COMMON BOTTLENOSE DOLPHIN (*Tursiops truncatus truncatus*): Northern Gulf of Mexico Bay, Sound, and Estuary Stocks

NOTE – NMFS is in the process of writing individual stock assessment reports for each of the 31 bay, sound, and estuary stocks of common bottlenose dolphins in the northern Gulf of Mexico. To date, eight stocks have individual reports completed or drafted (West Bay, Galveston Bay/East Bay/Trinity Bay, Terrebonne-Timbalier Bay Estuarine System, Barataria Bay Estuarine System, Mississippi Sound/Lake Borgne/Bay Boudreau, Choctawhatchee Bay, St. Andrew Bay, and St. Joseph Bay), and the remaining 23 stocks are assessed in this report.

STOCK DEFINITION AND GEOGRAPHIC RANGE

Common bottlenose dolphins are distributed throughout the bays, sounds and estuaries of the Gulf of Mexico (Mullin 1988). The identification of demographically independent populations of common bottlenose dolphins in these waters is complicated by the high degree of behavioral variability exhibited by this species (Shane *et al.* 1986; Wells and Scott 1999; Wells 2003), and by the lack of requisite information for much of the region.

Distinct stocks are designated in each of 31 areas of contiguous, enclosed or semi-enclosed bodies of water adjacent to the northern Gulf of Mexico (i.e., U.S. Gulf of Mexico; Table 1; Figure 1). The genesis of the delineation of these stocks was work initiated in the 1970s in Sarasota Bay, Florida (Irvine and Wells 1972; Irvine *et al.* 1981), and in bays in Texas (Shane 1977; Gruber 1981). These studies documented year-round residency of individual common bottlenose dolphins in estuarine waters. As a result, the expectation of year-round resident populations was extended to bay, sound and estuary (BSE) waters across the northern Gulf of Mexico when the first stock assessment reports were published in 1995 (Blaylock *et al.* 1995). Since these early studies, long-term (year-round, multi-year) residency has been reported from nearly every site where photographic identification (photo-ID) or tagging studies have been conducted in the Gulf of Mexico. In Texas, long-term resident dolphins have been reported in the Matagorda-Espiritu Santo Bay area (Gruber 1981; Lynn and Würsig 2002), Aransas Pass (Shane 1977; Weller 1998), San Luis Pass (Maze and Würsig 1999; Irwin and Würsig 2004), and Galveston Bay (Bräger 1993; Bräger *et al.* 1994; Fertl 1994; Fazioli and Mintzer 2020). In Louisiana, Miller (2003) concluded the common bottlenose dolphin population in the Barataria Basin was relatively closed, and Wells *et al.* (2017) documented long-term, year-round residency in Barataria Bay based on telemetry data. Hubard *et al.* (2004) reported sightings of dolphins in Mississippi Sound that were known from tagging efforts there 12–15 years prior; long-term residency was further documented by Mullin *et al*. (2017). In Florida, long-term residency has been reported from Tampa Bay (Wells 1986; Wells *et al.* 1996b; Urian *et al.* 2009), Sarasota Bay (Irvine and Wells 1972; Irvine *et al.* 1981; Wells 1986; 1991; 2003; 2014; Wells *et al.* 1987; Scott *et al.* 1990), Lemon Bay (Wells *et al.* 1996a; Bassos-Hull *et al.* 2013), Charlotte Harbor/Pine Island Sound (Shane 1990; Wells *et al.* 1996a, 1997; Shane 2004; Bassos-Hull *et al.* 2013), and Gasparilla Sound (Bassos-Hull *et al.* 2013). In Sarasota Bay, which has the longest research history, up to five concurrent generations of identifiable residents have been identified, including individuals identified through more than four decades (Wells 2014). Maximum immigration and emigration rates of about 2–3% have been estimated (Wells and Scott 1990).

Genetic data also support the concept of relatively discrete BSE stocks. Analyses of mitochondrial DNA haplotype distributions indicate the existence of clinal variations along the Gulf of Mexico coastline (Duffield and Wells 2002). Differences in reproductive seasonality from site to site also suggest genetic-based distinctions between communities (Urian *et al.* 1996). Mitochondrial DNA analyses suggest finer-scale structural levels as well. For example, dolphins in Matagorda Bay, Texas, appear to be a localized population, and differences in haplotype frequencies distinguish among adjacent communities in Tampa Bay, Sarasota Bay, and Charlotte Harbor/Pine Island Sound, along the central west coast of Florida (Duffield and Wells 1991; 2002). Additionally, Sellas *et al.* (2005) examined population subdivision among dolphins sampled in Sarasota Bay, Tampa Bay, Charlotte Harbor, Matagorda Bay, and the coastal Gulf of Mexico (1–12 km offshore) from just outside Tampa Bay to the southern end of Lemon Bay, and found evidence of significant population structure among all areas on the basis of both mitochondrial DNA control region sequence data and nine nuclear microsatellite loci. Rosel *et al.* (2017) also identified significant population differentiation between estuarine residents of Barataria Bay and the adjacent coastal stock. The Sellas *et* *al.* (2005) and Rosel *et al.* (2017) findings support the separate identification of BSE populations from those occurring in adjacent Gulf coastal waters.

In many cases, residents occur primarily in BSE waters, with limited movements through passes to the Gulf of Mexico (Shane 1977, 1990; Gruber 1981; Irvine *et al.* 1981; Maze and Würsig 1999; Lynn and Würsig 2002; Fazioli *et al.* 2006). These habitat use patterns are reflected in the ecology of the dolphins in some areas; for example, residents of Sarasota Bay, Florida, lacked squid in their diet, unlike non-resident dolphins stranded on nearby Gulf beaches (Barros and Wells 1998). However, in some areas year-round residents may co-occur with non-resident dolphins. For example, about 14–17% of group sightings involving resident Sarasota Bay dolphins include at least one non-resident as well (Wells *et al.* 1987; Fazioli *et al.* 2006). Mixing of inshore residents and non-residents has been seen at San Luis Pass, Texas (Maze and Würsig 1999), Cedar Keys, Florida (Quintana-Rizzo and Wells 2001), and Pine Island Sound, Florida (Shane 2004). Non-residents exhibit a variety of movement patterns, ranging from apparent nomadism recorded as transience to a given area, to apparent seasonal or non-seasonal migrations. Passes, especially the mouths of the larger estuaries, serve as mixing areas. For example, dolphins from several different areas were documented at the mouth of Tampa Bay, Florida (Wells 1986), and most of the dolphins identified in the mouths of Galveston Bay and Aransas Pass, Texas, were considered transients (Henningsen 1991; Bräger 1993; Weller 1998).

Seasonal movements of dolphins into and out of some of the bays, sounds and estuaries have also been documented. In Sarasota Bay, Florida, and San Luis Pass, Texas, residents have been documented using Gulf coastal waters more frequently in fall/winter, and inshore waters more in spring/summer (Irvine *et al.* 1981; Maze and Würsig 1999). Fall/winter increases in abundance have been noted for Tampa Bay (Scott *et al.* 1989) and are thought to occur in Matagorda Bay (Gruber 1981; Lynn and Würsig 2002) and Aransas Pass (Shane 1977; Weller 1998). Spring/summer increases in abundance occur in Mississippi Sound (Hubard *et al.* 2004) and are thought to occur in Galveston Bay (Henningsen 1991; Bräger 1993; Fertl 1994). However, Mullin *et al.* (2017) found that seasonal fluctuations in Mississippi Sound were less than previously reported.

Spring and fall increases in abundance have been reported for St. Joseph Bay, Florida. Mark-recapture abundance estimates were highest in spring and fall and lowest in summer and winter (Table 1; Balmer *et al.* 2008). Individuals with low site-fidelity indices were sighted more often in spring and fall, whereas individuals sighted during summer and winter displayed higher site-fidelity indices. In conjunction with health assessments, 23 dolphins were radio tagged during April 2005 and July 2006. Dolphins tagged in spring 2005 displayed variable utilization areas and variable site fidelity patterns. In contrast, during summer 2006 the majority of radio-tagged individuals displayed similar utilization areas and moderate to high site-fidelity patterns. The results of the studies suggest that during summer and winter St. Joseph Bay hosts dolphins that spend most of their time within this region, and these may represent a resident community. In spring and fall, St. Joseph Bay is visited by dolphins that range outside of this area (Balmer *et al.* 2008).

