Mexico and Atlantic, or at least between the eastern Gulf of Mexico and the Atlantic. A May 2011 mass stranding of 23 short-finned pilot whales in the Florida Keys was considered to be composed of northern Gulf of Mexico stock whales based on the stranding location, but two tagged and released individuals from this stranding travelled directly into the Atlantic (Wells et al. 2013). As Wells et al. (2009) note, it is difficult to determine the effects of stranding and rehabilitation on post-release behavior, so it is unknown whether these movements were representative of short-finned pilot whale ranging patterns in either the Gulf of Mexico or Atlantic Ocean. There are insufficient data to determine whether the northern Gulf of Mexico stock comprises multiple demographically independent populations. Additional morphological, acoustic, genetic, and/or behavioral data are needed to further delineate population structure within the Gulf of Mexico and across the broader geographic area. Studies are currently being conducted at the Southeast Fisheries Science Center to evaluate genetic population structure in short-finned pilot whales. Pending these results, the Globicephala macrorhynchus population occupying northern Gulf of Mexico waters is considered separate from both the U.S. western North Atlantic stock and short-finned pilot whales occupying Caribbean waters.

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

**POPULATION SIZE**

The best abundance estimate (Nbest) for the northern Gulf of Mexico short-finned pilot whale is 1,321 (CV=0.43; Table 1). This estimate is from summer 2017 and summer/fall 2018 oceanic surveys covering waters from the 200-m isobath to the seaward extent of the U.S. EEZ (Garrison et al. 2020). The best abundance estimate available for northern Gulf of Mexico short-finned pilot whales is 2,415 (CV=0.66; Table 1). This estimate is from a summer 2009 oceanic survey covering waters from the 200m isobath to the seaward extent of the U.S. EEZ.

**Earlier abundance estimates**

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

**Recent surveys and abundance estimates**

An abundance estimate for short-finned pilot whales was generated from vessel surveys conducted in the northern Gulf of Mexico from the continental shelf edge (~200-m isobath) to the seaward extent of the U.S. EEZ (Garrison et al. 2020). One survey was conducted from 2 July to 25 August 2017 and consisted of 7,302 km of on-effort trackline, and the second survey was conducted from 11 August to 6 October 2018 and consisted of 6,473 km of on-effort trackline within the surveyed strata. Both surveys used a double-platform data-collection procedure, which allowed estimation of the detection probability on the trackline using the independent observer approach assuming point independence (Laake and Borchers 2004). Abundance was calculated using mark-recapture distance sampling implemented in package mrds (version 2.21, Laake et al. 2020) in the R statistical programming language. This approach accounted for the effects of covariates (e.g., sea state, glare) on detection probability within the surveyed strip. The surveys were conducted in "passing mode" (e.g., Schwarz et al. 2010) while all prior surveys in the Gulf of Mexico have been conducted in "closing mode." The abundance estimate for this stock included sightings of unidentified small whales that were apportioned among identified species based on their relative density within the survey strata (Garrison et al. 2020). The inverse variance weighted mean abundance estimate for short-finned pilot whales in oceanic waters during 2017 and 2018 was 1,321 (CV=0.43; Table 1; Garrison et al. 2020). Unlike previous abundance estimates, this estimate was corrected for the probability of detection on the trackline. During summer 2009, a line-transect survey dedicated to estimating the abundance of oceanic cetaceans was conducted in the northern Gulf of Mexico. Survey lines were stratified in relation to depth and the location of the Loop Current. The abundance estimate for short-finned pilot whales in oceanic waters during 2009 was 2,415 (CV=0.66; Table 1).
**Table 1.** Most recent abundance estimate (N_best) and coefficient of variation (CV) of northern Gulf of Mexico short-finned pilot whales in oceanic waters (200 m to the offshore extent of the EEZ) based on an average from summer 2017 and summer/fall 2018 vessel surveys.

<table>
<thead>
<tr>
<th>Years</th>
<th>Area</th>
<th>N_best</th>
<th>CV</th>
</tr>
</thead>
<tbody>
<tr>
<td>2017, 2018</td>
<td>Gulf of Mexico</td>
<td>1,321</td>
<td>0.43</td>
</tr>
</tbody>
</table>

**Table 1.** Summary of abundance estimates for northern Gulf of Mexico short-finned pilot whales. Month, year and area covered during each abundance survey, and resulting abundance estimate (N_best) and coefficient of variation (CV).

