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

STOCK DEFINITION AND GEOGRAPHIC RANGE

Bottlenose dolphins are distributed throughout the bays, sounds and estuaries of the Gulf of Mexico (Mullin 1988). The identification of biologically-meaningful "stocks" of 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 provisionally identified in each of 33 areas of contiguous, enclosed or semi-enclosed bodies of water adjacent to the Gulf of Mexico (Table 1, based on descriptions of relatively discrete dolphin "communities" in some of these areas). A "community" includes resident dolphins that regularly share large portions of their ranges, exhibit similar distinct genetic profiles, and interact with each other to a much greater extent than with dolphins in adjacent waters. The term, as adapted from Wells *et al.* (1987), emphasizes geographic, genetic and social relationships of dolphins. Bottlenose dolphin communities do not constitute closed demographic populations, as individuals from adjacent communities are known to interbreed. Nevertheless, the geographic nature of these areas and long-term, multi-generational stability of residency patterns suggest that many of these communities exist as functioning units of their ecosystems, and under the Marine Mammal Protection Act must be maintained as such. Also, the stable patterns of residency observed within communities suggest that long periods would be required to repopulate the home range of a community were it eradicated or severely depleted. Thus, in the absence of information supporting management on a larger scale, it is appropriate to adopt a risk-averse approach and focus management efforts at the level of the community rather than at some larger demographic scale. Biological support for this risk-averse approach derives from several sources. Long-term (year-round, multi-year) residency by at least some individuals has been reported from nearly every site where photographic identification or tagging studies have been conducted in the Gulf of Mexico. In Texas, some of the dolphins 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) have been reported as long-term residents. Hubard *et al.* (2004) reported sightings of dolphins tagged 12-15 years previously in Mississippi Sound. In Florida, long-term residency has been reported from Choctawhatchee Bay (1989-1993), Tampa Bay (Wells 1986a; Wells *et al.* 1996a), Sarasota Bay (Irvine and Wells 1972; Irvine *et al.* 1981; Wells 1986a, 1991; Scott *et al.* 1990; Wells *et al.* 1987; Wells 2003), Lemon Bay (Wells *et al.* 1996b) and Charlotte Harbor/Pine Island Sound (Shane 1990; Wells *et al.* 1996b, 1997; Shane 2004). In Louisiana, Miller (2003) concluded the bottlenose dolphin population in the Barataria Basin was relatively closed. In many cases, residents emphasize use of the bay, sound or estuary 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).

Genetic data also support the concept of relatively discrete bay, sound and estuary 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, Matagorda Bay, Texas, dolphins appear to be a localized population, and differences in haplotype frequencies distinguish between 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). Examination of protein electrophoretic data resulted in similar conclusions for the Florida dolphins (Duffield and Wells 1986). Additionally, Sellas *et al.* (2005) examined population subdivision among Sarasota Bay, Tampa Bay, Charlotte Harbor, Matagorda Bay, and the coastal Gulf of Mexico (1-12km offshore) from just outside Tampa Bay to the south end of Lemon Bay, and found evidence of significant population structure among all areas on the basis of both mitochondrial DNA control region sequence data and 9 nuclear microsatellite loci. The Sellas *et al.* (2005) findings support the separate identification of bay, sound and estuarine communities from those occurring in adjacent Gulf coastal waters.

The long-term structure and stability of at least some of these communities is exemplified by the residents of Sarasota Bay, Florida. This community has been observed since 1970 (Irvine and Wells 1972; Scott *et al.* 1990; Wells 1991). At least 5 generations of identifiable residents currently inhabit the region, including one-third of those first identified in 1970. Maximum immigration and emigration rates of about 2-3% have been estimated (Wells and Scott 1990).

