# Models and Analyses for the Quantification of Injury to Gulf of Mexico Cetaceans from the Deepwater Horizon Oil Spill

*DWH NRDA Marine Mammal Technical Working Group Report*

***Prepared by: the Deepwater Horizon Marine Mammal Injury Quantification Team (DWH MMIQT)***

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# Contents

## Glossary of Terms

BB Barataria Bay

BSE Bay, Sound and Estuary

DWH *Deepwater Horizon*

LCY Lost Cetacean Years

MMPA Marine Mammal Protection Act

MSS Mississippi Sound

NRDA Natural Resources Damage Assessment

UME Unusual Mortality Event

YTR Years to Recovery

## 1. Introduction

The *Deepwater Horizon* (DWH) oil spill resulted in increased mortality, increased reproductive failure, and a constellation of adverse health effects, including lung disease, adrenal disease and poor body condition, in Gulf of Mexico cetaceans. Given the severity of the observed injuries and the wide distribution of cetaceans throughout the DWH oil spill footprint (Figure 1), the Trustees quantified the degree, and spatial and temporal extent, of injuries to cetaceans across the northern Gulf of Mexico (OPA § 990.52). Here we describe the specific methods used for the quantification of injuries for bay, sound and estuary (BSE) and coastal bottlenose dolphins, continental shelf dolphins, and for multiple species of offshore dolphins and whales.

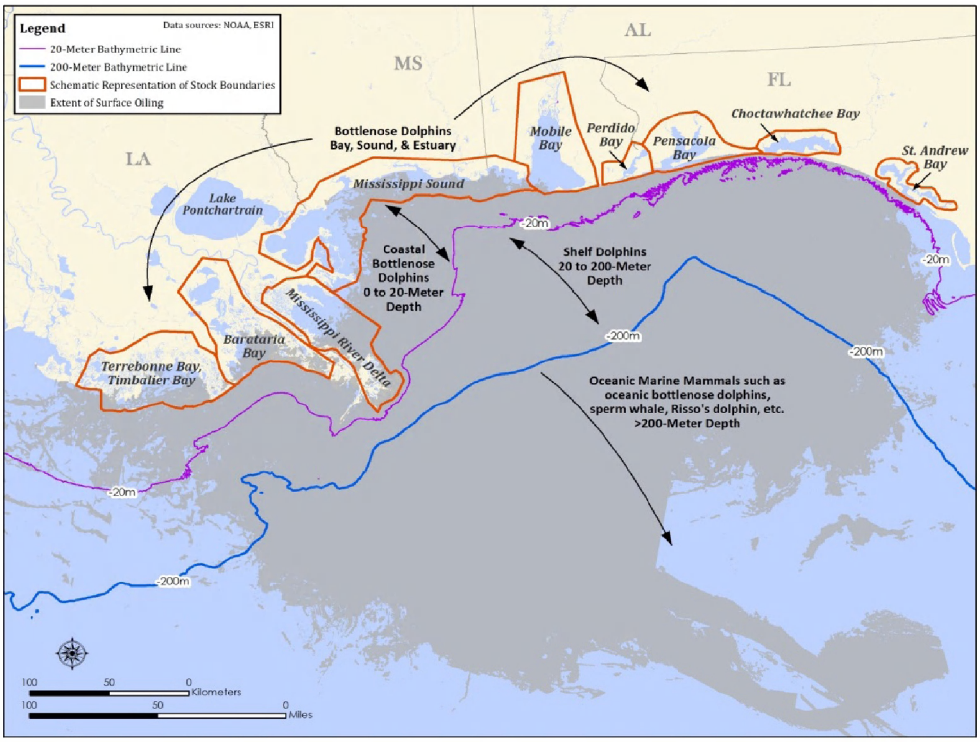



**Figure 1. Gulf of Mexico cetacean stocks within the oiling footprint from the DWH oil spill, including 9 bay, sound and estuary stocks of common bottlenose dolphins, 2 coastal, 2 continental shelf, and 18 oceanic stocks.**

## 2. Quantification of Injury to Bay, Sound, and Estuary and Coastal Common Bottlenose Dolphins

Injuries to multiple stocks of common bottlenose dolphin (*Tursiops truncatus*) in bays, sounds and estuaries (BSE) and in coastal waters of the northern Gulf of Mexico were quantified using a suite of models and analytical approaches tailored to the available data for each stock. Ultimately, the models contributed input to an integrative population model that quantified the entire scope of the DWH injury to each stock by predicting the likely population trajectory of the given stock, accounting for the documented reduced survival, reduced reproductive success, and persisting adverse health effects (Figure 2).

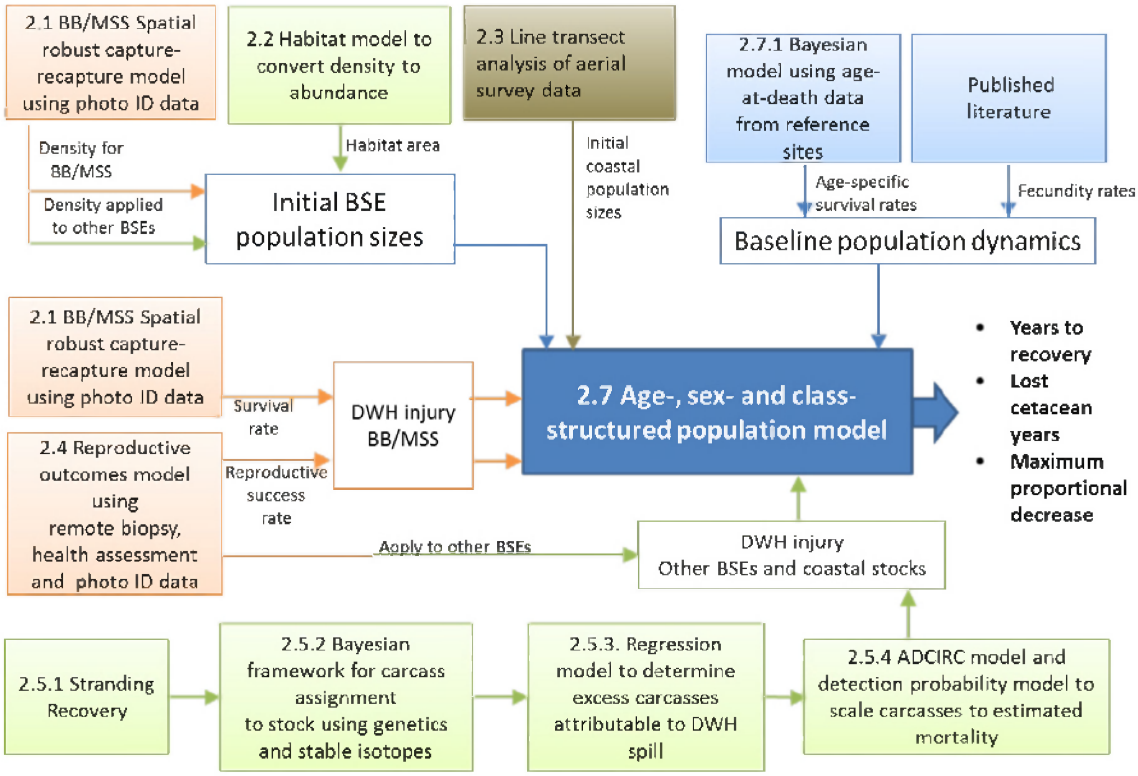



**Figure 2. Roadmap for quantification of injuries for BSE and coastal dolphin stocks following the DWH oil spill. Numbers in each box correspond with section numbers in this report that describe the methods and results for that component of the quantification. BB = Barataria Bay; MSS = Mississippi Sound; ADCIRC = ADvanced CIRCulation. Shading indicates the stocks that contributed data for the given analysis: orange/peach - BB and MSS; green – BB,MS and other BSE sites; olive green – coastal stocks, light blue – baseline data from reference stocks that were not exposed to DWH oil both within and outside of Gulf of Mexico.**

The greatest amount of information was available for dolphins in Barataria Bay and Mississippi Sound[1] as intensive NRDA studies were focused in these areas following the DWH spill. Photo-identification (-id) surveys were initiated soon after the spill occurred, and were continued periodically over the several years following the spill; surveys continued through 2012 in Mississippi Sound and through 2014 in Barataria Bay. These longer-term surveys allowed for evaluation of survival rates (Section 2.1) and tracking of individual female dolphins to determine reproductive success rates (Section 2.4). In addition, studies involving the temporary capture of dolphins to conduct in-depth evaluation of health effects were conducted in both Barataria Bay and Mississippi Sound. Less information was available for other

[1] Throughout the document the term "Mississippi Sound" refers specifically to the body of water known as Mississippi Sound, and primarily refers to eastern Mississippi Sound where most NRDA studies were performed. "Mississippi Sound Stock" refers to common bottlenose dolphins that are distributed across a broader geographic range that includes Mississippi Sound, Bay Bourdreau, and Lake Borgne, as this is the Mississippi Sound Stock range delineated by NMFS in its Stock Assessment Reports (Waring et al. 2013).

northern Gulf of Mexico estuarine and coastal stocks present within the oil footprint, and injuries for these stocks were therefore quantified by scaling the number of recovered carcasses (Section 2.5), as well as extrapolation from reproductive success rates and the prevalence of adverse health effects observed from the focused studies in Barataria Bay and Mississippi Sound.