<table>
<thead>
<tr>
<th>Month/Year</th>
<th>Area</th>
<th>N_best</th>
<th>CV</th>
</tr>
</thead>
<tbody>
<tr>
<td>Apr-Jun 1991-1994</td>
<td>Oceanic waters</td>
<td>353</td>
<td>0.89</td>
</tr>
<tr>
<td>Apr-Jun 1996-2001</td>
<td>Oceanic waters</td>
<td>2,388</td>
<td>0.48</td>
</tr>
<tr>
<td>Apr-Jun 2003-Apr-Jun 2004</td>
<td>Oceanic waters</td>
<td>716</td>
<td>0.34</td>
</tr>
<tr>
<td>Jun-Aug 2009</td>
<td>Oceanic waters</td>
<td>2,415</td>
<td>0.66</td>
</tr>
</tbody>
</table>

**Minimum Population Estimate**

The minimum population estimate (N_{min}) is the lower limit of the two-tailed 60% confidence interval of the log-normal distributed abundance estimate. This is equivalent to the 20th percentile of the log-normal distributed abundance estimate as specified by Wade and Angliss (1997). The best estimate of abundance for short-finned pilot whales is 1,321 to 2,415 (CV=0.43 to 0.66). The minimum population estimate for the northern Gulf of Mexico short-finned pilot whale is 934 (Table 2) to 1,456 short-finned pilot whales.

**Current Population Trend**

The statistical power to detect a trend in abundance for this stock is poor due to the relatively imprecise abundance estimates and long intervals between surveys. For example, the power to detect a precipitous decline in abundance (i.e., 50% decrease in 15 years) with estimates of low precision (e.g., CV>0.30) remains below 80% (alpha = 0.30) unless surveys are conducted on an annual basis (Taylor et al. 2007). In addition, because these surveys are restricted to U.S. waters, it is not possible to distinguish between changes in population size and Gulf-wide shifts in spatial distribution. Five point estimates of short-finned pilot whale abundance have been made based on data from surveys during 2003 (June-August), 2004 (April-June), 2009 (July-August), 2017 (July-August), and 2018 (August-October). Each of these surveys had a similar design and was conducted using the same vessel or a vessel with a similar observation platform. Prior year surveys only employed a single survey team. The estimates for 2017 and 2018 were conducted in "passing" mode which resulted in increased numbers of unidentified sightings and may have affected group size estimation for distant groups of dolphins and small whales. Apportioning unidentified species among identified taxa addresses the first issue, and there were no apparent relationships between sighting distance and estimated group size. The model for detection probability on the trackline from the summer 2017 survey was applied to the abundance estimates from the 2003, 2004, and 2009 surveys. The resulting abundance estimates were: 2003, N=2,740 (CV=0.52); 2004, N=587 (CV=0.88); 2009, N=4,788 (CV=0.74); 2017, N=1,274 (CV=0.54); and 2018,
N=1,403 (CV=0.71). Pairwise comparisons of the log-transformed means were conducted between years, and significant differences were assessed at alpha=0.10. P-values were adjusted for multiple comparisons. There were no significant differences between survey years. A trend analysis has not been conducted for this stock. Four point estimates of short-finned pilot whale abundance have been made based on data from surveys covering 1991–2009 (Table 1). The estimates vary by a maximum factor of nearly seven. To determine whether changes in abundance have occurred over this period, an analysis of all the survey data needs to be conducted which incorporates covariates (e.g., survey conditions, season) that could potentially affect estimates. It should be noted that since this is a transboundary stock and the abundance estimates are for U.S. waters only, it will be difficult to interpret any detected trends.

**CURRENT AND MAXIMUM NET PRODUCTIVITY RATES**

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

**POTENTIAL BIOLOGICAL REMOVAL**

Potential Biological Removal (PBR) is the product of the minimum population size, one half the maximum net productivity rate and a recovery factor (MMPA Sec. 3.16 U.S.C. 1362; Wade and Angliss 1997). The minimum population size is 934. The maximum productivity rate is 0.04, the default value for cetaceans. The recovery factor is 0.45 because the CV of the average mortality estimate is greater than 0.8 (Wade and Angliss 1997). The stock is of unknown status. PBR for the northern Gulf of Mexico short-finned pilot whale is 7.5 (Table 2).