Genetic exchange occurs between resident communities; hence the application of the demographically and behaviorally-based term "community" rather than "population" (Wells 1986a; Sellas *et al.* 2005). Some of the calves in

Sarasota Bay apparently have been sired by non-residents (Duffield and Wells 2002). A variety of potential exchange mechanisms occur in the Gulf. Small numbers of inshore dolphins traveling between regions have been reported, with patterns ranging from traveling through adjacent communities (Wells 1986b; Wells *et al.* 1996a,b) to movements over distances of several hundred km in Texas waters (Gruber 1981; Lynn and Würsig 2002). In many areas year-round residents co-occur with non-resident dolphins, providing potential opportunities for genetic exchange. About 17% of group sightings involving resident Sarasota Bay dolphins include at least 1 non-resident as well (Wells *et al.* 1987). Similar mixing of inshore residents and non-residents is seen off San Luis Pass, Texas (Maze and Würsig 1999), and Pine Island Sound, Florida (Shane 2004). Non-residents exhibit a variety of patterns, ranging from apparent nomadism recorded as transience in a given area, to apparent seasonal or non-seasonal migrations. Passes, especially the mouths of the larger estuaries, serve as mixing areas. For example, several communities mix at the mouth of Tampa Bay, Florida (Wells 1986a), 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 provide additional opportunities for genetic exchange with residents, and complicate the identification of stocks in coastal and inshore waters. In small bay systems such as Sarasota Bay, Florida, and San Luis Pass, Texas, residents move into Gulf coastal waters in fall/winter, and return inshore in spring/summer (Irvine *et al.* 1981; Maze and Würsig 1999). In larger bay systems, seasonal changes in abundance suggest possible migrations, with increases in more northerly bay systems in summer, and in more southerly systems in winter. Fall/winter increases in abundance have been noted for Tampa Bay (Scott *et al.* 1989) and Charlotte Harbor/Pine Island Sound (Thompson 1981; 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).

Much uncertainty remains regarding the structure of bottlenose dolphin stocks in many of the Gulf of Mexico bays, sounds and estuaries. Given the apparent co-occurrence of resident and non-resident dolphins in these areas, and the demonstrated variations in abundance, it appears that consideration should be given to the existence of a complex of stocks, and to the roles of bays, sounds and estuaries for stocks emphasizing Gulf of Mexico coastal waters. A starting point for management strategy should be the protection of the long-term resident communities, with their multi-generational geographic, genetic, demographic and social stability. These localized units would be at greatest risk from geographically-localized impacts. Complete characterization of many of these basic units would benefit from additional photo-identification, telemetry and genetic research (Wells 1994).

The current provisional stocks follow the designations in Table 1, with a few revisions. Available information suggests that Block B35, Little Sarasota Bay, can be subsumed under Sarasota Bay, and B36, Caloosahatchee River, can be considered a part of Pine Island Sound. As more information becomes available, additional combination or division may be warranted. For example, a number of geographically and socially distinct subgroupings of dolphins in regions such as Tampa Bay, Charlotte Harbor, Pine Island Sound, Aransas Pass and Matagorda Bay have been identified, but the importance of these distinctions to stock designations remain undetermined (Shane 1977; Gruber 1981; Wells *et al.* 1996a,b, 1997; Lynn and Würsig 2002; Urian 2002).

Understanding the full complement of the stock complex using the bay, sound and estuarine waters of the Gulf of Mexico will require much additional information. The development of biologically-based criteria to better define and manage stocks in this region should integrate multiple approaches, including studies of ranging patterns, genetics, morphology, social patterns, distribution, life history, stomach contents, isozyme analyses and contaminant concentrations. Spatially-explicit population modeling could aid in evaluating the implications of community-based stock definition. As these studies provide new information on what constitutes a bottlenose dolphin "biological stock," current provisional definitions will likely need to be revised. As stocks are more clearly identified, it will be possible to conduct abundance estimates using standardized methodology across sites (thereby avoiding some of the previous problems of mixing results of aerial and boat-based surveys), identify fisheries and other human impacts relative to specific stocks and perform individual stock assessments. As recommended by the Atlantic Scientific Review Group (November 1998, Portland, Maine), an expert panel reviewed the stock structure for bottlenose dolphins in the Gulf of Mexico during a workshop in March 2000 (Hubard and Swartz 2002). The panel sought to describe the scope of risks faced by bottlenose dolphins in the Gulf of Mexico, and outline an approach by which the stock structure could most efficiently be investigated and integrated with data from previous and ongoing studies. The panel agreed that it was appropriate to use the precautionary approach and retain the stocks currently named until further studies are conducted, and made a variety of recommendations for future research (Hubard and Swartz 2002). As a result of this, efforts are being made to conduct research in new locations, such as the central Gulf, in addition to the ongoing studies in Texas and Florida.