Estimates of pre-spill stock sizes were also required. NRDA aerial surveys (Section 2.3) and boat-based photo-id studies (Section 2.1) provided data to estimate densities of coastal stocks and Barataria Bay and Mississippi Sound stocks, respectively. As with injury estimates, population sizes for other BSE stocks were estimated by extrapolation using the available density estimates from Barataria Bay and Mississippi Sound together with estimates of the area of available habitat in the other regions (Section 2.2).

The models and analytical approaches are described in detail in the sections that follow. The overview of models in Figure 2 provides a link between each component of the quantification and the relevant section of text. Four different types of field studies contributed the required data for the modeling efforts:

- photo-id surveys, including a robust survey design for capture-recapture analysis to obtain population size and survival estimates, and monitoring of targeted individual dolphins to document reproductive outcomes,
- remote biopsy sampling to collect blubber samples that were analyzed to determine pregnancy, and skin samples to inform stock assignment analyses for modeling carcass stranding rates,
- capture-release sampling to conduct health assessments, determine pregnancy along with expected parturition date via ultrasound, and deploy satellite tags, and
- stranding response to document mortalities and collect carcasses for cause-of-death investigation.

## 2.1. Spatial Robust-design Capture-recapture Model using Photo-id Data to Estimate Population Densities and Survival Rates for Barataria Bay and Mississippi Sound

Post-spill survival rates and densities were estimated for common bottlenose dolphin (hereafter referred to as bottlenose dolphin) stocks in Barataria Bay and Mississippi Sound by applying a spatial robust-design capture-recapture model to data collected during photo-id surveys. This section provides a summary of the field methods and modeling approach.

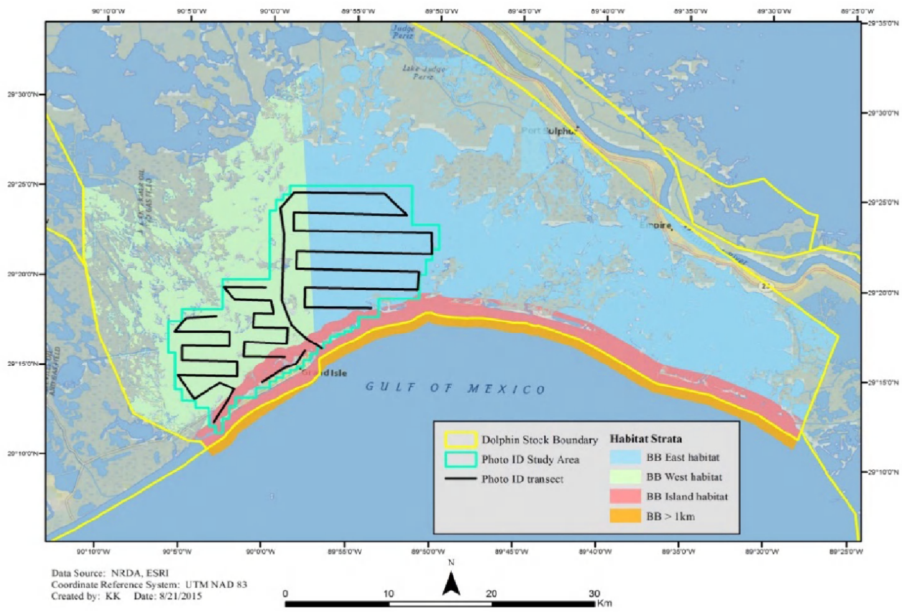
The population model described in Section 2.7 requires estimates of initial population size (i.e., number of animals), rather than density. Hence, a model of suitable dolphin habitat area was also developed (Section 2.2) and combined with strata-specific density estimates derived in this section to produce initial population sizes for Barataria Bay and Mississippi Sound stocks. Strata-specific density estimates were then extrapolated to other BSE stocks, and population sizes estimated based on application of the dolphin habitat model to those areas.

In the summer of 2010, photo-id capture-recapture surveys (Melancon et al. 2011) were initiated to gain a better understanding of bottlenose dolphin density and survival in Barataria Bay and Mississippi

Sound. In Barataria Bay, survey crews logged 838 hours of search effort along pre-defined routes (Figure 3) during 10 two-week (barring adverse weather or equipment failure) primary photo sessions between June 2010 and April 2014. In Mississippi Sound, survey crews logged 820 hours of effort during 8 two-week primary photo sessions between June 2010 and May 2012. Each two-week primary photo session consisted of three secondary photo sessions, which were separated by at least one day. Secondary sessions lasted between two and three days and encompassed a complete transit of the survey route.

All photo-id surveys were conducted by 3-4 researchers in small boats under optimal sighting conditions (Beaufort Sea State < 3) traveling along designated survey routes (transects) at speeds of 28-30km/hr. When a dolphin or a group of dolphins was sighted, the boat approached the group and attempted to obtain a photograph of the dorsal fin for all dolphins in the group (Speakman et al. 2010). Photos were sorted, graded for quality, and matches were made to the existing photo-catalog based on unique characteristics of individual fins using the software FinBase (version 2.0) (Adams et al. 2006, Melancon et al. 2011). All fin matches were independently verified by another researcher.

A spatial robust-design capture-recapture model similar to the one proposed by Ergon and Gardner (2013) was applied separately to the photo-id data collected in Barataria Bay and Mississippi Sound. The spatial robust model contained spatially explicit capture-recapture (SECR) models (Borchers and Efford 2008, Efford et al. 2009) estimated from data collected during the secondary occasions within each primary. The SECR model estimated a declining detection function for dolphin groups as distance from the survey trackline increased. The SECR model estimated density by considering the likelihood of obtaining a photograph at a particular distance from a dolphin's activity center, where the activity center was a theoretical construct loosely defined as the geographic center of a dolphin's home range. The model estimated the likelihood of a photograph by assuming locations were bivariate random variables that followed a (circular) bivariate normal distribution and whose means were centered on the activity center. Under this assumption, activity center locations could be estimated from observed locations and the assumed (normal) distribution when a dolphin was not sighted. Otherwise, if a dolphin was sighted every secondary occasion, the geographic center of observed locations was a reasonable estimate of the dolphin's activity center. Density was then estimated as the number of estimated activity centers in an area divided by the size of the area over which a dolphin was likely to be photographed if it was present, which is akin to dividing estimated abundance by the sampled area even though the sampled area does not have hard boundaries.



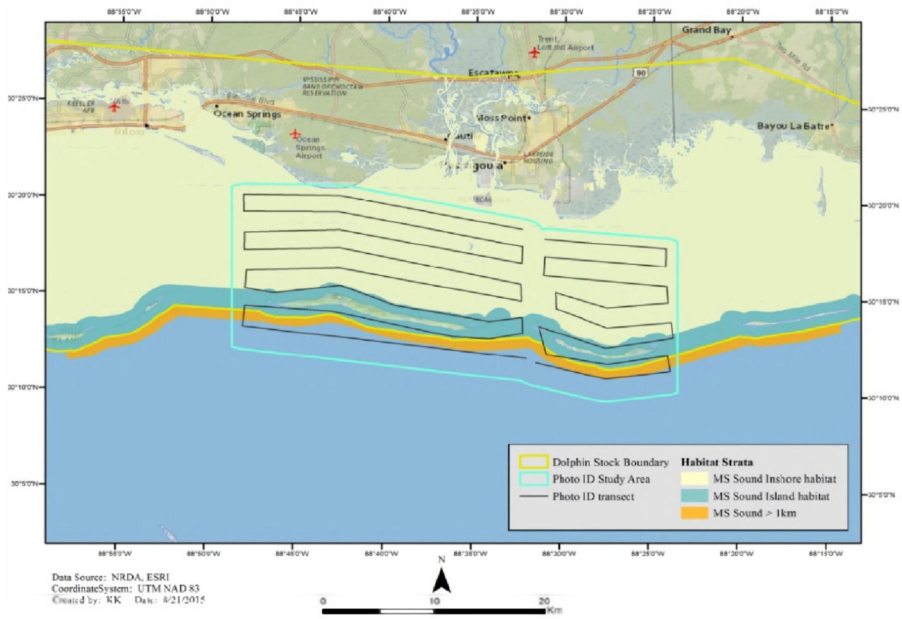




**Figure 3. Maps of Barataria Bay (top) and eastern Mississippi Sound (bottom) showing bottlenose dolphin photo-id transects, study area, and habitat strata.**

Between primary sessions, the spatial robust model adjusted capture rates for movement of activity centers. Based on primary occasions where individual dolphins were sighted, the model could estimate average movement between estimated activity centers. This allowed the model to infer when an individual was off the study area (i.e., emigrated) and beyond the range of detection. If the animal was inferred to be on the study area but was not sighted, death was a possibility, along with the possibility that the individual was alive but uncaptured. This aspect of the model introduced individual- and time-specific heterogeneity in capture probabilities, and allowed the model to correct survival rates for temporary and permanent emigration. Consequently, survival estimates from the spatial robust-design reflect true survival (Ergon and Gardner 2013, Schaub and Royle 2013) because the ambiguity between death and permanent emigration present in other capture-recapture models was corrected via accurate estimation of capture heterogeneity. Estimation of the spatial robust model was performed using the Bayesian approach of Ergon and Gardner (2013), with additional parameters for strata (see next paragraph) and included a habitat mask.