The current BSE stocks are designated as described in Table 1. There are some estuarine areas that are not currently part of any stock's range. Many of these are areas that dolphins cannot readily access. For example, the marshlands between Galveston Bay and Sabine Lake and between Sabine Lake and Calcasieu Lake are fronted by long, sandy beaches that prohibit dolphins from entering the marshes. The region between the Calcasieu Lake and Vermilion Bay/Atchafalaya Bay stocks has some access, but these marshes are predominantly freshwater rather than saltwater marshes, making them unsuitable for long-term survival of a viable population of common bottlenose dolphins. In other regions, there is insufficient estuarine habitat to harbor a demographically independent population, for instance between the Matagorda Bay and West Bay Stocks in Texas, and/or sufficient isolation of the estuarine habitat from coastal waters. The regions between the south end of the Estero Bay Stock area to just south of Naples and between Little Sarasota Bay and Lemon Bay are highly developed and contain little appropriate habitat. South of Naples to Marco Island and Gullivan Bay is also not currently covered within a stock boundary. This region contains common bottlenose dolphins, but the relationship of any dolphins in this region to other BSE stocks is unknown. They may be members of the Gullivan Bay to Chokoloskee Bay stock as there is passage behind Marco Island that would allow dolphins to move north. The regions between Apalachee Bay and Cedar Key/Waccasassa Bay, between Crystal Bay and St. Joseph Sound, and between Chokoloskee Bay and Whitewater Bay comprise thin strips of marshland with no barriers to adjacent coastal waters. Further work is necessary to determine whether year-round resident dolphins use these thin marshes or whether dolphins in these areas are members of the coastal stock that use the fringing marshland as well. Finally, the region between the eastern border of the Barataria Bay Estuarine System Stock and the Mississippi River Delta Stock to the east may harbor dolphins, but the area is small and work is necessary to determine whether any dolphins utilizing this habitat come from an adjacent BSE stock.

As more information becomes available, combination or division of these stocks, or alterations to stock boundaries, may be warranted. For example, research based on photo-ID data collected by Bassos-Hull *et al.* (2013) recommended combining Lemon Bay with Gasparilla Sound/Charlotte Harbor/Pine Island Sound. Therefore, these stocks have been combined (see Table 1). However, it should be noted this change was made in the absence of genetic data and could be revised again in the future when genetic data are available. Additionally, a number of geographically and socially distinct subgroupings of dolphins in regions such as Tampa Bay, Charlotte Harbor, Pine Island Sound, Barataria Bay, Aransas Pass, and Matagorda Bay have been identified (Shane 1977; Gruber 1981; Wells *et al.* 1996a, 1996b, 1997, 2017; Lynn and Würsig 2002; Urian 2002; Rosel *et al.* 2017). For Tampa Bay, Urian *et al.* (2009) described five discrete communities (including the adjacent Sarasota Bay community) that differed in their social interactions and ranging patterns. Structure was found despite a lack of physiographic barriers to movement within this large, open embayment. Urian *et al.* (2009) further suggested that fine-scale structure may be a common element among common bottlenose dolphins in the southeastern U.S. and recommended that management should account for fine-scale structure that exists within current stock designations. These results indicate that it is plausible some of these estuarine stocks, particularly those in larger bays and estuaries, comprise multiple demographically-independent populations.

Table 1. Most recent common bottlenose dolphin abundance estimate (N_{est}), coefficient of variation of N_{est} (CV N_{est}), minimum population estimate (N_{min}), Potential Biological Removal (PBR), year of the most recent abundance estimate and associated publication (Year), and minimum counts of annual human-caused mortality and serious injury (HCMSI) in northern Gulf of Mexico bay, sound and estuary stocks. When estimates are based on data collected more than eight years ago, they are considered unknown or undetermined for management purposes. Blocks refer to aerial survey blocks illustrated in Figure 1. UNK – unknown; UND – undetermined. For each stock denoted with a † symbol, please refer to the stand-alone report for this stock.

Blocks	Gulf of Mexico Estuary	N_{est}	CV N_{est}	N_{min}	PBR	Year (Reference)	Minimum Annual HCMSI, 2015–2019
B51	Laguna Madre	80	1.57	UNK	UND	1992 (A)	0.8
B52	Nueces Bay/Corpus Christi Bay	58	0.61	UNK	UND	1992 (A)	0.2
B50	Copano Bay/Aransas Bay/ San Antonio Bay/ Redfish Bay/Espiritu Santo Bay	55	0.82	UNK	UND	1992 (A)	0.6
B54	Matagorda Bay/ Tres Palacios Bay/Lavaca Bay	61	0.45	UNK	UND	1992 (A)	0.4
B55	West Bay†						
B56	Galveston Bay/East Bay/ Trinity Bay†						
B57	Sabine Lake	122[a]	0.19	104	0.9	2017 (B)	0
B58	Calcasieu Lake	0[b]	-	-	UND	1992 (A)	0.2
B59	Vermilion Bay/West Cote Blanche Bay/Atchafalaya Bay	0[b]	-	-	UND	1992 (A)	0
B60	Terrebonne-Timbalier Bay Estuarine System†						
B61	Barataria Bay Estuarine System†						

B30	Mississippi River Delta	1,446[c]	0.19	1,238	11	2017–2018 (C)	9.2
B02–05, 29, 31	Mississippi Sound/ Lake Borgne/Bay Boudreau†						
B06	Mobile Bay/Bonsecour Bay	122	0.34	UNK	UND	1993 (A)	16.0
B07	Perdido Bay	0[b]	-		UND	1993 (A)	0.8
B08	Pensacola Bay/East Bay	33	0.80	UNK	UND	1993 (A)	0.4
B09	Choctawhatchee Bay†						
B10	St. Andrew Bay†						
B11	St. Joseph Bay†						
B12–13	St. Vincent Sound/Apalachicola Bay/St. George Sound	439	0.14	UNK	UND	2007 (D)	0.2
B14–15	Apalachee Bay	491	0.39	UNK	UND	1993 (A)	0
B16	Waccasassa Bay/Withlacoochee Bay/Crystal Bay	UNK	-	UNK	UND	-	0.4
B17	St. Joseph Sound/ Clearwater Harbor	UNK	-	UNK	UND	-	0.8[d]
B32–34	Tampa Bay	UNK	-	UNK	UND	-	3.0
B20, 35	Sarasota Bay/Little Sarasota Bay	158	0.27	126	1.0	2015 (E)	0.2[e]
B21–23	Pine Island Sound/ Charlotte Harbor/ Gasparilla Sound/Lemon Bay	826	0.09	UNK	UND	2006 (F)	1.0[f]
B36	Caloosahatchee River	0[b]	-	-	UND	1985 (G)	0.4[g]
B24	Estero Bay	UNK	-	UNK	UND	-	0.4
B25	Chokoloskee Bay/Ten Thousand Islands/Gullivan Bay	UNK	-	UNK	UND	-	0.2
B27	Whitewater Bay	UNK	-	UNK	UND	-	0
B28	Florida Keys (southwest Marathon Key to Marquesas Keys)	UNK	-	UNK	UND	-	0.2

References: A – Blaylock and Hoggard 1994; B – Ronje *et al.* 2020; C – Garrison *et al.* 2021; D – Tyson *et al.* 2011; E – Tyson and Wells 2016; F – Bassos-Hull *et al.* 2013; G – Scott *et al.* 1989

Notes:

a. Winter seasonal estimate, Selective dataset.

b. During earlier surveys (Scott *et al.* 1989), the range of seasonal abundances was as follows: Calcasieu Lake, 0–6 (0.34); Vermilion Bay/West Cote Blanche Bay/Atchafalaya Bay, 0–0; Perdido Bay, 0–0; Lemon Bay, 0–15 (0.43); and Caloosahatchee River, 0–0.

c. Abundance estimate utilizes density estimate from adjacent waters. See Garrison *et al.* (2021) for details.

d. The minimum count would have been higher (1.2 instead of 0.8) had it not been for mitigation efforts.

e. The minimum count would have been higher (0.4 instead of 0.2) had it not been for mitigation efforts.

f. The minimum count would have been higher (1.4 instead of 1.0) had it not been for mitigation efforts.

g. The minimum count would have been higher (0.6 instead of 0.4) had it not been for mitigation efforts.