Table 1. Most recent bottlenose dolphin abundance (N_{BEST}), coefficient of variation (CV) and minimum population estimate (N_{MIN}) in U.S. Gulf of Mexico bays, sounds and estuaries. Because they are based on data collected more than 8 years ago, most estimates are considered unknown for management purposes. Blocks refer to 33 aerial survey blocks illustrated in Figure 1. PBR - Potential Biological Removal; UNK - unknown.

Blocks	Gulf of Mexico Estuary	N_{BEST}	CV	N_{MIN}	PBR	Year	Reference
B51	Laguna Madre	80	1.57	UNK	UNK	1992	A
B52	Nueces Bay, Corpus Christi Bay	58	0.61	UNK	UNK	1992	A
B50	Compano Bay, Aransas Bay, San Antonio Bay, Redfish Bay, Espiritu Santo Bay	55	0.82	UNK	UNK	1992	A
B54	Matagorda Bay, Tres Palacios Bay, Lavaca Bay	61	0.45	UNK	UNK	1992	A
B55	West Bay	32	0.15	28	0.3	2000	E
B56	Galveston Bay, East Bay, Trinity Bay	152	0.43	UNK	UNK	1992	A
B57	Sabine Lake	0[a]	-		UNK	1992	A
B58	Calcasieu Lake	0[a]	-		UNK	1992	A
B59	Vermillion Bay, West Cote Blanche Bay, Atchafalaya Bay	0[a]	-		UNK	1992	A
B60	Terrebonne Bay, Timbalier Bay	100	0.53	UNK	UNK	1993	A
B61	Barataria Bay	138	0.08	129	1.3	2001	D
B30	Mississippi River Delta	0[1]	-		UNK	1993	A
B02-05, 29,31	Bay Boudreau, Mississippi Sound	1,401	0.13	UNK	UNK	1993	A
B06	Mobile Bay, Bonsecour Bay	122	0.34	UNK	UNK	1993	A
B07	Perdido Bay	0[a]	-		UNK	1993	A
B08	Pensacola Bay, East Bay	33	0.80	UNK	UNK	1993	A
B09	Choctawhatchee Bay	242	0.31	UNK	UNK	1993	A
B10	St. Andrew Bay	124	0.57	UNK	UNK	1993	A
B11	St. Joseph Bay	0[a]	-		UNK	1993	A
B12-13	St. Vincent Sound, Apalachicola Bay, St. Georges Sound	387	0.34	UNK	UNK	1993	A
B14-15	Apalachee Bay	491	0.39	UNK	UNK	1993	A
B16	Waccasassa Bay, Withlacoochee Bay, Crystal Bay	100	0.85	UNK	UNK	1994	A
B17	St. Joseph Sound, Clearwater Harbor	37	1.06	UNK	UNK	1994	A
B32-34	Tampa Bay	559	0.24	UNK	UNK	1994	A
B20	Sarasota Bay	97	na[c]	UNK	UNK	1992	B
B35	Little Sarasota Bay	2[b]	0.24	UNK	UNK	1985	C
B21	Lemon Bay	0[a]	-		UNK	1994	A
B22-23	Pine Sound, Charlotte Harbor, Gasparilla Sound	209	0.38	UNK	UNK	1994	A
B36	Caloosahatchee River	0[a,b]	-		UNK	1985	C
B24	Estero Bay	104	0.67	UNK	UNK	1994	A
B25	Chokoloskee Bay, Ten Thousand Islands, Gullivan Bay	208	0.46	UNK	UNK	1994	A
B27	Whitewater Bay	242	0.37	UNK	UNK	1994	A
B28	Florida Keys (Bahia Honda to Key West)	29	1.00	UNK	UNK	1994	A

References: A- Blaylock and Hoggard 1994; B- Wells 1992; C- Scott *et al.* 1989; D- Miller 2003; E- Irwin and Würsig 2004

Notes:

a. During earlier surveys (Scott *et al.* 1989), the range of seasonal abundances was as follows: B57, 0-2 (CV= 0.38); B58, 0-6 (0.34); B59, 0-0; B30, 0-182(0.14); B07, 0-0; B21, 0-15(0.43); and B36, 0-0.

b. Block not surveyed during surveys reported in Blaylock and Hoggard 1994.

c. No CV because NBEST was a direct count of known individuals.