**Table 2. Best and minimum abundance estimates for the northern Gulf of Mexico short-finned pilot whale stock with Maximum Productivity Rate (Rmax), Recovery Factor (Fr) and PBR.**

<table>
<thead>
<tr>
<th>Nbest</th>
<th>CV</th>
<th>Nmin</th>
<th>Fr</th>
<th>Rmax</th>
<th>PBR</th>
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<tr>
<td>1,321</td>
<td>0.43</td>
<td>934</td>
<td>0.4</td>
<td>0.04</td>
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</table>

**ANNUAL HUMAN-CAUSED MORTALITY AND SERIOUS INJURY**

The estimated mean annual fishery-related mortality and serious injury for this stock during 2009–2014–2018 was 0.5 short-finned pilot whales (CV=1.00; Table 2) due to interactions with the large pelagics longline fishery (see Fisheries Information sections below; Tables 3–4). Mean annual mortality and serious injury during 2014–2018 due to other human-caused actions (the Deepwater Horizon oil spill) was predicted to be 3.5. The minimum total mean annual human-caused mortality and serious injury for this stock during 2014–2018 was, therefore, 3.9.

**Table 3. Total annual estimated fishery-related mortality and serious injury for the northern Gulf of Mexico short-finned pilot whale stock.**

<table>
<thead>
<tr>
<th>Years</th>
<th>Source</th>
<th>Annual Avg.</th>
<th>CV</th>
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</thead>
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<tr>
<td>2014–2018</td>
<td>U.S. fisheries using observer data</td>
<td>0.4</td>
<td>1.00</td>
</tr>
</tbody>
</table>

**New Serious Injury Guidelines**

NMFS updated its serious injury designation and reporting process, which uses guidance from previous serious injury workshops, expert opinion, and analysis of historic injury cases to develop new criteria for distinguishing serious from non-serious injury (Angliss and DeMaster 1998; Andersen et al. 2008; NOAA 2012). NMFS defines serious injury as an “injury that is more likely than not to result in mortality.” Injury determinations for stock assessments revised in 2013 or later incorporate the new serious injury guidelines, based on the most recent 5-year period for which data are available.

**Fisheries Information**

There are two commercial fisheries that interact, or that potentially could interact, with this stock in the Gulf of Mexico. These are the Category I Atlantic Highly Migratory Species (high seas) longline fishery and the Category I Atlantic Ocean, Caribbean, Gulf of Mexico large pelagics longline fishery (Appendix III). Percent observer coverage...
(percentage of sets observed) for these longline fisheries for each year during 2014–2018 was 18, 19, 23, 13 and 20, respectively. There is very little effort within the Gulf of Mexico by the Atlantic Highly Migratory Species (high seas) longline fishery, and no takes of short-finned pilot whales within high seas waters of the Gulf of Mexico have been observed or reported thus far. The commercial fishery that interacts with this stock in the Gulf of Mexico is the Atlantic Ocean, Caribbean, Gulf of Mexico large pelagic longline fishery (Appendix III). Pelagic swordfish, tunas and billfish are the targets of the longline fishery operating in the northern Gulf of Mexico. The average annual serious injury and mortality in the Gulf of Mexico pelagic longline fishery for the five-year period from 2009–2014 to 2018–2013 is 0–0.44 (CV=1.00; Table 42; Garrison and Stokes 2016; 2017; 2019; 2020; in review). During the first quarter of 2016, one short-finned pilot whale was observed to be seriously injured. A short-finned pilot whale was observed to be seriously injured, and 1 additional short-finned pilot whale was released alive with no presumed serious injuries (both during quarter 2) (Garrison and Stokes 2019–2014). There were no reports of mortality or serious injury to short-finned pilot whales by this fishery during 1998–2012 (Yeung 1999; Yeung 2001; Garrison 2003; Garrison and Richards 2004; Garrison 2005; Fairfield Walsh and Garrison 2006; Fairfield Walsh and Garrison 2007; Fairfield and Garrison 2008; Garrison et al. 2009; Garrison and Stokes 2010; 2012a,b; 2013). Prior to the 2013 interactions, the most recent interaction documented occurred during 2006 when 1 short-finned pilot whale was observed entangled and released alive with no serious injury (Fairfield-Walsh and Garrison 2007). There was 1 logbook report of a fishery-related injury of a pilot whale in the northern Gulf of Mexico in 1991.