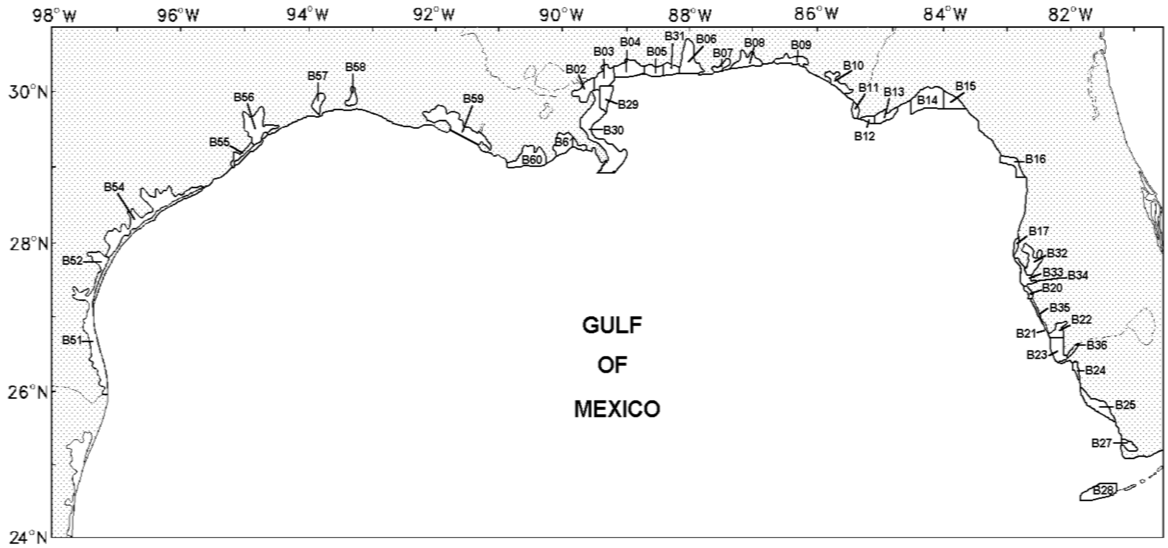

Figure 1. Northern Gulf of Mexico bays, sounds, and estuaries. Each of the alpha-numerically designated blocks corresponds to one of the NMFS Southeast Fisheries Science Center logistical aerial survey areas listed in Table 1. The common bottlenose dolphins inhabiting each bay, sound, or estuary are considered to comprise a unique stock for purposes of this assessment. Eight stocks have their own stock assessment report (see Table 1).

POPULATION SIZE

Population size estimates for most of these stocks are more than eight years old and therefore the current population sizes for all but three are considered unknown (Wade and Angliss 1997). However, a capture-mark-recapture population size estimate is available for Sarasota Bay/Little Sarasota Bay for 2015 (Tyson and Wells 2016) and Sabine Lake for 2017 (Ronje *et al.* 2020). Recent aerial survey line-transect population size estimates are available for Mississippi River Delta for 2017–2018 (Garrison *et al.* 2021; Table 1). Population size estimates for many stocks were generated from preliminary analyses of line-transect data collected during aerial surveys conducted in September–October 1992 in Texas and Louisiana and in September–October 1993 in Louisiana, Mississippi, Alabama, and the Florida Panhandle (Blaylock and Hoggard 1994; Table 1). Standard line-transect perpendicular sighting distance analytical methods (Buckland *et al.* 1993) and the computer program DISTANCE (Laake *et al.* 1993) were used.

Minimum Population Estimate

The population sizes for all but three stocks are currently unknown and the minimum population estimates are given for those three stocks in Table 1. The minimum population estimate is the lower limit of the two-tailed 60% confidence interval of the log-normally distributed best abundance estimate. This is equivalent to the 20th percentile of the log-normal distribution as specified by Wade and Angliss (1997). The minimum population estimate was calculated for each block from the estimated population size and its associated coefficient of variation.

Current Population Trend

The data are insufficient to determine population trends for most of the Gulf of Mexico BSE common bottlenose dolphin stocks.

CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

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

POTENTIAL BIOLOGICAL REMOVAL

Potential Biological Removal (PBR) is the product of minimum population size, one-half the maximum productivity rate and a recovery factor (Wade and Angliss 1997). The recovery factor is 0.45 for Louisiana, Mississippi, and Alabama BSE stocks because the CV of the shrimp trawl mortality estimate for those stocks is greater than 0.6. The recovery factor is 0.4 for Texas and Florida BSE stocks because the CV of the shrimp trawl mortality estimate for those stocks is greater than 0.8 (Wade and Angliss 1997). PBR is undetermined for all but three stocks because the population size estimates are more than eight years old. PBR for those stocks with population size estimates less than eight years old is given in Table 1.

ANNUAL HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

The total annual human-caused mortality and serious injury for these stocks of common bottlenose dolphins during 2015–2019 is unknown. Minimum estimates of human-caused mortality and serious injury for each stock are given in Table 1. These estimates are considered a minimum because: 1) not all fisheries that could interact with these stocks are observed and/or observer coverage is very low, 2) stranding data are used as an indicator of fishery-related interactions and not all dead animals are recovered by the stranding network (Peltier *et al*. 2012; Wells *et al*. 2015), 3) cause of death is not (or cannot be) routinely determined for stranded carcasses, 4) the estimate of fishery-related interactions includes an actual count of verified fishery-caused deaths and serious injuries and should be considered a minimum (NMFS 2016), 5) the estimate does not include shrimp trawl bycatch because estimates are not available for individual BSE stocks (see Shrimp Trawl section), and 6) various assumptions were made in the population model used to estimate population decline for northern Gulf of Mexico BSE stocks impacted by the *Deepwater Horizon* (DWH) oil spill.

Fishery Information

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

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

Shrimp Trawl

During 2015–2019, based on limited observer coverage in Louisiana BSE waters under the NMFS MARFIN program, there was one observed mortality and no observed serious injuries of common bottlenose dolphins from Gulf of Mexico BSE stocks by commercial shrimp trawls. Between 1997 and 2019, 13 common bottlenose dolphins and nine unidentified dolphins, which could have been either common bottlenose dolphins or Atlantic spotted dolphins, became entangled in the net, lazy line, turtle excluder device or tickler chain gear in the commercial shrimp trawl fishery in the Gulf of Mexico (Soldevilla *et al.* 2015, 2016, 2021). All dolphin bycatch interactions resulted in mortalities except for one unidentified dolphin that was released alive without serious injury in 2009 (Maze-Foley and Garrison 2016). Soldevilla *et al*. (2015, 2016, 2021) provided mortality estimates calculated from analysis of shrimp fishery effort data and NMFS's Observer Program bycatch data. Limited observer coverage in Louisiana BSE waters started in 2015, but has not yet reached sufficient levels for estimating BSE bycatch rates; therefore time-area stratified bycatch rates were extrapolated into inshore waters to estimate the most recent five-year unweighted mean mortality estimate for 2015–2019 based on inshore fishing effort (Soldevilla *et al.* 2021). The 4-area (Texas, Louisiana, Mississippi/Alabama, Florida) stratification method was chosen because it best approximates how fisheries operate (Soldevilla *et al*. 2015, 2016, 2021). The BSE stock mortality estimates were aggregated at the state area level as this was the spatial resolution at which fishery effort is modeled (e.g., Nance *et al.* 2008). The mean annual mortality estimates for the BSE stocks were as follows: Texas BSE (from Galveston Bay/East Bay/Trinity Bay south to Laguna Madre): 0.4 (CV=1.62); Louisiana BSE (from Sabine Lake east to Barataria Bay): 45 (CV=0.65);

Mississippi/Alabama BSE (from Mississippi River Delta east to Mobile Bay/Bonsecour Bay): 33 (CV=0.70); and Florida BSE (from Perdido Bay east and south to the Florida Keys): 0.7 (CV=1.58). These estimates do not include skimmer trawl effort, which accounts for 61% of shrimp fishery effort in western Louisiana, and 38% of shrimp fishery effort in eastern Louisiana, Mississippi, and Alabama inshore waters, because observer program coverage of skimmer trawls is limited. Limitations and biases of annual bycatch mortality estimates are described in detail in Soldevilla *et al.* (2015, 2016, 2021). It should be noted that because bycatch for individual BSE stocks cannot be quantified at this time, shrimp trawl bycatch is not being included in the annual human-caused mortality and serious injury total for any BSE stock.