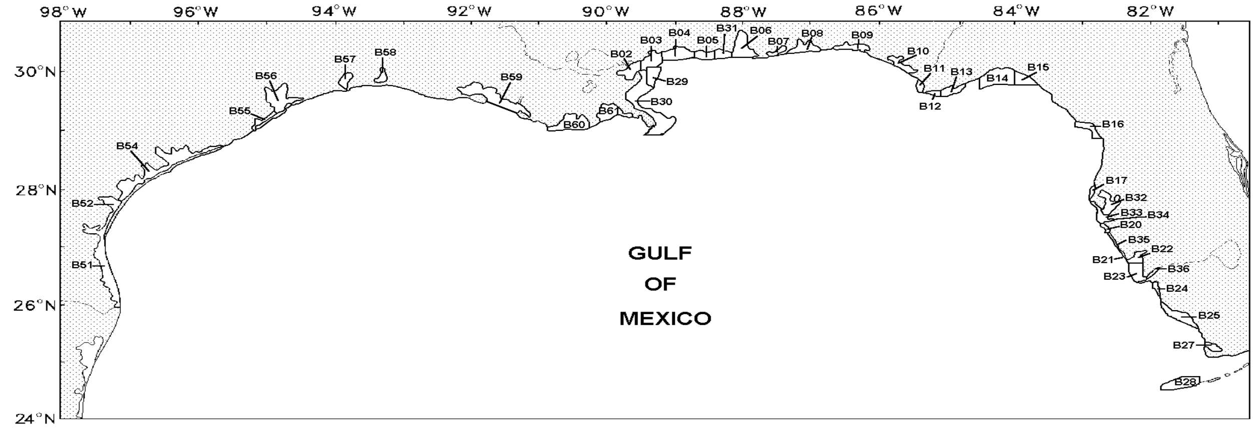

Figure 1. *U.S.A Gulf of Mexico bays and sounds. Each of the alpha-numerically designated blocks corresponds to one1 of the NMFS Southeast Fisheries Science Center logistical aerial survey areas listed in Table 1. The bottlenose dolphins inhabiting each bay and sound are considered to comprise a unique stock for purposes of this assessment.*

POPULATION SIZE

Population size estimates for most of the stocks are greater than 8 years old and therefore the current population size for each stock is considered unknown (Wade and Angliss 1997). Recent mark-recapture population size estimates are available for West Bay, Texas, and Barataria Bay, Louisiana (Table 1). Previous population size (Table 1) was estimated from preliminary analyses of line-transect data collected during aerial surveys conducted in September-October 1992 in Texas and Louisiana; in September-October 1993 in Louisiana, Mississippi, Alabama and the Florida panhandle (Blaylock and Hoggard 1994); and in September-November 1994 along the west coast of Florida (NMFS unpublished data). Standard line-transect perpendicular sighting distance analytical methods (Buckland *et al.* 1993) and the computer program DISTANCE (Laake *et al.* 1993) were used. Stock size in Sarasota Bay, Florida, was obtained through direct count of known individuals (Wells 1992). Analyses are currently underway that should provide updated abundance estimates for Sarasota Bay, Lemon Bay, Gasparilla Sound, Charlotte Harbor, Pine Island Sound, and St. Joseph Bay during 2007 (R.S. Wells, pers. comm.).

Minimum Population Estimate

The population size for all but 2 stocks is currently unknown and the minimum population estimates are given for those 2 stocks in Table 1. The minimum population estimate is the lower limit of the two-tailed 60% confidence interval of the log-normally distributed abundance estimate. This is equivalent to the 20th percentile of the log-normal distribution as specified by Wade and Angliss (1997). The minimum population estimate was calculated for each block from the estimated population size and its associated coefficient of variation. Where the population size resulted from a direct count of known individuals, the minimum population size was identical to the estimated population size.

Current Population Trend

The data are insufficient to determine population trends for all of the Gulf of Mexico bay, sound and estuary bottlenose dolphin communities. Six anomalous mortality events have occurred among portions of these dolphin communities between 1990 and 2004; however, it is not possible to accurately partition the mortalities between bay and coastal stocks, thus the impact of these mortality events on communities is not known.