During the first and second quarters (15 April–15 June) of 2009–2014–2018–2013, observer coverage in the Gulf of Mexico pelagic longline fishery was greatly enhanced (approaching 55%) to collect more robust information on the interactions between pelagic longline vessels and spawning bluefin tuna. Therefore, the high annual observer coverage rates during 2009–2014–2018–2013 (Table 42) primarily reflect high coverage rates during the first and second quarters of each year. During these second quarters, this elevated coverage results in an increased probability that relatively rare interactions will be detected. Species within the oceanic Gulf of Mexico are presumed to be resident year-round; however, it is unknown if the bycatch rates observed during the first and second quarters are representative of that which occurs throughout the year.

### Table 4. Summary of the incidental mortality and serious injury of short-finned pilot whales by the pelagic longline commercial fishery including the years sampled (Years), the type of data used (Data Type), the annual observer coverage (Observer Coverage), the annual observed serious injury and mortality recorded by on-board observers, the annual estimated serious injury and mortality, the combined annual estimates of serious injury and mortality (Est. Combined Mortality), the estimated CV of the combined annual mortality estimates (Est. CVs), the mean of the combined annual mortality estimates, and the CV of the mean combined annual mortality estimate (CV of Mean).

<table>
<thead>
<tr>
<th>Fishery</th>
<th>Years</th>
<th>Data Typea</th>
<th>Observer Coverageb</th>
<th>Observed Serious Injuryb</th>
<th>Observed Mortalityc</th>
<th>Estimated Serious Injuryc</th>
<th>Est. Mort.</th>
<th>Est. Combined Mortality</th>
<th>Est. CVs</th>
<th>Mean Combined Annual Mortality</th>
<th>CV of Mean</th>
</tr>
</thead>
<tbody>
<tr>
<td>Pelagic Longline</td>
<td>2014</td>
<td>Obs. Data, Trip Logbook</td>
<td>0.18</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>-</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td></td>
<td>2015</td>
<td></td>
<td>0.19</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>-</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td></td>
<td>2016</td>
<td></td>
<td>0.23</td>
<td>1</td>
<td>0</td>
<td>2.2</td>
<td>0</td>
<td>2.2</td>
<td>1</td>
<td>0.4</td>
<td>1.00</td>
</tr>
<tr>
<td></td>
<td>2017</td>
<td></td>
<td>0.13</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>-</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td></td>
<td>2018</td>
<td></td>
<td>0.20</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>-</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td>TOTAL</td>
<td></td>
<td></td>
<td>0.15</td>
<td>0</td>
<td>0</td>
<td>2.2</td>
<td>0</td>
<td>2.2</td>
<td>0.4</td>
<td>1.00</td>
<td></td>
</tr>
</tbody>
</table>
Table 2. Summary of the incidental mortality and serious injury of northern Gulf of Mexico short-finned pilot whales in the pelagic longline commercial fishery including the years sampled (Years), the number of vessels active within the fishery (Vessels), the type of data used (Data Type), the annual observer coverage (Observer Coverage), the observed mortalities and serious injuries recorded by on-board observers, the estimated annual mortality and serious injury, the combined annual estimates of mortality and serious injury (Estimated Combined Mortality), the estimated CV of the combined estimates (Estimated CVs) and the mean of the combined estimates (CV in parentheses).

<table>
<thead>
<tr>
<th>Fishery</th>
<th>Years</th>
<th>Vessels</th>
<th>Data Type</th>
<th>Observer Coverage</th>
<th>Observed Mortality</th>
<th>Observed Serious Injury</th>
<th>Estimated Serious Injury</th>
<th>Estimated Mortality</th>
<th>Estimated Combined Mortality</th>
<th>Estimated CVs</th>
<th>Mean Annual Mortality</th>
</tr>
</thead>
<tbody>
<tr>
<td>Pelagic Longline</td>
<td>09-13</td>
<td>47, 46, 42, 47, 42</td>
<td>Logbook</td>
<td>0.22, 0.28, 0.18, 0.11, 0.25</td>
<td>0, 0, 0, 0, 0</td>
<td>0, 0, 0, 0, 0</td>
<td>0, 0, 0, 0, 0</td>
<td>0, 0, 0, 0, 0</td>
<td>0, 0, 0, 0, 0</td>
<td>NA, NA, NA, NA, 1.0</td>
<td>0.5 (1.00)</td>
</tr>
</tbody>
</table>

*a* Number of vessels in the fishery is based on vessels reporting effort to the pelagic longline logbook.