During 2015–2019, stranding data documented two mortalities of common bottlenose dolphins associated with entanglement in shrimp trawl gear. Both mortalities occurred in 2016—one in Pensacola Bay and one in Perdido Bay.

During 2016 the Marine Mammal Authorization Program (MMAP) documented a self-reported incidental take (mortality) of a common bottlenose dolphin by a commercial fisherman trawling in Mobile Bay. The dolphin was entangled in the lazy line of the gear.

Menhaden Purse Seine

During 2015–2019 there was one mortality documented within waters of the Mississippi River Delta Stock that involved the menhaden purse seine fishery (Table 2). Through the Marine Mammal Authorization Program (MMAP), one animal was reported as entangled within a purse seine during 2018. There is currently no observer program for the Gulf of Mexico menhaden purse seine fishery.

Without an ongoing observer program, it is not possible to obtain statistically reliable incidental mortality and serious injury rates for this fishery, and the stocks from which common bottlenose dolphins are being taken. The documented mortality in this commercial fishery represents a minimum known count of mortalities and serious injuries in the last five years.

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

During 2015–2019, there were nine documented interactions between trap/pot fisheries and BSE stocks. During 2019, two serious injuries occurred, one due to entanglement in commercial spiny lobster trap/pot gear, ascribed to the Florida Keys Stock, and the second due to entanglement in unidentified trap/pot gear, ascribed to the Waccassasa Bay Stock. Also during 2019, an animal was disentangled from commercial blue crab trap/pot gear and released alive. It could not be determined if the animal was seriously injured following mitigation efforts (the initial determination was seriously injured). This animal was ascribed to the Caloosahatchee River Stock. During 2017, one mortality and one serious injury occurred, both due to entanglement in commercial blue crab trap/pot gear. The mortality was ascribed to the Caloosahatchee River Stock, and the serious injury to the Waccasassa Bay Stock. During 2016, one animal was partially disentangled from unidentified trap/pot gear and released alive seriously injured. This animal was ascribed to the Pine Island Sound/Charlotte Harbor/Gasparilla Sound/Lemon Bay Stock. Also in 2016, an animal was disentangled from commercial stone crab trap/pot gear and released alive not seriously injured following disentanglement efforts (the initial determination was seriously injured). This animal was ascribed to the Sarasota Bay/Little Sarasota Bay Stock. During 2015, one mortality occurred due to entanglement in commercial blue crab trap/pot gear. This animal was ascribed to the Mobile Bay/Bonsecour Bay Stock. Also in 2015, one animal was disentangled and released alive from unidentified crab trap/pot gear but it could not be determined if the animal was seriously injured following mitigation efforts (the initial determination was seriously injured). This freeze-branded animal was known to belong to the Sarasota Bay/Little Sarasota Bay Stock. The specific fishery could not be identified for the trap/pot gear involved in several of the live releases. The mortality and the animals released alive were all included in the stranding database (NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 25 August 2020) and are included in the stranding totals in Table 4. The details for serious determinations for the live animals are provided in Maze-Foley and Garrison (2021).

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

Gillnet

During 2015–2019, there was one documented interaction with gillnet gear and a BSE stock. During 2019, a stranded carcass was recovered with gillnet gear wrapped around its rostrum, and it was ascribed to the St. Vincent

Sound, Apalachicola Bay, St. George Sound Stock. There has been limited observer coverage of this fishery in state waters. During 2012–2018, NMFS placed observers on commercial vessels (state permitted gillnet vessels) in the coastal waters of Alabama, Mississippi, and Louisiana (Mathers *et al.* 2016). No takes were observed in state waters during this time. However, stranding data indicate that gillnet interactions do occur, causing mortality and serious injury. During 2015–2019, three stranded common bottlenose dolphins were recovered with markings indicative of interaction with gillnet gear (Read and Murray 2000), but no gear was attached to the carcasses and it is unknown whether the interactions with the gear contributed to the death of these animals. Two of the strandings were ascribed to the Mobile Bay Stock and one to the Perdido Bay Stock. Because there is no observer program within BSE waters, it is not possible to estimate total mortalities and serious injuries incidental to gillnet fisheries.

In 1995, a Florida state constitutional amendment banned gillnets and large nets from bays, sounds, estuaries, and other inshore waters. Commercial and recreational gillnet fishing is also prohibited in Texas state waters.

For details on research-related entanglements in gillnet gear, see the Other Mortality section and Table 3 below.

Hook and Line (Rod and Reel)

During 2015–2019 there were 20 documented interactions (entanglements or ingestions) between hook and line gear and BSE stocks—14 mortalities and six live animals (disentanglement efforts were made for four of the six). The stranding data indicate that for six of the 14 mortalities, the hook and line gear interaction contributed to the cause of death. For five mortalities, evidence suggested the hook and line gear interaction was incidental and was not a contributing factor to cause of death. For three mortalities, it could not be determined if the hook and line gear interaction contributed to cause of death. Two live animals were considered seriously injured and no disentanglement efforts were made. Attempts were made to disentangle the remaining four live animals from hook and line gear. All four were considered seriously injured by the gear prior to mitigation efforts, but based on observations during mitigations, three animals were considered not seriously injured post-mitigation. For the remaining live animal, following mitigation it could not be determined if the animal was seriously injured. In summary, the evidence available from stranding data suggested that at least six mortalities and two serious injuries to animals from BSE stocks resulted from interactions with rod and reel hook and line gear. This number would have been higher without mitigation efforts to disentangle four live animals.

Interactions by year with hook and line gear were as follows: During 2015, there was one mortality. During 2016, there were three mortalities, two live animals considered seriously injured, and one live animal for which it could not be determined if it was seriously injured following disentanglement efforts (the initial determination was seriously injured). During 2017, there were four mortalities. During 2018, there were three mortalities and two live animals considered not seriously injured following disentanglement efforts (the initial determinations were seriously injured; one animal was initially sighted in 2018 and later disentangled in 2019). During 2019, there were three mortalities and one live animal considered not seriously injured following disentanglement efforts (the initial determination was seriously injured).

The mortalities and serious injuries likely involved animals from the following BSE stocks: Laguna Madre, Mobile Bay/Bonsecour Bay, Perdido Bay, Waccasassa Bay/Withlacoochee Bay/Crystal Bay, St. Joseph Sound/Clearwater Harbor, Tampa Bay, Sarasota Bay/Little Sarasota Bay, Pine Island Sound/Charlotte Harbor/Gasparilla Sound/Lemon Bay, and Estero Bay.

All mortalities and live entanglements were included in the stranding database (NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 25 August 2020) and are included in the stranding totals presented in Table 4. The details for serious determinations for the live animals are provided in Maze-Foley and Garrison (2021).

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

Other Mortality

A population model was developed to estimate long-term injury to stocks affected by the DWH oil spill (see Habitat Issues section), taking into account long-term effects resulting from mortality, reproductive failure, and reduced survival rates (DWH MMIQT 2015; Schwacke *et al*. 2017). For the Mississippi River Delta Stock, the model predicted the stock experienced a 71% (95%CI: 40–97) maximum reduction in population size due to the oil spill (DWH MMIQT 2015; Schwacke *et al.* 2017), and for the years 2015–2019, the model projected 45 mortalities. For the Mobile Bay/Bonsecour Bay Stock, the model predicted a 31% (95% CI: 20–51) maximum reduction in population size due to the oil spill (DWH MMIQT 2015; Schwacke *et al*. 2017), and for the years 2015–2019, the model projected 73 mortalities. This population model has a number of sources of uncertainty. Because no current abundance estimates existed at the time of the spill, the baseline population sizes were estimated from studies initiated after initial exposure to DWH oil occurred. Therefore, it is possible that the pre-spill population sizes were larger than this baseline level and some mortality occurring early in the event was not quantified. The duration of elevated mortality and reduced reproductive success after exposure is unknown, and expert opinion was used to predict the rate at which these parameters would return to baseline levels. Where possible, uncertainty in model parameters was included in the estimates of excess mortality by re-sampling from statistical distributions of the parameters (DWH MMIQT 2015; DWH NRDAT 2016; Schwacke *et al*. 2017).