For Barataria Bay, Louisiana, Miller (2003) estimated a population size ranging from 138 to 238 bottlenose dolphins (95% CI = 128-297) using mark-recapture techniques with data collected from June 1999 to May 2002. The previous estimate for Barataria Bay from 1994, 219 dolphins, falls at the high end of this range. Irwin and Würsig (2004) estimated annual population sizes ranging from 28 to 38 dolphins during 1997-2001 for the San Luis Pass/Chocolate portion of West Bay, Texas, where the previous estimate from 1992 was 29 dolphins.

CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

Current and maximum net productivity rates are not known for the dolphin communities that comprise these stocks. While productivity rates may be estimated for individual females within communities, such estimates are confounded at the stock level due to the influx of dolphins from adjacent areas which balance losses, and the unexplained loss of some individuals which offset births and recruitment (Wells 1998). Continued monitoring and expanded survey coverage will be required to address and develop estimates of productivity for these dolphin communities. 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 unknown for most stocks because the population size estimate is more than 8 years old. 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, which accounts for endangered, depleted, and threatened stocks, or stocks of unknown status relative to optimum sustainable population (OSP), is assumed to be 0.5 because these stocks are of unknown status. PBR for those stocks with population size estimates less than 8 years old is given in Table 1.

ANNUAL HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

There are a number of difficulties associated with the interpretation of stranding data. 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 probably underestimate the extent of fishery-related mortality and serious injury because not all of the dolphins which die or are seriously injured in fishery interactions 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, and the condition of the carcass if badly decomposed can inhibit the interpretation of cause of death.

A total of 1,404 bottlenose dolphins were found stranded in the U.S. Gulf of Mexico from 2001 through 2005 (Table 2) (NMFS unpublished data). Of these, 76 showed evidence of human interactions as the cause of death (e.g., gear entanglement, mutilation, gunshot wounds). Bottlenose dolphins are known to become entangled in recreational and commercial fishing gear (Wells and Scott 1994; Wells *et al.* 1998; Gorzelany 1998) and some are struck by recreational and commercial vessels (Wells and Scott 1997). In 1998 alone, 2 resident bottlenose dolphins and an associated calf were killed by vessel strikes and a resident young-of-the-year died from entanglement in a crab-pot float line (R.S. Wells, pers. comm.).

The Gulf of Mexico menhaden fishery was observed to take 9 bottlenose dolphins (3 fatally) between 1992 and 1995 (NMFS unpublished data). During that period, there were 1,366 sets observed out of 26,097 total sets, which if extrapolated for all years suggests that as many as 172 bottlenose dolphins could have been taken in this fishery with up to 57 animals killed. Without an observer program it is not possible to obtain statistically reliable information for this fishery on the number of sets annually, the incidental take and mortality rates, and the communities from which bottlenose dolphins are being taken.

Some of the bay, sound and estuarine communities were the focus of a live-capture fishery for bottlenose dolphins which supplied dolphins to the U.S. Navy and to oceanaria for research and public display for more than 2 decades ending in 1989 (NMFS unpublished data). During the period 1972-89, 490 bottlenose dolphins, an average of 29 dolphins annually, were removed from a few locations in the Gulf of Mexico, including the Florida Keys, Charlotte Harbor, Tampa Bay, and elsewhere. Mississippi Sound sustained the highest level of removals with 202 dolphins taken from this stock during this period, representing 41% of the total and an annual average of 12 dolphins (compared to a previous PBR of 13). The annual average number of removals never exceeded previous PBR levels, but it may be biologically significant that 73% of the dolphins removed during 1982-88 were females. The impact of those removals on the stocks is unknown.

Feeding or provisioning, and swimming with wild bottlenose dolphins have been documented in Florida, particularly near Panama City Beach in the Panhandle. 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, Samuels and Bejder (2004) observed a high rate of uncontrolled provisioning near Panama City Beach in 1998, and Cunningham-Smith *et al.* (2006) have observed provisioning south of Sarasota Bay continuing since 1990. The effects of swim-with activities on dolphins and their legality under the MMPA are less clear and are currently under review. Near Panama City Beach, Samuels and Bejder (2004) concluded that dolphins were amenable to swimmers due to provisioning. There are emerging questions regarding potential linkages between provisioning and depredation of recreational fishing gear and associated entanglement and ingestion of gear, which is increasing through much of Florida. During 2006, an estimated 2% of the long-term resident dolphins of Sarasota Bay died from ingestion of recreational fishing gear (R.S. Wells, pers. comm.).