Close examination of dolphin movements from satellite-linked tag data revealed that dolphins in Barataria Bay appear to partition their distribution across habitats and exhibit preference for 3 different geographic areas (Wells and Balmer 2012, Wells et al. 2014a). One group spent a majority of time in the area immediately surrounding Grand Isle, one of the larger barrier islands separating the bay and coastal waters. A second group spent most of their time in estuarine waters of western sections of Barataria Bay, but occasionally ventured to areas near Grand Isle. A third group spent the majority of their time in estuarine waters of eastern sections of Barataria Bay, but occasionally ventured to areas near Grand Isle. Little, if any, movement between eastern and western parts of Barataria Bay was observed. Based on these observations, three geographic strata were established in Barataria Bay both inside and outside the study area (East, West, and Island; Figure 3). Satellite-linked tag data in Mississippi Sound revealed two general geographic affinities, one for the islands and one for the bay inshore of the islands. Based on these observations, two strata (Inshore and Island) were established in Mississippi Sound (Figure 3).

The spatial robust model did not differentiate movements of activity centers in different directions. The only parameter estimated for activity center movement was distance. In Barataria Bay, estimates of activity center movement distances were based on bay-side observations, and were large relative to the width of the islands. As a result, the spatial robust model imputed small but non-zero densities >1km offshore of the islands even though no surveys were conducted in those areas. To allow inferred dolphin activity centers >1km offshore, and to separate those activity centers from others, a fourth stratum (labeled ">1km") was added to Barataria Bay; however, densities in the >1km stratum were discarded as unreliable due to a lack of survey effort in those areas.

In Mississippi Sound, observations >1km offshore of barrier islands were infrequent, and movement parameter estimation in the spatial robust model relied primarily on observations in the other two strata. These movement parameters were thought to be accurate for the Island and Inshore strata of Mississippi Sound, but not offshore areas. Similar to Barataria Bay, an ">1km" stratum was added to Mississippi Sound, but densities in that stratum were discarded as unreliable due to the lack of information on movements into and out of those areas.

In the East and West strata of Barataria Bay, and in the Inshore stratum of Mississippi Sound, dolphin density varied between approximately 0 and 2 dolphins/km$^2$ (Figure 4, Table 1), and remained relatively constant throughout the study period (June 2010 through April 2014 for Barataria Bay, July 2010 through May 2012 for Mississippi Sound). Densities in the Island strata of both bays were higher. Soon after the spill (late June 2010), density in the Island strata of Barataria Bay was low (approximately 8.2 dolphins/km$^2$) relative to subsequent estimates (Figure 4). After that, between November 2010 and April 2013, density increased to approximately 11 dolphins/km$^2$, with the exception of an extreme high estimate of 17.0/km$^2$ in June 2011 and an extreme low estimate of 6.7/km$^2$ in February 2012. Potential causes of the one-time high and low estimates are unknown. In late 2013 and early 2014, density in the Island strata of Barataria Bay moved above the previous level to an average of 13.9 dolphins/km$^2$. Density in the Island strata of Mississippi Sound varied between 3.5 and 5.7 dolphins/km$^2$ and was relatively constant throughout the study period. Approximate 95% confidence intervals on these density estimates appear in Figure 4 and Table 1.

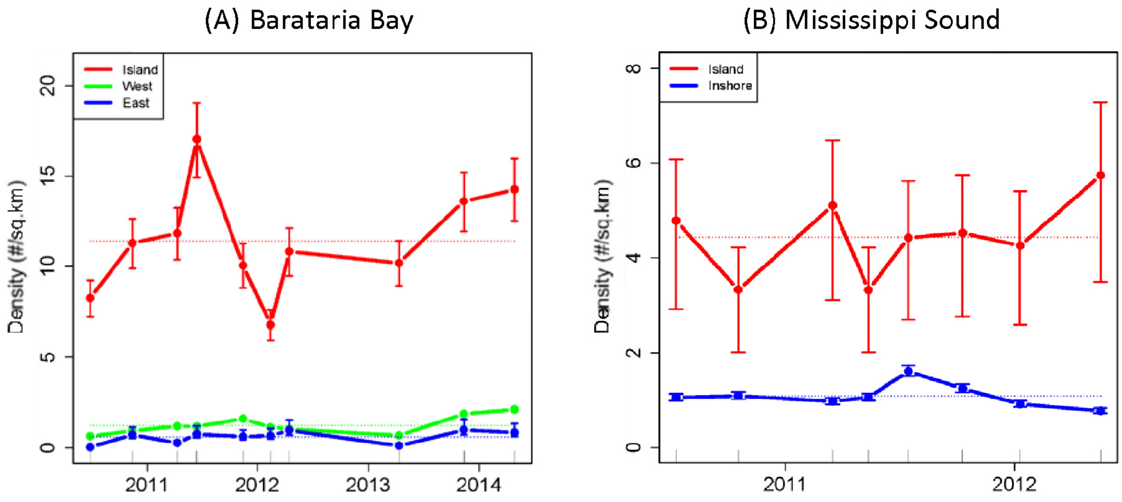



**Figure 4. Density estimates by stratum produced by the spatial robust-design capture-recapture model applied to photo-id data from (A) Barataria Bay and (B) Mississippi Sound. Vertical bars are approximate 95% confidence intervals. Ticks above x-axes are mid-point dates of primary sampling occasions. Dashed horizontal lines are average density. Note different scales among panels.**

**Table 1. Estimated dolphin density averaged over the study period, standard deviation (Std Dev) of density, and upper and lower confidence intervals in strata of Barataria Bay and Mississippi Sound.**

| Bay or Sound | Stratum | Estimated Density (#/km$^2$) | Std Dev | 95% Confidence Interval | |
|---|---|---|---|---|---|
| | | | | Lower | Upper |
| Barataria | East | 0.601 | 0.1849 | 0.239 | 0.963 |
| Barataria | West | 1.244 | 0.0319 | 1.182 | 1.306 |
| Barataria | Island | 11.401 | 0.884 | 9.668 | 13.134 |
| Mississippi | Inshore | 1.091 | 0.042 | 1.009 | 1.173 |
| Mississippi | Island | 4.432 | 1.1012 | 2.274 | 6.59 |

In the first (approximate) year following the oil spill, average annual survival in Barataria Bay was low, 0.846 (95% CI: 0.787-0.901), and decreased slightly to 0.804 (95% CI: 0.766-0.847) three years later (Table 2, Figure 5). After April 2013, during the fourth year of study, survival in Barataria Bay was estimated at 0.973 (95% CI: 0.937-0.996), or just above survival estimates reported near Charleston, SC (0.951) (Speakman et al. 2010) and in Sarasota Bay, FL (0.961) (Wells and Scott 1990). However, survival estimates between the last two occasions in a capture-recapture model are known to be unreliable due to partial or complete confounding with capture probability (Lebreton et al. 1992). Consequently, it is reasonable to assert that survival increased after April 2013, but the exact magnitude of survival during the fourth year of study is in question because survival between the last two occasions was included in the average.

In Mississippi Sound, survival during the first (approximate) year following the oil spill averaged 0.726 (95% CI: 0.673-0.784), below that in Barataria Bay where there was higher surface oiling. It is possible that Mississippi Sound dolphins were exposed to additional environmental factors (*e.g.*, cold, pathogens) that exacerbated the effects from the DWH spill, producing even higher mortality. A compromised ability to fight off infection related to the adverse health effects caused by the DWH spill, combined with a higher exposure to other stressors could have led to higher mortality in Mississippi Sound. Consistent with the estimated low survival rates, clusters of dead dolphins, stranding at rates well above baseline, were observed in both of these areas during the same time periods: 2010 and 2011 in Barataria Bay, and 2011 in Mississippi (Venn-Watson et al. 2015b).