There were two live dolphins during 2015–2019 that were entangled in unidentified fishing gear or unidentified gear, and one occurred in the Pine Island Sound/Charlotte Harbor/Gasparilla Sound/Lemon Bay Stock area in 2017 and the other occurred in Perdido Bay in 2018. The animal from 2018 was considered not seriously injured, and the 2017 animal was initially considered seriously injured, but following mitigation efforts, was released alive without serious injury (Maze-Foley and Garrison 2018). During 2015, an animal in the St. Joseph Sound/Clearwater Harbor Stock area (Florida) was released alive without serious injury following entrapment behind an oil boom (Maze-Foley and Garrison 2018). During 2016, there was a dead dolphin in the Copano Bay/Aransas Bay/San Antonio Bay/Redfish Bay/Espiritu Santo Bay Stock area found entangled in electrical cord. It is unknown whether the entanglement contributed to the death of the animal. All of these cases were included in the stranding database (NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 25 August 2020) and are included in the stranding totals presented in Table 4.

In addition to animals included in the stranding database, during 2015–2019, there were 42 at-sea observations in BSE stock areas of common bottlenose dolphins entangled in fishing gear or unidentified gear (hook and line, crab trap/pot and unidentified gear/line/rope) or displaying vessel-strike injuries. In 27 of these cases, the animals were seriously injured; in six cases the animals were not seriously injured, and for the remaining nine cases, it could not be determined (CBD) if the animals were seriously injured (Maze-Foley and Garrison 2021; see Table 2).

Table 2. At-sea observations of common bottlenose dolphins entangled in fishing gear or unidentified gear during 2015–2019, including the serious injury determination (mortality, serious injury, not a serious injury [Not serious], or could not be determined [CBD] if seriously injured) and stock to which each animal likely belonged based on sighting location. Further details can be found in Maze-Foley and Garrison (2021).

Year (Identifier from Maze-Foley and Garrison [2021])	Determination	Stock
2015 (row 92)	Serious injury	Calcasieu Lake
2015 (row 93)	Not serious	Tampa Bay
2015 (row 98)	Serious injury	Tampa Bay
2015 (row 99)	Serious injury	Laguna Madre
2015 (row 101)	CBD	Sarasota Bay/Little Sarasota Bay
2015 (row 102)	Serious injury	St. Joseph Sound/Clearwater Harbor
2015 (row 104)	CBD	Mobile Bay/Bonsecour Bay (or Northern Coastal)
2015 (row 106)	Not serious	Sarasota Bay/Little Sarasota Bay
2015 (row 109)	CBD	Apalachee Bay

2016 (row 120)	Serious injury	Laguna Madre
2016 (row 126)	CBD	St. Joseph Sound/Clearwater Harbor
2016 (row 127)	Serious injury	Mobile Bay/Bonsecour Bay
2017 (row 129)	CBD	Sarasota Bay/Little Sarasota Bay
2017 (row 130)	CBD	Sarasota Bay/Little Sarasota Bay
2017 (row 131)	Serious injury	undefined stock area (Miller's Bayou, Florida; in between the Waccasassa Bay/Withlacoochee Bay/Crystal Bay Stock and the St. Joseph Sound/Clearwater Harbor Stock)
2017 (row 135)	Serious injury	Sarasota Bay/Little Sarasota Bay
2017 (row 137)	Serious injury	Copano Bay/Aransas Bay/San Antonio Bay/ Redfish Bay/Espiritu Santo Bay
2017 (row 139)	Serious injury	Tampa Bay
2017 (row 140)	Serious injury	Tampa Bay
2017 (row 148)	Serious injury	Tampa Bay
2017 (row 150)	Serious injury	St. Joseph Sound/Clearwater Harbor
2018 (row 153)	Serious injury	Tampa Bay (or Eastern Coastal)
2018 (row 155)	CBD	Tampa Bay
2018 (row 156)	CBD	St. Joseph Sound/Clearwater Harbor (or Eastern Coastal)
2018 (row 158)	Not serious	Chokoloskee Bay/Ten Thousand Islands/ Gullivan Bay
2018 (row 160)	Serious injury	Estero Bay
2018 (row 162)	Serious injury	Laguna Madre
2018 (row 166)	Serious injury	Perdido Bay
2018 (row 168)	CBD	Perdido Bay
2018 (row 171)	Serious injury	Tampa Bay
2018 (row 25, vessel strike tab)	Serious injury	Perdido Bay
2019 (row 172)	Not serious	Chokoloskee Bay/Ten Thousand Islands/ Gullivan Bay
2019 (row 173)	Serious injury	Chokoloskee Bay/Ten Thousand Islands/ Gullivan Bay

2019 (row 175)	Serious injury	Pine Island Sound/Charlotte Harbor/ Gasparilla Sound/Lemon Bay
2019 (row 176)	Serious injury	Tampa Bay
2019 (row 179)	Not serious	St. Joseph Sound/Clearwater Harbor (or Eastern Coastal)
2019 (row 182/183)	Serious injury	Tampa Bay
2019 (row 189)	Serious injury	Tampa Bay
2019 (row 190)	Serious injury	Tampa Bay
2019 (row 192)	Not serious	Tampa Bay or St. Joseph Sound/Clearwater Harbor
2019 (row 194)	Serious injury	Laguna Madre
2019 (row 27, vessel strike tab)	Serious injury	Pine Island Sound/Charlotte Harbor/ Gasparilla Sound/Lemon Bay

Interactions between common bottlenose dolphins and research-fishery gear are also known to occur. During 2015–2019, nine dolphins were entangled in research-related gillnets—in Texas (7), Alabama (1) and Florida (1). One of the nine entanglements resulted in a mortality; five entanglements resulted in serious injuries; and three entanglements were released alive without serious injury (Maze-Foley and Garrison 2021; see Table 3). All of the interactions with research gear were included in the stranding database (NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 25 August 2020).

Table 3. Research-related takes of common bottlenose dolphins during 2015–2019, including the serious injury determination for each animal (mortality, serious injury, not a serious injury [Not serious], or could not be determined [CBD] if seriously injured) and stock to which each animal likely belonged based on location of the interaction. All of these interactions were included in the stranding database (NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 25 August 2020). Further details on injury determinations can be found in Maze-Foley and Garrison (2021).

Year	Gear Type	Determination	Stock
2015	Gillnet	Serious injury	Matagorda Bay/Tres Palacios Bay/Lavaca Bay
2016	Gillnet	Serious injury	Matagorda Bay/Tres Palacios Bay/Lavaca Bay
2016	Gillnet	Not serious	Laguna Madre
2017	Gillnet	Serious injury	Copano Bay/Aransas Bay/San Antonio Bay/ Redfish Bay/Espiritu Santo Bay
2018	Gillnet	Not serious	Sarasota Bay/Little Sarasota Bay
2019	Gillnet	Not serious	Perdido Bay
2019	Gillnet	Mortality	Nueces Bay/Corpus Christi Bay
2019	Gillnet	Serious injury	Laguna Madre
2019	Gillnet	Serious injury	Copano Bay/Aransas Bay/San Antonio Bay/ Redfish Bay/Espiritu Santo Bay

NOAA's Office of Law Enforcement has been investigating increasing numbers of reports from the northern Gulf of Mexico coast of violence against common bottlenose dolphins, including shootings using guns and bows and arrows, throwing pipe bombs and cherry bombs, and stabbings (Vail 2016). There have been several documented stabbings of BSE common bottlenose dolphins in recent years. In 2018, an animal was impaled by a metal rod resulting in mortality, and this mortality was ascribed to the Pensacola Bay/East Bay Stock. Also in 2018, an animal ascribed to the Tampa Bay Stock was documented with a puncture wound associated with fractured vertebrae and a necrotic tissue track, likely resulting in mortality. In 2019, an animal was stabbed/impaled in its head with a spear-like object while the animal was still alive, resulting in mortality. This animal was ascribed to the Pine Island Sound/Charlotte Harbor/Gasparilla Sound/Lemon Bay Stock. All three of these cases were included in the stranding database (NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 25 August 2020) and in Table 4.