One research-related mortality occurred during November 2002 in Sarasota Bay, FL. The animal was a 35-year-old male, and it died in a health assessment research project. The histopathology report stated that drowning was the cause of death. However, the necropsy revealed that the animal was in poor condition as follows: anemic, thin (ribs evident, blubber thin and grossly lacking lipid), no food in the stomach and little evidence of recent feeding in the digestive tract, vertebral fractures with muscle atrophy, with additional conditions present. This has been the only such loss during capture/release research conducted over a 36-year period on Florida's central west coast.

A research-related mortality occurred during July 2006 in St. Joseph Bay, near Panama City, FL, during a NMFS health assessment research project to investigate a series of Unusual Mortality Events in the region.

Fishery Information

The commercial fisheries which potentially could interact with these stocks in the Gulf of Mexico are the shrimp trawl, blue crab trap/pot, stone crab trap/pot, menhaden and gillnet fisheries (Appendix I). Historically, there have been very low numbers of incidental mortality or injury in the stocks associated with the shrimp trawl fishery. Bottlenose dolphins have been reported stranded with polypropylene rope around their flukes (NMFS 1991; McFee and Brooks, Jr. 1998; NMFS unpublished data), indicating the possibility of entanglement with crab pot lines. The blue crab fishery has not been monitored by observers and there are no estimates of bottlenose dolphin mortality or serious injury for this fishery. There is no observer program data for the menhaden fishery but incidental mortality of bottlenose dolphins has been reported for this fishery (Reynolds 1985). No marine mammal mortalities associated with gillnet fisheries have been reported, but stranding data suggest that gillnet and marine mammal interaction does occur, causing mortality and serious injury. In 1995, a Florida state constitutional amendment banned gillnets and large nets from bay, sounds, estuaries and other inshore waters.

Table 2. Bottlenose dolphin strandings in the U.S. Gulf of Mexico (West Florida to Texas) from 2001 to 2005. Data are from the Southeast Marine Mammal Stranding Database (SESUS). Percent of animals with indications of human interactions were calculated based on animals which were determined as “yes” or “no” for human interactions. Animals that were “CBD” (could not be determined) were excluded from % with human interactions calculations. Please note human interaction does not necessarily mean the interaction caused the animal’s death.

STATE		2001	2002	2003	2004	2005	TOTAL
Florida							
	No. Stranded	57	82 [a]	64 [d]	162	135	500
	No. Human Interactions	2	6	7	4	4	23
	No. CBD	26	44	34	63	84	251
	% With Human Interactions	6%	16%	23%	4%	8%	9%
Alabama							
	No. Stranded	17	12	7	18	19	73
	No. Human Interactions	2	0	1	0	0	3
	No. CBD	8	9	4	18	15	54
	% With Human Interactions	22%	0%	33%	CBD	0%	16%
Mississippi							
	No. Stranded	22	21 [b]	37 [e]	27	11	118
	No. Human Interactions	0	0	0	1	0	1
	No. CBD	8	6	29	13	6	62
	% With Human Interactions	0%	0%	0%	7%	0%	2%
Louisiana							
	No. Stranded	0	2	33 [f]	26	22	83
	No. Human Interactions	-	0	0	2	1	3
	No. CBD	-	2	29	24	15	70
	% With Human Interactions	-	CBD	0%	100%	14%	23%
Texas							

	No. Stranded	116	154 [c]	154 [g]	110	96	630
	No. Human Interactions	6	15	10	12	3	46
	No. CBD	5	57	101	41	17	221
	% With Human Interactions	5%	15%	19%	17%	4%	11%
TOTAL							
	No. Stranded	212	271	295	343	283	1404
	No. Human Interactions	10	21	18	19	8	76
	No. CBD	47	118	197	159	137	658
	% With Human Interactions	6%	14%	18%	10%	5%	10%

a Florida mass stranding of 2 animals in December 2002
b Mississippi mass stranding of 2 animals in March 2002
c Texas mass strandings (2 animals in January 2002, 2 animals in March 2002)
d Florida mass stranding of 2 animals in May 2003
e Mississippi mass stranding of 2 animals in April 2003
f Louisiana mass stranding of 3 animals in July 2003
g Texas mass stranding of 5 animals in March 2003

Other Mortality

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. The area surrounding Galveston Bay, for example, has a coastal population of over 3 million people. More than 50% of all chemical products manufactured in the U.S. are produced there and 17% of the oil produced in the Gulf of Mexico is refined there (Henningsen and Würsig 1991). Many of the enclosed bays in Texas are surrounded by agricultural lands which receive periodic pesticide applications.