**Table 2. Estimated dolphin survival (Pr(Survive)) and upper and lower confidence intervals (CI) in Barataria Bay and Mississippi Sound averaged over approximate 1-year periods of the study. Estimates were scaled to 12-months by raising Inter-Primary (see Figure 5) averages to the power of (1/L), where L was length of the period in years.**

| | | | | | 95% CI | |
|---|---|---|---|---|---|---|
| Bay or Sound | Period | Start Date | End Date | Pr(Survive) | Lower | Upper |
| Barataria | Interval 1 | 26-Jun-10 | 12-Jun-11 | 0.846 | 0.787 | 0.901 |
| Barataria | Interval 2 | 12-Jun-11 | 15-Apr-12 | 0.827 | 0.790 | 0.862 |
| Barataria | Interval 3 | 15-Apr-12 | 12-Apr-13 | 0.804 | 0.766 | 0.847 |
| Barataria | Interval 4 | 12-Apr-13 | 26-Apr-14 | 0.973 | 0.937 | 0.996 |
| Mississippi | Interval 1 | 10-Jul-10 | 16-Jul-11 | 0.726 | 0.673 | 0.784 |
| Mississippi | Interval 2 | 16-Jul-11 | 17-May-12 | 0.517 | 0.468 | 0.563 |

Survival for Mississippi Sound during the next 10 months, annualized to a full year, was estimated at 0.517 (95% CI: 0.468-0.563; Table 2, Figure 5). Again, due to partial confounding of survival with capture probability between the last two occasions, but also because mid-May 2012 to mid-July 2012 (a period that generally has lower stranding rates as compared to earlier Spring months, see Section 2.5.3) was not sampled, the survival estimate between summer 2011 and summer 2012 is thought to be unreliable. Dropping the final inter-primary period (10Jan12 to 17May12) and annualizing the result produces a survival estimate of 0.78 (95% CI: 0.74-0.81) for the second year in Mississippi Sound. Despite questionable reliability of the final estimate, data support the assertion that survival in Mississippi

Sound remained constant for approximately two years following the spill, and did not increase markedly during that period.

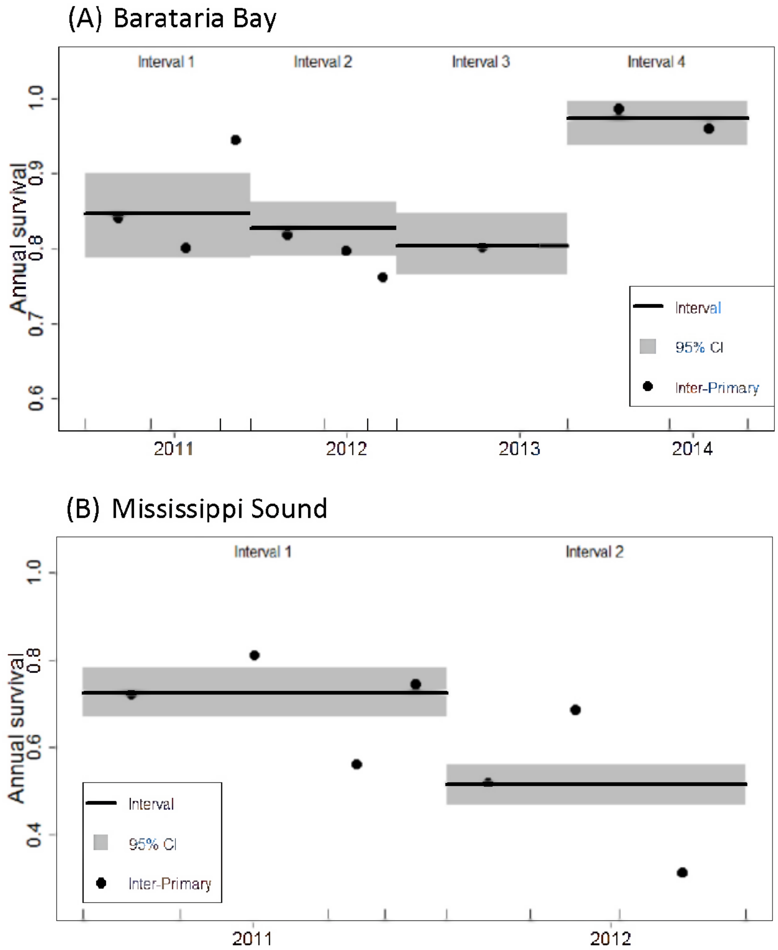



**Figure 5. Annual survival estimates in (A) Barataria Bay and (B) Mississippi Sound. Estimates labeled "Interval" (gray bars) are average survival probability during the approximate 1 year interval, scaled to 12 months. "Inter-Primary" estimates are survival between primary occasions, scaled to 12 months. Ticks above x-axes are mid-point dates of primary sampling occasions. Note different scales among panels. The final "Inter-primary" estimates in both areas are considered unreliable due to confounding (see text), and final "Interval" estimates should be used with caution.**

## 2.2. Habitat Model to Estimate Initial Population Size of Estuarine Stocks

Estimation of common bottlenose dolphin abundance involved first estimating the average extent of dolphin habitat and then multiplying by estimated or assumed density. This section contains the methods and results used to estimate the extent of dolphin habitat, as well as the resulting dolphin abundance estimates, in five BSE areas.

Bottlenose dolphins are a marine species. Though they can tolerate low salinities for short periods of time, such waters are not suitable long-term as they cause a variety of ill health effects (Wilson et al. 1999, Holyoake et al. 2010, Mullin et al. 2015). Consequently, salinity plays an important role in determining suitable dolphin habitat. To estimate suitable habitat for each BSE, salinity values estimated by a spatio-temporal kriging model were temporally matched with dolphin satellite telemetry observations to determine a threshold between higher salinity waters and those low salinity waters that dolphins tolerate for minimal amounts of time.

The spatio-temporal model (Szpiro et al. 2010, Sampson et al. 2011, Lindstrom et al. 2014, McDonald et al. 2015) estimated daily salinity (in parts per thousand, or ppt) from 2006 to 2012 at every point in a 200 by 200 meter grid overlaying each of the following five BSE areas: Terrebonne/Timbalier, Barataria Bay, Mississippi River Delta, Mississippi Sound/Bay Boudreau, and Mobile Bay. Wells and Balmer (2012) provided telemetry location from 25 dolphins outfitted with satellite tags in Barataria Bay between August 3, 2011 and April 10, 2012. Habitat was estimated as the average extent (square kilometers) of higher salinity waters in each BSE stock area (Figure 6). The threshold between the high and low salinity waters was established from two sources of information. First, dolphin telemetry locations were plotted on daily salinity surfaces estimated by the spatio-temporal kriging model to investigate dolphin movement in relation to changing salinity. Second, dolphin telemetry locations were matched to salinity estimates from the same day and to the average over a 3 X 3 grid of 200 by 200m cells closest to the location, and the first, fifth, tenth, and twenty-fifth percentiles (Table 3) of these derived salinities were considered candidates for the high-low salinity threshold.

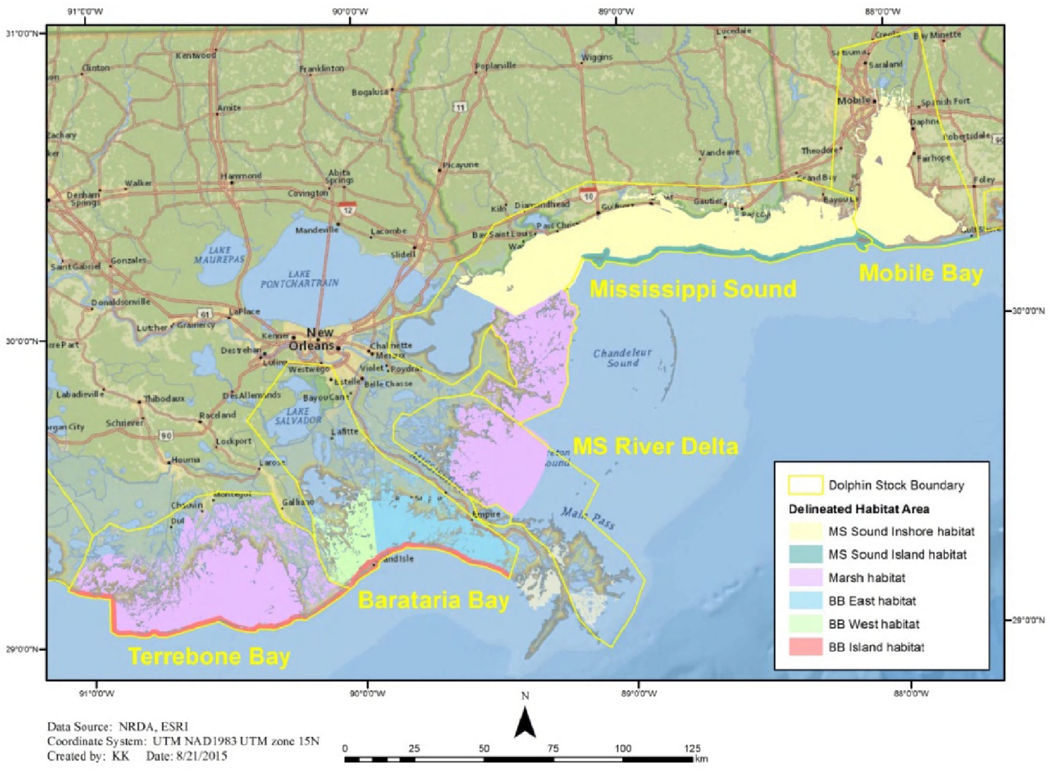



**Figure 6. Suitable habitat areas delineated into one of the following habitat strata: MS Sound Inshore, MS Sound Island, BB (Barataria Bay) East, BB West, Marsh, and BB Island habitats.**

The salinity in locations where dolphins in Barataria Bay were present (n=3,581) ranged from 1.61 to 32.08 ppt. Based on visual inspection of dolphin locations and salinity contours, nearly all dolphins that ventured into waters with salinity less than approximately 8 ppt moved to higher salinity within 24 hours. Eight ppt was close to the fifth-percentile of salinity estimated at the time and place of dolphin locations (7.89 ppt, Table 3). Little or no correlation between movements and contours of the other percentiles was observed. Consequently, the fifth-percentile, or 7.89 ppt, was chosen as the high-low salinity threshold.