*b* Observer data (Obs. Data) are used to measure bycatch rates, and the data are collected within the Northeast Fisheries Observer Program. Mandatory logbook data were used to measure total effort for the longline fishery. These data are collected at the Southeast Fisheries Science Center (SEFSC). Observer coverage in the GOM is dominated by very high coverage rates during April-June associated with efforts to improve estimates of bluefin tuna bycatch.

*c* Proportion of sets observed.

Other Mortality

There have been were 93 reported strandings, including five mass strandings plus individual strandings, events of short-finned pilot whales in the Gulf of Mexico during 2009-2014-2018-2013 (Table 5; NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 21 May 2019-11 June 2014). All strandings occurred in the state of Florida. During 2014 there were three mass stranding events, and there was one mass stranding each during 2016 and 2017 (Table 5). During May 2011 there was a mass stranding of 23 short-finned pilot whales in the Florida Keys, including 8 live animals and 15 dead animals. During November 2013 there was a stranding of a single short-finned pilot whale in Florida. During December 2013 there was a mass stranding of an
An Unusual Mortality Event (UME) was declared for cetaceans in the northern Gulf of Mexico beginning 1 February/March 2010 and ending 31 July 2014, and as of September 2014, the event is still ongoing (Litz et al. 2014; https://www.fisheries.noaa.gov/national/marine-life-distress/2010-2014-cetacean-unusual-mortality-event-northern-gulf-mexico). It included cetaceans that stranded prior to the Deepwater Horizon (DWH) oil spill (see “Habitat Issues” below), during the spill, and after. Exposure to the DWH oil spill was determined to be the primary underlying cause of the elevated stranding numbers in the northern Gulf of Mexico after the spill (e.g., Schwacke et al. 2014; Venn-Watson et al. 2015; Colegrove et al. 2016; DWH NRDAT 2016; see Habitat Issues section). One short-finned pilot whale stranding from 2013 in Florida was considered to be part of the UME.

A population model was developed to estimate the injury and time to recovery for stocks affected by the DWH oil spill, taking into account long-term effects resulting from mortality, reproductive failure, reduced survival rates, and the proportion of the stock exposed to DWH oil (DWH MMIQT 2015). Overall, the model estimated that this stock experienced a 3% maximum reduction in population size due to the oil spill (DWH MMIQT 2015). The mortality projected for the years 2010–2013 due to the spill has not been reported previously. Based on the population model, it was projected that 34 short-finned pilot whales died during 2010–2013 (four year annual average of 8.6) due to elevated mortality associated with oil exposure (see Appendix VI). For the 2014–2018 reporting period of this SAR, the population model estimated 17 short-finned pilot whales died due to elevated mortality associated with oil exposure. The population model used to predict short-finned pilot whale mortality due to the DWH event has a number of sources of uncertainty. Model parameters (e.g., survival rates, reproductive rates, and life-history parameters) were derived from literature sources for short-finned pilot whales occupying waters outside of the Gulf of Mexico. In addition, proxy values for the effects of DWH oil exposure on both survival rates and reproductive success were applied based upon estimated values for common bottlenose dolphins in Barataria Bay. Finally, there was no estimation of uncertainty in model parameters or outputs.

### Table 5. Short-finned pilot whale strandings along the northern Gulf of Mexico coast, 2014–2018. Data are from the NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 21 May 2019. There were no strandings of short-finned pilot whales in Alabama, Louisiana, Mississippi, or Texas.

<table>
<thead>
<tr>
<th>State</th>
<th>2014</th>
<th>2015</th>
<th>2016</th>
<th>2017</th>
<th>2018</th>
<th>Total</th>
</tr>
</thead>
<tbody>
<tr>
<td>Florida</td>
<td>44a</td>
<td>0</td>
<td>35b</td>
<td>11c</td>
<td>3</td>
<td>93</td>
</tr>
</tbody>
</table>

*a. This included three mass strandings: one mass stranding of 4 animals; one mass stranding of 14 animals (6 of the estimated 14 animals were examined or handled by NMFS and included in the database); and one mass stranding of 39 animals (33 of the estimated 39 animals were examined or handled by NMFS and included in the database).

*b. This includes one mass stranding of 35 animals.

*c. This includes one mass stranding of 10 animals.