Depredation of fishing catch and/or bait is a growing problem in the Gulf of Mexico and globally, and can lead to serious injury or mortality via ingestion of or entanglement in gear (e.g., Zollett and Read 2006; Read 2008; Powell and Wells 2011; Vail 2016), as well as changes to the dolphin's activity patterns, such as decreases in natural foraging (Powell and Wells 2011). It has been suggested that provisioning of wild common bottlenose dolphins may encourage depredation because provisioning conditions dolphins to approach humans and vessels, where they then may prey on bait and catches (Vail 2016). Christiansen *et al.* (2016) found that via direct and indirect food provisioning, an increasing percentage of the long-term Sarasota Bay residents were becoming conditioned to human interactions. In addition, when comparing conditioned to unconditioned dolphins, Christiansen *et al.* (2016) reported it was more likely for a conditioned dolphin to be injured by human interactions.

Illegal feeding or provisioning of wild common bottlenose dolphins has been documented in Florida, particularly near St. Andrew Bay (Panama City Beach) in the Panhandle (Samuels and Bejder 2004; Powell *et al.* 2018) and in and near Sarasota Bay (Cunningham-Smith *et al.* 2006; Powell and Wells 2011), and also in Texas near Corpus Christi (Bryant 1994). Feeding wild dolphins is defined under the MMPA as a form of 'take' because it can alter their natural behavior and increase their risk of injury or death. Nevertheless, a high rate of provisioning was observed near Panama City Beach in 1998 (Samuels and Bejder 2004) and in 2014 (Powell *et al.* 2018), and provisioning was observed frequently and predictably south of Sarasota Bay during 1990–2007 (Cunningham-Smith *et al*. 2006; Powell and Wells 2011). Provisioning of four dolphins was documented within the Tampa Bay Stock area during 2019 while the dolphins were swimming in a local canal (NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 25 August 2020). There are emerging questions regarding potential linkages between provisioning and depredation of recreational fishing gear and associated entanglement and ingestion of gear, which is increasing through much of Florida. During 2006, at least 2% of the long-term resident dolphins of Sarasota Bay died from ingestion of recreational fishing gear (Powell and Wells 2011).

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

As noted previously, common bottlenose dolphins are known to be struck by vessels (Wells and Scott 1997; Wells *et al.* 2008). During 2015–2019, 16 stranded bottlenose dolphins (of 523 total strandings) showed signs of a boat collision (NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 25 August 2020). It is possible some of the instances were post-mortem collisions. In addition to vessel collisions, the presence of vessels may also impact common bottlenose dolphin behavior in bays, sounds and estuaries. Nowacek *et al.* (2001) reported that boats pass within 100 m of each bottlenose dolphin in Sarasota Bay once every six minutes on average, leading to changes in dive patterns and group cohesion. Buckstaff (2004) noted changes in communication patterns of Sarasota Bay dolphins when boats approached. Miller *et al*. (2008) investigated the immediate responses of common bottlenose dolphins to "high-speed personal watercraft" (i.e., recreational boats) in Mississippi Sound. They found an immediate impact on dolphin behavior demonstrated by an increase in traveling behavior and dive duration, and a decrease in feeding behavior for non-traveling groups. The findings suggested that dolphins attempted to avoid high-speed personal watercraft. It is likely that repeated short-term effects will result in long-term consequences like reduced health and viability or habitat displacement of dolphins (Bejder *et al*. 2006). Further studies are needed to determine the impacts throughout the Gulf of Mexico.

As part of its annual coastal dredging program, the Army Corps of Engineers conducts sea turtle relocation trawling during hopper dredging as a protective measure for marine turtles. Historically, there have been interactions, including mortalities, documented for common bottlenose dolphins likely belonging to BSE stocks. However, no interactions with common bottlenose dolphins have been documented during the most recent five years, 2015–2019.

Historically, there have been two documented mortalities of common bottlenose dolphins during health-assessment research projects in the Gulf of Mexico, but none have occurred during the most recent five years, 2015–2019.

Some of the BSE communities were the focus of a live-capture fishery for common bottlenose dolphins which supplied dolphins to the U.S. Navy and to oceanaria and laboratories for research and public display for more than two decades (Reeves and Leatherwood 1984; Scott 1990). Between 1973 and 1988, 533 common bottlenose dolphins were removed from Southeastern U.S. waters (Scott 1990). The impact of these removals on the stocks is unknown. In 1989, the Alliance of Marine Mammal Parks and Aquariums declared a self-imposed moratorium on the capture of common bottlenose dolphins in the Gulf of Mexico (Corkeron 2009).

Strandings

During 2015–2019, 527 common bottlenose dolphins were found stranded within bays, sounds and estuaries of the northern Gulf of Mexico (Table 4; NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 25 August 2020). There was evidence of human interaction (HI) for 102 of the strandings. No evidence of human interaction was detected for 25 strandings, and for the remaining 400 strandings, it could not be determined if there was evidence of human interaction. Human interactions were from numerous sources, including entanglements with hook and line gear, trap/pot gear, commercial shrimp trawl gear, research gillnet gear, stabbings/impalements, an entrapment between oil booms, and animals with evidence of a vessel strike (see Table 4). Strandings with evidence of fishery-related interactions are reported above in the respective gear sections. It should be noted that evidence of human interaction does not necessarily mean the interaction caused the animal's stranding or death.

There are a number of difficulties associated with the interpretation of stranding data. Except in rare cases, such as Sarasota Bay, Florida, where residency can be determined, it is possible that some or all of the stranded dolphins may have been from a nearby coastal stock. However, the proportion of stranded dolphins belonging to another stock cannot be determined because of the difficulty of determining from where the stranded carcasses originated. Stranding data underestimate the extent of human and fishery-related mortality and serious injury because not all of the dolphins that die or are seriously injured in human interactions wash ashore, or, if they do, they are not all recovered (Peltier *et al.* 2012; Wells *et al.* 2015; Carretta *et al.* 2016). Additionally, not all carcasses will show evidence of human interaction, entanglement, or other fishery-related interaction due to decomposition, scavenger damage, etc. (Byrd *et al.* 2014). Finally, the level of technical expertise among stranding network personnel varies widely as does the ability to recognize signs of human interaction.

Since 1990, there have been 15 common bottlenose dolphin die-offs or Unusual Mortality Events (UMEs) in the northern Gulf of Mexico (Litz *et al.* 2014; http://www.nmfs.noaa.gov/pr/health/mmume/events.html, accessed 5 November 2020).

1) From January through May 1990, 344 common bottlenose dolphins stranded in the northern Gulf of Mexico. Overall this represented a two-fold increase in the prior maximum recorded number of strandings for the same period, but in some locations (i.e., Alabama) strandings were 10 times the average number. The cause of the 1990 mortality event could not be determined (Hansen 1992), however, morbillivirus may have contributed to this event (Litz *et al.* 2014).

2) A UME was declared for Sarasota Bay, Florida, in 1991 involving 31 common bottlenose dolphins. The cause was not determined, but it is believed biotoxins may have contributed to this event (Litz *et al.* 2014).

3) In March and April 1992, 119 common bottlenose dolphins stranded in Texas - about nine times the average number. The cause of this event was not determined, but low salinity due to record rainfall combined with pesticide runoff and exposure to morbillivirus were suggested as potential contributing factors (Duignan *et al.* 1996; Colbert *et al.* 1999; Litz *et al.* 2014).

4) In 1993–1994 a UME of common bottlenose dolphins caused by morbillivirus started in the Florida Panhandle and spread west with most of the mortalities occurring in Texas (Lipscomb *et al.* 1994; Litz *et al.* 2014). From February through April 1994, 236 common bottlenose dolphins were found dead on Texas beaches, of which 67 occurred in a single 10-day period.

5) In 1996 a UME was declared for common bottlenose dolphins in Mississippi when 31 common bottlenose dolphins stranded during November and December. The cause was not determined, but a *Karenia brevis* (red tide) harmful algal bloom was suspected to be responsible (Litz *et al.* 2014).