**Table 3. Percentiles for salinity encountered by satellite-tagged dolphins in Barataria Bay. Salinity values were matched temporally and spatially to dolphin locations by averaging over the nine 200 by 200 meter grid cells surrounding the location.**

| Percentile | 1% | 5% | 10% | 25% |
|---|---|---|---|---|
| **Predicted Salinity (ppt)** | **4.69** | **7.89** | **10.83** | **15.26** |

Within each BSE, the final extent of habitat was defined as waters that contained, on average, salinity greater than the high-low salinity threshold. Average geographic placement of the high-low threshold was determined by first calculating a temporal average salinity for each grid cell. This established an average salinity surface, which was then clipped to a land-water interface (land defined by Research Planning, Incorporated, dated 2/7/2012). The contour associated with the high-low threshold of this average salinity surface defined placement, and hence extent, of dolphin habitat (in units of $km^2$) in each BSE.

In Barataria Bay and Mississippi Sound, these habitat areas were further subdivided into habitat strata as defined in Section 2.1 and used in the spatial robust-design model. Insufficient information on dolphin use of most of Lake Borgne was available to characterize this habitat; therefore, most of Lake Borgne was excluded from the analysis. In addition to the habitat strata previously defined, a "Marsh" habitat was defined as a mixture of attributes from BB East and BB West strata. Similar habitats were identified in the remaining BSEs (Figure 6). An average density calculated from the densities in BB East and BB West was used for the "Marsh" density. Densities were estimated from the spatial robust-design model for Barataria Bay and Mississippi Sound strata. It was assumed that densities in Mobile Bay would be comparable to densities for similar habitats in Mississippi Sound, and that densities in Terrebonne/Timbalier Bay would be comparable to densities for similar habitat in Barataria Bay. All of MS River Delta was defined as Marsh Habitat because it contains a mixture of open water and large areas of marsh like the interior of Barataria Bay. Densities in BSE's other than Barataria Bay and Mississippi Sound are referred to as "assumed densities" and their assignments are listed in Table 4 under "Habitat Strata". Final abundance estimates in each BSE were the product of strata-specific density and habitat in the strata, summed over the BSE (Table 5). Because variation in density dwarfed variation in average salinity (McDonald et al. 2015), 95% confidence intervals on final estimates of abundance were computed by considering variation in density only.

Mississippi Sound contained the most dolphin habitat, while Mississippi River Delta contained the least. In total, 8,597.42 $km^2$ were identified as dolphin habitat from Terrebonne Bay east to Mobile Bay. Total estimated abundance for all quantified BSEs was estimated to be approximately 13,000 dolphins (12,932 individuals).

**Table 4. Habitat areas, estimated or assumed densities, and estimated abundance within all strata of each dolphin BSE stock area. Estimated densities (indicated by *) for Barataria Bay and Mississippi Sound were obtained from the spatial robust-design capture-recapture model (section 2.1). For other BSEs, assumed densities (indicated by #) were calculated based on density estimates from Barataria Bay or Mississippi Sound.**

| BSE Stock Area | Habitat Strata | Habitat Area ($km^2$) | Estimated or Assumed Density ($\#/km^2$) | Population Estimate |
|---|---|---|---|---|
| Terrebonne/ Timbalier Bay | Marsh | 1879.14 | 0.923# | 1734 |
| | BB Island | 206.02 | 11.401# | 2349 |
| Barataria Bay | BB East | 684.73 | 0.601* | 412 |
| | BB West | 355.28 | 1.244* | 442 |
| | BB Island | 127.38 | 11.401* | 1452 |
| Mississippi River Delta | Marsh | 888.93 | 0.923# | 820 |
| Mississippi Sound | Marsh | 778.41 | 0.923# | 718 |
| | MS Sound Inshore | 2195.98 | 1.091* | 2396 |
| | MS Sound Island | 242.35 | 4.432* | 1074 |
| Mobile Bay | MS Sound Inshore | 1069.95 | 1.091# | 1167 |
| | MS Sound Island | 51.03 | 4.432# | 226 |

**Table 5. Abundance, standard error, and 95% confidence interval for each BSE stock.**

| BSE Stock Area | Total Habitat Area (km$^2$) | Abundance Estimate | Standard Error | 95% Confidence Interval | |
|---|---|---|---|---|---|
| | | | | Lower | Upper |
| Terrebonne/ Timbalier Bay | 2085.16 | 4083 | 253.47 | 3586 | 4580 |
| Barataria Bay | 1167.39 | 2306 | 169.81 | 1973 | 2639 |
| Mississippi River Delta | 888.93 | 820 | 83.39 | 657 | 984 |
| Mississippi Sound | 3216.74 | 4188 | 291.66 | 3617 | 4760 |
| Mobile Bay | 1120.98 | 1393 | 71.95 | 1252 | 1535 |

## 2.3. Analysis of Line Transect Survey Data to Estimate Initial Population Sizes of Coastal Bottlenose Dolphin Stocks

The Northern and Western Coastal stocks of bottlenose dolphins occupy waters of the Gulf of Mexico from the shoreline to 20m depth. Abundance estimates for these stocks were derived from seasonal aerial surveys conducted between spring 2011 and winter 2012 that covered waters from the shoreline to the continental shelf from Brownsville, TX (U.S./Mexico border) to north of the Dry Tortugas. Surveys typically were flown during favorable sighting conditions at Beaufort Sea State less than or equal to four. Trained visual observers searched for marine mammals and sea turtles from directly beneath the aircraft out to a perpendicular distance of approximately 600 meters from the trackline. The analysis framework described by Buckland et al. (2004) was used to estimate detection probability and ultimately abundance, using the software Distance (Thomas et al. 2010). Two survey teams operated independently on the aircraft, and data collected were used to model and estimate detection probability on the trackline (p(0)) using the independent-observer approach assuming point-independence as described. Sighting condition variables (sea state, glare, water color, turbidity, and light conditions) were included in models to account for their effects on detection probability. Variance estimation for the model parameters was accomplished through bootstrap resampling of tracklines. Tracklines and sightings were stratified to correspond to the stock boundaries so that stock-specific abundance and precision could be estimated (Figure 7).

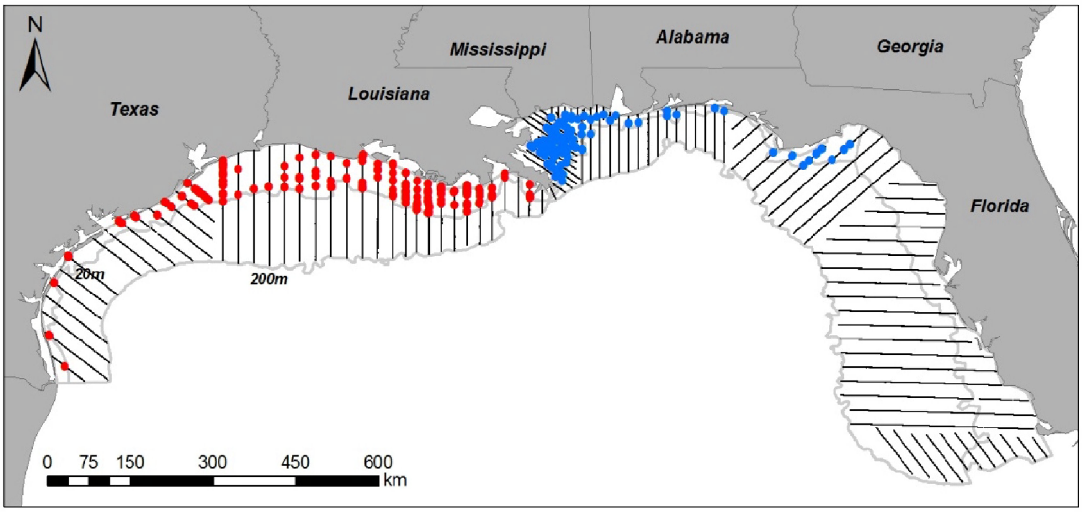



**Figure 7. Bottlenose dolphin sightings in the Western (red) and Northern (blue) Coastal stock areas (0-20m depth) during the 2011-2012 aerial surveys. Sightings from the Eastern coastal and continental shelf stocks are not shown. Stock boundaries for the Continental shelf and Eastern Coastal stocks are also shown.**

A total of 56,422 kilometers of trackline was surveyed during the spring (13 April-31 May, 2011; 13,608 km), summer (11 July – 4 September, 2011; 15,800 km), fall (12 October – 4 December, 2011; 13,971 km) and winter (13 January – 9 March, 2012; 13,043). Bottlenose dolphins were sighted throughout the survey range in estuarine, coastal, and continental shelf waters. There were 126 bottlenose dolphin groups detected within the boundaries of the northern coastal stock and 171 groups in the western stock (Figure 7). Abundance estimates for each stock were derived as weighted means of the estimates from each of the four seasonal surveys where the weighting was inversely proportional to the variance of the estimate (Table 6).