### HABITAT ISSUES

The Deepwater Horizon (DWH) MC252 drilling platform, located approximately 50 miles 80 km southeast of the Mississippi River Delta in waters about 1,500 m deep, exploded on 20 April 2010. The rig sank, and over 87 days up to 3.24.9 million barrels of oil were discharged from the wellhead until it was capped on 15 July 2010 (DWH NRDAT 2016; McNutt et al. 2012). During the response effort dispersants were applied extensively at the seafloor and at the sea surface (Lehr et al. 2010; OSAT 2010). In situ burning, or controlled burning of oil at the surface, was also used...
extensively as a response tool (Lehr et al. 2010). The oil, dispersant and burn residue compounds present ecological concerns (Buist et al. 1999; NOAA 2011). The magnitude of this oil spill was unprecedented in U.S. history, causing impacts to wildlife, natural habitats and human communities along coastal areas from western Louisiana to the Florida Panhandle (NOAA 2011). It could be years before the entire scope of damage is ascertained (NOAA 2011).

Shortly after the oil spill, the Natural Resource Damage Assessment (NRDA) process was initiated under the Oil Pollution Act of 1990. A variety of NRDA research studies were conducted to determine potential impacts of the spill on marine mammals. These studies estimated that 6% (95% CI: 4–9) of short-finned pilot whales in the Gulf were exposed to oil, that 3% (95% CI: 1–4) of females suffered from reproductive failure, and 2% (95% CI: 1–3) of short-finned pilot whales suffered adverse health effects (DWH MMIQT 2015). A population model estimated the stock experienced a maximum 3% reduction in population size (see Other Mortality section above).

Anthropogenic sound in the world’s oceans has been shown to affect marine mammals, with vessel traffic, seismic surveys, and active naval sonars being the main anthropogenic contributors to low- and mid-frequency noise in oceanic waters (e.g., Nowacek et al. 2015; Gomez et al. 2016; NMFS 2018). The long-term and population consequences of these impacts are less well-documented and likely vary by species and other factors. Impacts on marine mammal prey from sound are also possible (Carroll et al. 2017), but the duration and severity of any such prey effects on marine mammals are unknown. These studies have focused on identifying the type, magnitude, severity, length and impact of oil exposure to oceanic, continental shelf, coastal and estuarine marine mammals. For continental shelf and oceanic cetaceans, the NOAA-led efforts include: aerial surveys to document the distribution, abundance, species and exposure relative to oil from the DWH spill; and ship surveys to evaluate exposure to oil and other chemicals and to assess changes in animal behavior and distribution relative to oil exposure through visual and acoustic surveys, deployment of passive acoustic monitoring systems, collection of tissue samples, and deployment of satellite tags on sperm and Bryde’s whales.

Vessel and aerial surveys documented bottlenose dolphins, Atlantic spotted dolphins, rough-toothed dolphins, spinner dolphins, pantropical spotted dolphins, Risso’s dolphins, striped dolphins, sperm whales, dwarf/pygmy sperm whales and a Cuvier’s beaked whale swimming in oil or potentially oil-derived substances (e.g., sheen, mousse) in offshore waters of the northern Gulf of Mexico following the DWH oil spill. The effects of oil exposure on marine mammals depend on a number of factors including the type and mixture of chemicals involved, the amount, frequency and duration of exposure, the route of exposure (inhaled, ingested, absorbed, or external) and biomedical risk factors of the particular animal (Geraci 1990). In general, direct external contact with petroleum compounds or dispersants with skin may cause skin irritation, chemical burns and infections. Inhalation of volatile petroleum compounds or dispersants may irritate or injure the respiratory tract, which could lead to pneumonia or inflammation. Ingestion of petroleum compounds may cause injury to the gastrointestinal tract, which could affect an animal’s ability to digest or absorb food. Absorption of petroleum compounds or dispersants may damage kidney, liver and brain function in addition to causing immune suppression and anemia. Long-term chronic effects such as lowered reproductive success and decreased survival may occur (Geraci 1990).

STATUS OF STOCK

Short-finned pilot whales are not listed as threatened or endangered under the Endangered Species Act, and the northern Gulf of Mexico stock is not considered strategic under the MMPA. Total fishery-related mortality and serious injury for this stock is less than 10% of PBR and, therefore, can be considered to be insignificant and approaching zero mortality and serious injury rate. The status of short-finned pilot whales in the northern Gulf of Mexico, relative to OSP, is unknown. There was no statistically significant trend in population size for this stock in the northern Gulf of Mexico. There are insufficient data to determine the population trends for this stock.

REFERENCES


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