6) Between August 1999 and May 2000, 150 common bottlenose dolphins died coincident with *K. brevis* blooms and fish kills in the Florida Panhandle (additional strandings included three Atlantic spotted dolphins, *Stenella frontalis*, one Risso's dolphin, *Grampus griseus*, two Blainville's beaked whales, *Mesoplodon densirostris*, and four unidentified dolphins. Brevetoxin was determined to be the cause of this event (Twiner *et al.* 2012; Litz *et al.* 2014).

7) In March and April 2004, in another Florida Panhandle UME attributed to *K. brevis* blooms, 105 common bottlenose dolphins and two unidentified dolphins stranded dead (Litz *et al.* 2014). Although there was no indication of a *K. brevis* bloom at the time, high levels of brevetoxin were found in the stomach contents of the stranded dolphins (Flewelling *et al.* 2005; Twiner *et al.* 2012).

8) In 2005, a particularly destructive red tide (*K. brevis*) bloom occurred off central west Florida. Manatee, sea turtle, bird and fish mortalities were reported in the area in early 2005 and a manatee UME had been declared. Dolphin mortalities began to rise above the historical averages by late July 2005, continued to increase through October 2005, and were then declared to be part of a multi-species UME. The multi-species UME extended into 2006, and ended in November 2006. In total, 190 dolphins were involved, primarily common bottlenose dolphins (plus strandings of one Atlantic spotted dolphin and 23 unidentified dolphins). The evidence suggests a red tide bloom contributed to the cause of this event (Litz *et al.* 2014).

9) A separate UME was declared in the Florida Panhandle after elevated numbers of dolphin strandings occurred in association with a *K. brevis* bloom in September 2005. Dolphin strandings remained elevated through the spring of 2006 and brevetoxin was again detected in the tissues of most of the stranded dolphins and determined to be the cause of the event (Twiner *et al.* 2012; Litz *et al.* 2014). Between September 2005 and April 2006 when the event was officially declared over, a total of 88 common bottlenose dolphin strandings occurred (plus strandings of five unidentified dolphins).

10) During February and March of 2007 an event was declared for northeast Texas and western Louisiana involving 64 common bottlenose dolphins and two unidentified dolphins. Decomposition prevented conclusive analyses on most carcasses (Litz *et al.* 2014).

11) During February and March of 2008 an additional event was declared in Texas involving 111 common bottlenose dolphin strandings (plus strandings of one unidentified dolphin and one melon-headed whale, *Peponocephala electra*). Most of the animals recovered were in a decomposed state. A direct cause could not be identified. However, there were numerous, co-occurring harmful algal bloom toxins detected during the time period of this UME which may have contributed to the mortalities (Fire *et al.* 2011).

12) A UME was declared for cetaceans in the northern Gulf of Mexico beginning 1 February 2010 and ending 31 July 2014 (Litz et al. 2014; http://www.nmfs.noaa.gov/pr/health/mmume/cetacean_gulfofmexico.htm, accessed 1 June 2016). The UME began a few months prior to the DWH oil spill, however most of the strandings prior to May 2010 were in Lake Pontchartrain, Louisiana, and western Mississippi and were likely a result of low salinity and cold temperatures (Venn-Watson *et al.* 2015a). The largest increase in strandings (compared to historical data) occurred after May 2010 following the DWH spill, and strandings were focused in areas exposed to DWH oil. Investigations to date have determined that the DWH oil spill is the primary underlying cause of the elevated stranding numbers in the northern Gulf of Mexico after the spill (e.g., Schwacke *et al.* 2014; Venn-Watson *et al.* 2015b; Colegrove *et al.* 2016; DWH NRDAT 2016; see Habitat Issues section).

13) A UME occurred from November 2011 to March 2012 across five Texas counties and included 126 common bottlenose dolphin strandings. The strandings were coincident with a harmful algal bloom of *K. brevis*, but researchers have not determined that was the cause of the event. During 2011, six animals from BSE stocks were considered to be part of the UME; during 2012, 24 animals.

14) A bottlenose dolphin UME occurred in southwest Florida from 1 July 2018 through 30 June 2019, with peak strandings occurring between 1 July 2018 and 30 April 2019. A total of 183 dolphins were reported (note the dates and numbers are subject to change as the closure package has not yet been approved by the UME Working Group). All age classes of dolphins were represented and the majority of the animals recovered were in moderate to advanced stages of decomposition. The cause of the bottlenose dolphin UME was determined to be due to biotoxin exposure

from the *K. brevis* harmful algal bloom off the coast of southwest Florida. The additional supporting evidence of fish kills and other species die-offs linked to brevetoxin during the same time and space support that the impacts of the harmful algal bloom caused the dolphin mortalities (Rycyk *et al.* 2020).

15) During 1 February 2019 to 30 November 2019, a UME was declared for the area from the eastern border of Taylor County, Florida, west through Alabama, Mississippi, and Louisiana (http://www.nmfs.noaa.gov/pr/health/mmume/cetacean_gulfofmexico.htm, accessed 5 November 2020). A total of 337 common bottlenose dolphins stranded during this event. The largest number of mortalities occurred in eastern Louisiana and Mississippi. An investigation concluded the event was caused by exposure to low salinity waters as a result of extreme freshwater discharge from rivers. The unprecedented amount of freshwater discharge during 2019 (e.g., Gasparini and Yuill 2020) resulted in low salinity levels across the region.

Table 4. Common bottlenose dolphin strandings occurring in bays, sounds, and estuaries in the northern Gulf of Mexico from 2015 to 2019, as well as number of strandings for which evidence of human interaction (HI) was detected and number of strandings for which it could not be determined (CBD) if there was evidence of HI. Data are from the NOAA National Marine Mammal Health and Stranding Response Database (unpublished data, accessed 25 August 2020). Please note HI does not necessarily mean the interaction caused the animal's death. Please also note that this table does not include strandings from West Bay, Galveston Bay/East Bay/Trinity Bay, Terrebonne-Timbalier Bay Estuarine System, Barataria Bay Estuarine System, Mississippi Sound/Lake Borgne/Bay Boudreau, Choctawhatchee Bay, St. Andrew Bay, and St. Joseph Bay.

Category	2015	2016	2017	2018	2019	Total
Total Stranded	68	87	91	115	166	527
HI--Yes	12[a]	23[b]	18[c]	16[d]	33[e]	102
HI--No	1	3	7	8	6	25
HI--CBD	55	61	66	91	127	400

a. Includes 1 entanglement interaction with hook and line gear (mortality); 1 entanglement interaction in commercial blue crab trap/pot gear (mortality); 1 entanglement interaction with unidentified trap/pot gear (released alive, could not be determined if seriously injured or not); 1 entanglement interaction with research gillnet gear (released alive, seriously injured); 1 live release without serious injury following entrapment between oil booms (animal was initially seriously injured, but due to mitigation efforts, was released without serious injury); and 3 animals with evidence of a vessel strike (2 mortalities, 1 live animal without serious injury).

b. Includes 6 entanglement interactions with hook and line gear (3 mortalities [1 also had evidence of a vessel strike and 1 had evidence of entanglement with shrimp trawl gear] and 3 released alive seriously injured); 6 mortalities with evidence of a vessel strike (1 was also an entanglement interaction with hook and line gear); 1 entanglement interaction with trap/pot gear (released alive, seriously injured); 1 entanglement interaction with commercial stone crab trap/pot gear (live animal without serious injury); 1 entanglement interaction with research gillnet gear (released alive, seriously injured); and 1 entanglement interaction with shrimp trawl gear (mortality, also an interaction with hook and line gear); and 1 animal with markings indicative of interaction with gillnet gear (mortality).

c. Includes 3 entanglement interactions with hook and line gear (mortalities), 1 entanglement interaction with commercial blue crab trap/pot gear (mortality); 1 entanglement interaction with trap/pot gear (released alive, seriously injured); 1 entanglement interaction with research gillnet gear (released alive, seriously injured); and 4 animals with evidence of a vessel strike (mortalities).

d. Includes 5 entanglement interactions with hook and line gear (3 mortalities and 2 animals initially seriously injured, but due to mitigation efforts, were released alive without serious injury); 2 stabbings (mortalities); 1 animal with markings indicative of interaction with gillnet gear (mortality); and 1 entanglement in possible gillnet gear (live animal without serious injury)

e. Includes 4 entanglement interactions with hook and line gear (3 mortalities and 1 animal initially seriously injured, but due to mitigation efforts, was released alive without serious injury); 1 stabbing (mortality); 3 animals with evidence of a vessel strike (mortalities); 1 entanglement interaction with commercial blue crab trap/pot gear (animal was initially seriously injured, but due to mitigation efforts, was released without serious injury); 1 entanglement interaction with crab trap/pot gear (mortality); 1 entanglement interaction with commercial spiny lobster trap/pot gear (seriously injured); 1 animal with markings indicative of interaction with gillnet gear (mortality); 4 entanglement interactions with research gillnet gear (1 mortality and 3 live animals, 2 of which were seriously injured and 1 without serious injury); and 1 interaction with unidentified gillnet gear (mortality).