**Table 6. Abundance estimates and 95% confidence intervals for Northern and Western Coastal bottlenose dolphin stocks obtained through seasonal aerial surveys conducted between April 2011 and March 2012.**

| Stock | Stock Area ($km^2$) | Average Abundance | Standard Error | 95% Confidence Interval | |
|---|---|---|---|---|---|
| | | | | Low | High |
| Northern Coastal | 15,905 | 7,185 | 1,491 | 4,800 | 10,754 |
| Western Coastal | 42,991 | 20,161 | 3,425 | 14,482 | 28,066 |

## 2.4. Reproductive Outcomes Model and Proportion of Females with Reproductive Failure

Injury to recruitment was evaluated by assessing the reproductive success rate, *i.e.*, the rate at which pregnant females successfully produced viable calves[2], and comparing to an expected baseline. The assessment of the reproductive success rate was conducted using data collected in Barataria Bay and Mississippi Sound in the years during and after the spill (2010-2014).

Data for assessing reproductive success rate (and its complement, reproductive failure rate) came from health assessment, remote biopsy and photo-id field efforts. Pregnancy was determined using established protocols from either 1) ultrasound examinations of the reproductive tract during capture-release health assessments (Lacave et al. 2004, Smith et al. 2013), or 2) endocrine levels in blubber tissue collected from dart biopsies (Kellar et al. 2006, Kellar et al. 2013, Kellar et al. 2014). Photo-id surveys were used to track the status of pregnant females and any associated neonate calves for a minimum of one year after the initial pregnancy detection (IPD). For those pregnant females observed after IPD, individuals seen with a calf (reproductive success) and without one (reproductive failure) were recorded. Reproductive successes and failures of expectant mothers not observed during the year following IPD were not included in the calculation.

Observations were pooled across the two areas (*i.e.*, Barataria Bay and Mississippi Sound) and across years to increase sample size for a more precise calculation of reproductive-failure rate. The rate from these areas was then compared to published failure rates from reference areas not impacted by the DWH spill: Sarasota Bay, FL (Wells et al. 2014b); Indian River Lagoon, FL (Bergfelt et al. 2013); and Charleston Harbor, SC (Bergfelt et al. 2013). The rate of failure within the oil-affected stocks in excess of the rates found within the reference stocks was attributed to the effects of the spill.

The resulting estimated aggregated reproductive success for Barataria Bay (0.18, n=17) and Mississippi Sound (0.22, n=9) was 0.19 for 26 pregnant females that were tracked for at minimum 1 year after IPD. Only 1 out of every 5 detected pregnancies resulted in a viable calf. In comparison, the expected success rate based on the aggregate of previous observations in reference areas is over 3-fold higher (0.65, n=34). The reference areas were ((Sarasota Bay, FL (n=14); Indian River Lagoon, FL (n=14); and Charleston Harbor, SC (n=6))[3]. Additional surveys were conducted in Barataria Bay, April-July 2015 but were not included in the analysis presented here as the resulting data were available in time for inclusion in subsequent analyses (*i.e.* the population modeling). However, results in 2015 were similar in that only 2 out of 10 pregnant females were seen with a calf.

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[2] A "viable calf" in this context is an offspring that survives for sufficient duration in order to be observed during one of the targeted vessel surveys. In all situations, losses were attributed to either fetal or neonatal death.

[3] To add to the results reported in Wells *et al.* 2014, two additional reproductive outcomes from Sarasota Bay, FL, were assessed using blubber endocrine evaluations. These were added in efforts to 1) obtain a higher degree of precision for the reference failure rate estimation and 2) ensure each group (i.e., the oil-affected stocks and reference stocks) contained blubber endocrine evaluations for pregnancy detection. Also, note that one of the observations from Charleston Harbor, SC, (individual "82503"; Bergfelt *et al.* 2013) was updated with additional sighting data leading to one correction of status from presumed failure to known success.

We evaluated the relationships between reproductive failure and five potential covariates (sighting frequency, ordinal date, percent lipid, blubber progesterone concentration, and blubber cortisol concentration) with a Bayesian-logistic-regression –model-averaging technique (with reproductive success as the Bernoulli response(Carlin and Chib 1995). The results indicate no strong evidence (Bayes factor < 20) of a relationship between reproductive failure and these covariates. The high reproductive failure rates measured in both oil-affected stocks following the DWH oil spill are consistent with mammalian literature that shows a link between petroleum exposure and reproductive abnormalities and failures (Bui et al. 1986, Mazet et al. 2001, Archibong et al. 2002, Detmar and Jurisicova 2010, Adedara et al. 2014, Zhan et al. 2015), and also could be related to poor maternal health (Gordon et al. 1970, Rodrigues and Niederman 1992, Schwacke et al. 2014, Smith et al. 2015a).

## 2.5. Estimation of Mortality Due to DWH for Other BSE and Coastal Stocks Based on Strandings

Cetacean strandings in the northern Gulf increased in 2010 prompting the declaration of the Northern Gulf of Mexico Cetacean Unusual Mortality Event (UME; http://www.nmfs.noaa.gov/pr/health/mmume/cetacean_gulfofmexico.htm). The largest increase in bottlenose dolphin strandings occurred after the DWH oil spill in coastal areas most impacted by the spill (Litz et al. 2014, Venn-Watson et al. 2015b). Tissue analyses showed that dolphins stranded after the DWH spill, and within the DWH oil spill footprint, were more likely than reference dolphins to have distinct lung and adrenal lesions (Venn-Watson et al. 2015a). These lesions were also consistent with the adrenal and lung disease detected in Barataria Bay dolphins during the health assessments. This evidence supports the hypothesis that the DWH oil spill was a major contributor to elevated mortalities that precipitated the UME.

Multiple estuarine and coastal bottlenose dolphin stocks live within the DWH footprint, but it is not possible to assign visually a stranded bottlenose dolphin to stock. In addition, not every dolphin that dies is found. Many carcasses are scavenged by predators, or decompose and sink before they reach land, and for those that do reach land the chance that they will be found is dependent on a variety of factors (Peltier et al. 2012, Peltier et al. 2014) including the ability to access an area (for example, it is very difficult in the remote marsh habitats of Louisiana). Therefore, in order to estimate the total number of dolphins that died, a series of analyses was performed. The analyses started with estimating what proportion of the recovered dead stranded bottlenose dolphins belong to estuarine stocks and what proportion belonged to coastal stocks (Section 2.5.2). Next, twenty years of historical stranding data were used to understand baseline stranding rates and estimate the degree of excess mortality due to the DWH spill through a regression analysis (Section 2.5.3). Finally, the results of the carcass assignment proportions and the regression analysis were used in conjunction with estimates of the probability that a dolphin living in the estuaries or in coastal waters would strand on shore and be detected if it died to scale the number of dead carcasses up to an estimate of total mortality for each stock (Section 2.5.4). The mortality estimates are then used to estimate the excess proportion of dolphins killed from each stock as a metric of injury (Section 2.6), and also used in the population model (Section 2.7) to predict long-term impacts to each stock.

### 2.5.1. Stranding Response

The Southeast US Marine Mammal Stranding Network (SEUS MMSN) is a network of primarily volunteer-based organizations authorized to respond to and collect standardized data from dolphin and whale strandings (Litz et al. 2014, Venn-Watson et al. 2015b). For each stranding, the responders fill out a standardized form (Level A data form), which has the basic details of the stranding event such as: species, date, location, length, sex, etc. Those data are tracked by National Oceanic and Atmospheric Administration's (NOAA) National Marine Fisheries Service (NMFS) Marine Mammal Health and Stranding Response Program (MMHSRP). In addition, depending on carcass decomposition state, a suite of tissue samples is collected and preserved for a variety of analytical tests including genetic and stable isotope analysis. During the Northern Gulf of Mexico Unusual Mortality Event, samples were collected, archived, and distributed under chain of custody controls/protocols.