Concentrations of chlorinated hydrocarbons and metals were examined in conjunction with an anomalous mortality event of 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 bottlenose dolphins in Sarasota Bay, Florida, have found that the concentrations found 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 estuarine dolphins, the exposure to environmental pollutants and subsequent effects on population health is an area of concern and active research.

Since 1990, there have been 8 bottlenose dolphin die-offs in the northern Gulf of Mexico. From January through May 1990, a total of 367 bottlenose dolphins stranded in the northern Gulf of Mexico. Overall this represented a two-fold increase in the prior maximum recorded strandings for the same period, but in some locations (i.e., Alabama) strandings were 10 times the average number. The cause of the 1990 mortality event could not be determined (Hansen 1992). In March and April 1992, 111 bottlenose dolphins stranded in Texas; about 9 times the average number. Seven of 34 live-captured bottlenose dolphins (20%) in 1992 from Matagorda Bay, Texas, tested positive for previous exposure to cetacean morbillivirus, and it is possible that other estuarine resident stocks have been exposed to the morbillivirus (Duignan *et al.* 1996).

In 1992, with the enactment of the Marine Mammal Health and Stranding Response Act, the Working Group on Marine Mammal Unusual Mortality Events was created to determine when an usual mortality event (UME) is occurring, and then to direct responses to such events. Since 1992, 7 bottlenose dolphin UMEs have been declared in the Gulf of Mexico. In 1993-1994 a UME of bottlenose dolphins caused by morbillivirus started in the Florida Panhandle and spread west with most of the mortalities occurring in Texas (Lipscomb 1993; Lipscomb *et al.* 1994). In 1996 a UME was declared for bottlenose dolphins in Mississippi when 27 bottlenose dolphins stranded during November and December. The cause was not determined, but a *Karenia brevis* (red tide) bloom was suspected to be responsible. Between August 1999 and February 2000, at least 120 bottlenose dolphins died coincident with *K. brevis* blooms and fish kills in the

Florida Panhandle. In March and April 2004, in another Florida Panhandle UME possibly related to *K. brevis* blooms, 107 bottlenose dolphins stranded dead (NMFS 2004). From February through April 2004, 220 bottlenose dolphins were found dead on Texas beaches, of which 67 occurred in a single 10-day period. In 2005, a particularly destructive red tide (*K. brevis*) bloom occurred off of central west Florida. Manatee, sea turtle, bird and fish mortalities were reported in the area in early 2005 and a manatee UME had been declared. 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. From July to December 2005, a total of 79 bottlenose dolphins stranded. The multi-species UME extended into 2006, and proposed dates for UME closure are in review. Finally, a separate 2005-2006 UME was declared in the Florida panhandle after elevated numbers of dolphin strandings occurred in association with a *K. brevis* bloom. Between September 2005 and September 2006, 98 bottlenose dolphin strandings occurred (plus 1 stranding of a striped dolphin, *Stenella coeruleoalba*). In September 2006 the event was officially declared over.

STATUS OF STOCK

The status of these stocks relative to OSP is unknown and this species is not listed as threatened or endangered under the Endangered Species Act. The occurrence of 6 anomalous mortality events among bottlenose dolphins along the U.S. Gulf of Mexico coast since 1990 (NMFS unpublished data) is cause for concern; however, the effects of the mortality events on stock abundance have not yet been determined.

The relatively high number of bottlenose dolphin deaths which occurred during the mortality events since 1990 suggests that some of these stocks may be stressed. Human-caused mortality and serious injury for each of these stocks is not known, but considering the evidence from stranding data (Table 2), the total human-caused mortality and serious injury exceeds 10% of the total known PBR or previous PBR, and, therefore, it is probably not insignificant and approaching the zero mortality and serious injury rate. Because these stocks are small, PBR would be exceeded with only a small number of mortalities or serious injuries. Therefore, NMFS considers that each of these stocks is a strategic stock.

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