HABITAT ISSUES

Issues Related to the *Deepwater Horizon* (DWH) Oil Spill

The DWH MC252 drilling platform, located approximately 80 km southeast of the Mississippi River Delta in waters about 1500 m deep, exploded on 20 April 2010. The rig sank, and over 87 days up to ~3.2 million barrels of oil were discharged from the wellhead until it was capped on 15 July 2010 (DWH NRDAT 2016). A substantial number of beaches and wetlands along the Louisiana coast experienced heavy or moderate oiling (OSAT-2 2011;

Michel *et al*. 2013). The heaviest oiling in Louisiana occurred west of the Mississippi River on the Mississippi Delta and in Barataria and Terrebonne Bays, and to the east of the river on the Chandeleur Islands. Some heavy to moderate oiling occurred on Alabama and Florida beaches, with the heaviest stretch occurring from Dauphin Island, Alabama, to Gulf Breeze, Florida. Light to trace oil was reported along the majority of Mississippi's mainland coast, from Gulf Breeze to Panama City, Florida, and outside of Atchafalaya and Vermilion Bays in western Louisiana. Heavy to light oiling occurred on Mississippi's barrier islands (Michel *et al*. 2013).

Shortly after the oil spill, the Natural Resource Damage Assessment (NRDA) process was initiated under the Oil Pollution Act of 1990. A variety of NRDA research studies were conducted to determine potential impacts of the spill on marine mammals. These studies estimated that for the Mississippi River Delta Stock of common bottlenose dolphins, 46% (95%CI: 21–65) of females suffered from reproductive failure, and 37% (95%CI: 14–57) suffered adverse health effects (DWH MMIQT 2015). A population model estimated that the stock experienced a 71% maximum reduction in population size (see Other Mortality section above). For the Mobile Bay Stock of common bottlenose dolphins, 46% (95%CI: 21–65) of females suffered from reproductive failure, and 24% (95%CI: 0–48) suffered adverse health effects (DWH MMIQT 2015). The population model estimated that the stock experienced a 31% maximum reduction in population size (see Other Mortality section above).

Stranding rates in the northern Gulf of Mexico rose significantly in the years of and following the DWH oil spill to levels higher than previously recorded (Litz *et al*. 2014; Venn-Watson *et al*. 2015b) and a UME was declared for cetaceans in the northern Gulf of Mexico beginning 1 March 2010 and ending 31 July 2014 (Litz *et al*. 2014; http://www.nmfs.noaa.gov/pr/health/mmume/cetacean_gulfofmexico.htm, accessed 1 June 2016). The primary cause for the UME was attributed to exposure to the DWH oil spill (Venn-Watson *et al.* 2015a; Colegrove *et al*. 2016; DWH NRDAT 2016) as other possible causes (e.g., morbillivirus infection, brucellosis, and biotoxins) were ruled out (Venn-Watson *et al*. 2015a). Balmer *et al*. (2015) indicated it is unlikely that persistent organic pollutants (POPs) significantly contributed to the unusually high stranding rates following the DWH oil spill. POP concentrations in dolphins sampled between 2010 and 2012 at six northern Gulf sites that experienced DWH oiling were comparable to or lower than those previously measured by Kucklick *et al*. (2011) from southeastern U.S. sites; however, the authors cautioned that potential synergistic effects of oil exposure and POPs should be considered as the extra stress from oil exposure added to the background POP levels could have intensified toxicological effects.

The DWH NRDA Trustees quantified injuries to four BSE stocks of common bottlenose dolphins, including two stocks included in this report, the Mississippi River Delta Stock and the Mobile Bay/Bonsecour Bay Stock, as well two stocks that have their own SARs (Barataria Bay Estuarine System Stock and Mississippi Sound/Lake Borgne/Bay Bourdreau Stock). A suite of research efforts indicated the DWH oil spill negatively affected these stocks of common bottlenose dolphins (Schwacke *et al*. 2014; Venn-Watson *et al*. 2015a; Colegrove *et al*. 2016). Capture-release health assessments and analysis of stranded dolphins during the oil spill both found evidence of moderate to severe lung disease and compromised adrenal function (Schwacke *et al*. 2014; Venn-Watson *et al*. 2015a). Colegrove *et al*. (2016) examined perinate strandings in Louisiana, Mississippi, and Alabama during 2010–2013 and found that common bottlenose dolphins were prone to late-term failed pregnancies and *in utero* infections, including pneumonia and brucellosis.

In the absence of any additional non-natural mortality or restoration efforts, the DWH damage assessment estimated the Mississippi River Delta Stock will take 52 years to recover to pre-spill population size, and the Mobile Bay/Bonsecour Bay Stock, 31 years (DWH MMIQT 2015).

Other Habitat Issues

The nearshore habitat occupied by many of these stocks is adjacent to areas of high human population, and in some bays, such as Mobile Bay in Alabama and Galveston Bay in Texas, is highly industrialized. Many of the enclosed bays in Texas are surrounded by agricultural lands that receive periodic pesticide applications.

Concentrations of chlorinated hydrocarbons and metals were examined in conjunction with an anomalous mortality event of common bottlenose dolphins in Texas bays in 1990 and found to be relatively low in most; however, some had concentrations at levels of possible toxicological concern (Varanasi *et al*. 1992). No studies to date have determined the amount, if any, of indirect human-induced mortality resulting from pollution or habitat degradation.

Analyses of organochlorine concentrations in the tissues of common bottlenose dolphins in Sarasota Bay, Florida, have found that the concentrations in male dolphins exceeded toxic threshold values that may result in adverse effects on health or reproductive rates (Schwacke *et al*. 2002). Studies of contaminant concentrations relative to life history parameters showed higher levels of mortality in first-born offspring, and higher contaminant concentrations in these calves and in primiparous females (Wells *et al.* 2005). While there are no direct measurements of adverse effects of pollutants on estuary dolphins, the exposure to environmental pollutants and subsequent effects on population health are areas of concern and active research.

STATUS OF STOCKS

The status of these stocks relative to optimum sustainable population is unknown and this species is not listed as threatened or endangered under the Endangered Species Act. The occurrence of 15 Unusual Mortality Events (UMEs) among common bottlenose dolphins along the northern Gulf of Mexico coast since 1990 (Litz *et al*. 2014; http://www.nmfs.noaa.gov/pr/health/mmume/events.html, accessed 5 November 2020) is cause for concern. Notably, stock areas in Louisiana, Mississippi, Alabama, and the western Florida panhandle have recently been impacted by several UMEs. However, the effects of the mortality events on stock abundance have not yet been determined, in large part because it has not been possible to assign mortalities to specific stocks and a lack of current abundance estimates for some stocks.

Human-caused mortality and serious injury for each of these stocks is unknown. Considering the evidence from stranding data (Table 4) and the low PBRs for stocks with recent abundance estimates, the total fishery-related mortality and serious injury likely exceeds 10% of the total known PBR or previous PBR, and therefore, it is probably not insignificant and not approaching the zero mortality and serious injury rate. NMFS considers each of these stocks, except for the Sabine Lake, Mississippi River Delta, and Sarasota Bay/Little Sarasota Bay stocks, to be strategic because most of the stock sizes are currently unknown, but are likely small such that relatively few mortalities and serious injuries would exceed PBR.

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