### 2.5.2. Carcass Assignment

Although assigning stranded bottlenose dolphins to stock cannot be done visually, other methods, including genetic assignment and use of stable isotope analysis, may be used. Genetic differentiation between estuarine and coastal stocks is significant (Sellas et al. 2005) and habitat and prey differences for dolphins living in estuarine and coastal environments can lead to different stable isotope signatures between estuarine and coastal dwelling dolphins (Barros et al. 2009). Therefore, in the first step of determining what proportion of stranded carcasses collected by the stranding network came from estuarine versus coastal stocks, genetic assignment tests and stable isotope analyses were conducted on recovered carcasses from Louisiana, Mississippi, Alabama and the western panhandle of Florida (Figure 8). A Bayesian framework was then used to combine the output of these two datasets and estimate total proportions of carcasses attributable to estuarine and coastal stocks.

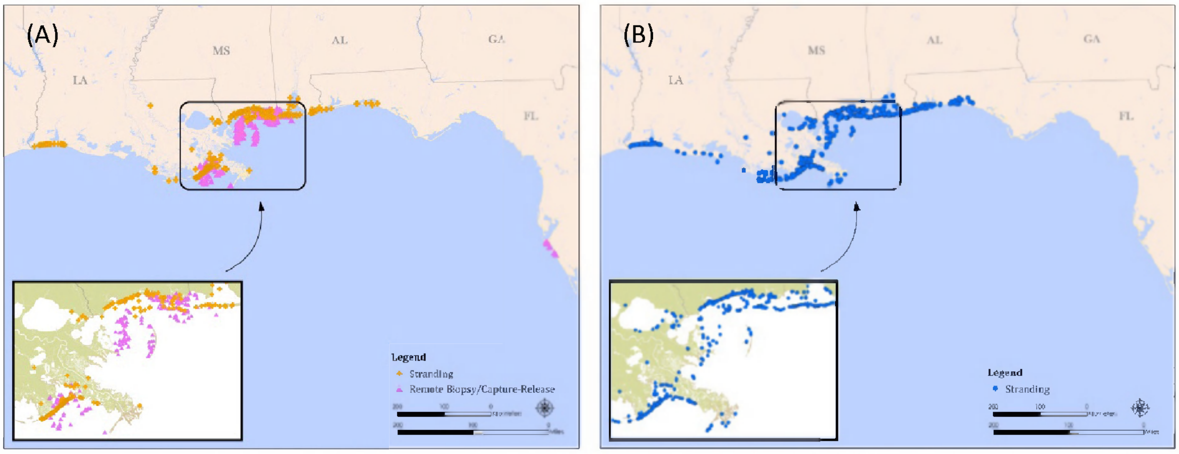


**Figure 8. Locations of samples used for carcass assignment (A) from live and stranded dolphins. Live dolphin samples were collected using remote biopsy or capture-release, and were used as a baseline to train carcass assignment algorithms. The algorithms were then applied to the samples from the dead carcasses to predict source stock type (BSE or coastal). Additional dolphins stranded (B) and were assigned based on the output of the Bayesian analysis but were not sampled for genetic or stable isotope analyses, usually due to state of decomposition.**

### 2.5.2.1. Genetic Assignment Test Analysis

Genetic assignment test methods (Manel et al. 2005) were applied to dolphins stranded in the Barataria Bay area of southeastern Louisiana to determine the proportion originating from the local estuarine stock versus the larger stock found in adjacent coastal waters. Forty-one nuclear microsatellite loci optimized for bottlenose dolphins were used to genotype 156 live dolphins biopsied remotely to provide baseline samples representing the two bottlenose dolphin stocks that would serve as the most likely populations of origin for the stranded dolphins in the Barataria Bay area (i.e., the Barataria Bay stock and the Western Coastal Stock). Because the Barataria Bay stock was not genetically homogeneous, the baseline samples were allocated to two strata (Barataria Bay Islands and Barataria Bay estuarine, similar to the strata created for the spatial robust model). Stranded dolphins (n= 129) of unknown origin recovered from the same region were also genotyped at the same 42 loci. Both classical assignment tests and genetic stock identification methods using GeneClass2 (Piry 2004) and ONCOR (Kalinowski et al. 2007), respectively, were applied to the data. GeneClass2 was used to eliminate any strandings that did not appear to originate from Barataria Bay or the Coastal stock. The ONCOR software was then used to produce probabilities of assignment for each stranded animal to BB Islands, BB estuarine or the Coastal stock. These probabilities were used as input to the Bayesian framework described below.

### 2.5.2.2. Stable Isotope Analysis

Stable isotopes are forms of a chemical element differing by the number of neutrons in the nucleus and not undergoing radioactive decay; that is, the number of neutrons remains constant (stable) over time. Essentially, stable isotope ratio refers to the percentage of the heavier form(s) (more neutrons, e.g., $^{13}C$) to the lighter form (e.g., $^{12}C$) in a sample, which is denoted as $\delta^{13}C$. The relative amount of heavier (rarer) isotope effectively becomes a chemical tracer reflecting diet and habitat (Newsome et al. 2010). The most common stable isotopes in use are those of carbon ($\delta^{13}C$), nitrogen ($\delta^{15}N$), and sulfur ($\delta^{34}S$).

Stable isotope ratios were evaluated as a means of assigning stranded dolphins to coastal or estuarine stocks; as with the genetic assignment analysis estuarine dolphins were further stratified to identify barrier island inhabitants. To describe the relationship between stable isotope ratios and stock, a subset of biopsy samples of skin collected from Louisiana and Mississippi (n=335) provided a training dataset of estuarine, barrier island, and coastal dolphins that was analyzed using a Classification and Regression Tree (CART) (Sutton 2005, Therneau and Atkinson 2015). Candidate covariates were discrete variables Region (East or West of the Mississippi River delta) and Sex, and continuous variables $\delta^{13}C$, $\delta^{15}N$, and $\delta^{34}S$. In sequence from CART results, significant covariates were $\delta^{34}S$, Region, and $\delta^{13}C$. A model including these variables gave a correct classification rate of about 80%. Stable isotope ratios were analyzed for a subset of stranded dolphins (n=217) defined by location (ranging from western Florida to western Louisiana, incorporating the area of biopsy sampling), extent of decomposition (primarily fresh and moderately decomposed), and total length (> 170cm to exclude nutritionally dependent calves). Findings from the biopsy training dataset were applied to these strandings to assign them to an estuarine, barrier island or coastal group. These probabilities were used as input to the Bayesian framework described below.

#### 2.5.2.3. Bayesian Framework to Estimate Proportions of Carcasses Attributable to Estuarine and Coastal Stocks

The genetic stock assignments of the 156 strandings from the Barataria Bay area and the stable isotope assignments of the 217 strandings collected from western Florida to western Louisiana were combined into a Bayesian framework so that the two independent datasets could be used for final inference of the number of strandings from estuarine stocks versus those from coastal stocks. A nonparametric bootstrap (Davison and Hinkley 1997) was used to obtain a distribution representing uncertainty in results from each dataset (genetic assignment and stable isotope assignment). The bootstrap distribution of assignment probabilities was then fit to a semiparametric distribution (a mixture of beta distributions) to obtain a distribution of assignment probabilities that was used as input to the Bayesian model. To perform the combined inference, a Bayesian hierarchical model (Parent and Rivot 2012) was constructed that weighted each data source for each animal using a measure of estimated precision (the effective sample size (Morita et al. 2008), and applied what was learned from the animals that had genetic and stable isotope assignment data to those without. Estimates of the proportion of strandings originating from a Coastal Stock based on the combined datasets were produced for three regions: "East" (of the Mississippi outflow), "West" (of the Mississippi outflow) and "Western Louisiana" (west of Vermilion Bay, LA). Overall, a large majority of animals was estimated to have come from the BSE stocks, except in the Western Louisiana region (Table 7).

**Table 7. Summary of inferences from 2 data sources about the probability (p) of being from a coastal stock. The columns are posterior mean p, standard deviation (SE) and lower and upper 95% credible intervals (CI).**

| Region | Probability (p) | SE(p) | Lower CI | Upper CI |
|---|---|---|---|---|
| East | 0.215 | 0.025 | 0.169 | 0.263 |
| West | 0.065 | 0.016 | 0.036 | 0.099 |
| Western Louisiana | 0.652 | 0.081 | 0.487 | 0.803 |

### 2.5.3. Regression Model to Determine Excess Mortality Due to DWH

Data for a regression analysis were extracted from the Historical Southeastern U.S. Marine Mammal Stranding Database (1990 – 1995) and the MMHSRP National Marine Mammal Stranding Database (1996 – 2013). These data are frequently updated as more information becomes available; however, the data used for this analysis were the most accurate available at the time of extraction (April, 2014). The objective of this analysis was to examine correlation between stranding rates in regions of the northern Gulf of Mexico relative to historical baselines, and the occurrence of DWH oil within coastal and estuarine dolphin habitats. In addition, a metric of winter temperature (cooling degree days) was included as an explanatory variable in regression models to evaluate the potential effect of cold winter temperatures that occurred throughout the Gulf during the winters of 2010 and 2011. The goal of this analysis was to identify which stocks experienced higher than expected bottlenose dolphin stranding numbers and whether they were associated with the presence of DWH oil.

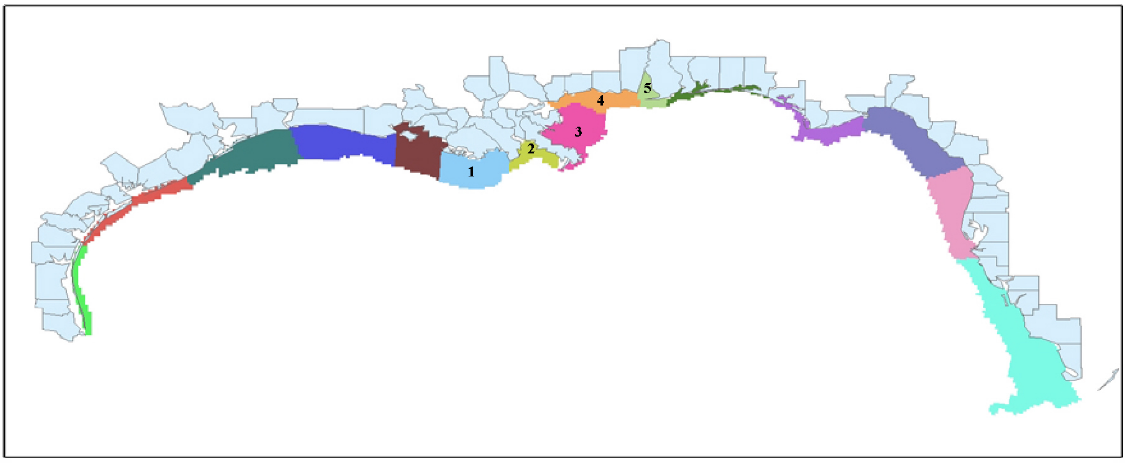


**Figure 9. Fifteen regions from Texas through Florida used in the analysis of excess bottlenose dolphin strandings. Numbered regions are: 1(light blue) Terrebonne/Timbalier Bays; 2 (olive green) Barataria Bay; 3 (pink) Chandeluer Sound/Mississippi River Delta; 4 (orange) Mississippi Sound; 5 (light green) Mobile Bay.**

Regions were defined to align with bottlenose stock boundaries and regional stranding network coverage patterns (Venn-Watson et al. 2015b), see Figure 9. The offshore boundaries of the regions corresponded to the 20m isobath to reflect the impacts of possible exposure to coastal stocks. However, not every region aligns completely with stock boundaries. In particular, the Chandeleur Sound/Mississippi River Delta stranding region encompasses the Mississippi River Delta stock and a portion of the Mississippi Sound stock and the adjacent Northern Coastal Stock. Similarly, the panhandle stranding region encompasses the Perdido Bay, Choctawhatchee Bay, Pensacola Bay, and St. Andrews Bay stocks along with a portion of the adjacent Northern Coastal stock. Regions outside of the DWH oil footprint along the Texas, western Louisiana, and west Florida coasts did not exhibit significant changes in the numbers of strandings relative to their historical baseline. Therefore, the current analysis focused on possible impacts within Terrebonne & Timbalier Bay, Barataria Bay, Chandeleur Sound/Mississippi River Delta, Mississippi Sound, Alabama coastal and estuarine waters, and the Florida panhandle that were within the footprint of DWH oil. Non-perinate (length >115 cm) bottlenose dolphin strandings from 1 January 1990 (1 January 1994 in Louisiana) to 31 December 2013 were assigned to these regions based upon the stranding location. This monthly stranding number was the response variable in a set of log-linear regression models (details below) for each region. Explanatory variables included terms for year, month (collapsed to seasons), a metric of stranding network activity level as described in (Venn-Watson et al. 2015b), an index of DWH oil coverage, and an index of winter "coldness" (CDD: cooling degree days).

A cumulative index of oil coverage was derived from each stranding region based upon daily summaries of surface oil percent coverage developed by the Oil On Water (OOW) technical working group. This data set summarized available information from remote sensing platforms to quantify the percent coverage of surface oil within 5x5 km square spatial cells covering estuarine, coastal, and oceanic waters on a daily basis from 25 April to 28 July 2010 (Graettinger et al. 2015). To develop a cumulative monthly index of oil coverage, the daily 5x5 km cells within each stranding region (Figure 9) were aggregated by weeks to calculate a weekly average percentage area covered by region. These weekly percentages were then added across weeks to derive a monthly cumulative index of oil coverage within each region (Figure 10). The highest cumulative oiling index was observed in the Chandeleur Sound/Mississippi River Delta region followed by Barataria Bay. Mississippi Sound, Alabama, and the Florida panhandle had similar levels of oil coverage; however, it should be noted that oiling did not extend into estuarine habitats along the Florida panhandle. Regions in Texas and along the west Florida coast did not have evidence of surface oil.

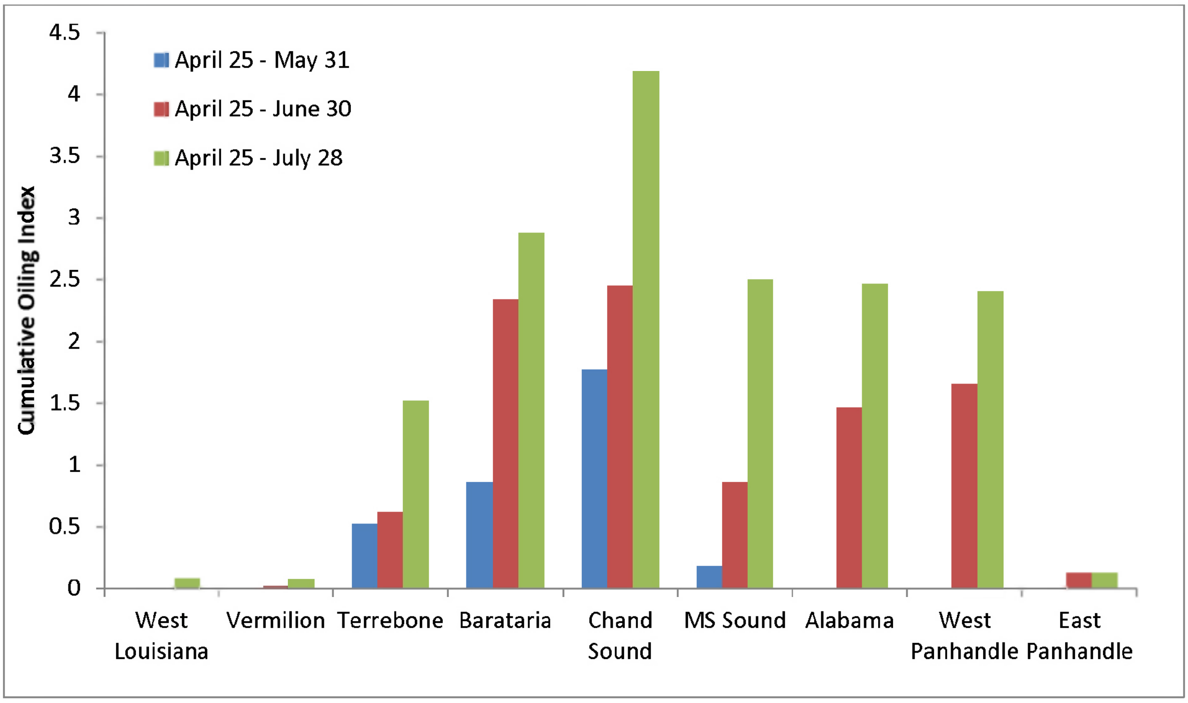


**Figure 10. Cumulative oiling index for stranding regions over the time period 25 April to 28 July 2010. Chand Sound refers to the Chandeleur/Mississippi River Delta stranding region. MS Sound = Mississippi Sound.**

The cooling degree days index was derived from a daily sea surface temperature (SST) data set that covers the entire Gulf of Mexico from 1 January 1990 – 31 December 2013 at a 0.25 degree spatial resolution. These data were derived from both remotely sensed and modeled SST data developed by NOAA's Atlantic Oceanographic and Meteorological Laboratory (John Quinlan, SEFSC, pers. comm.). Within each region, the 10$^{th}$ percentile of winter water temperatures for the entire time series was identified, and the annual index of cooling degree-days then calculated as the sum of the difference between this lower bound and the observed daily temperatures during the winter of that year (Figure 11). The resulting index demonstrates that 2010 and 2011 both had very cold winters throughout the Gulf of Mexico.

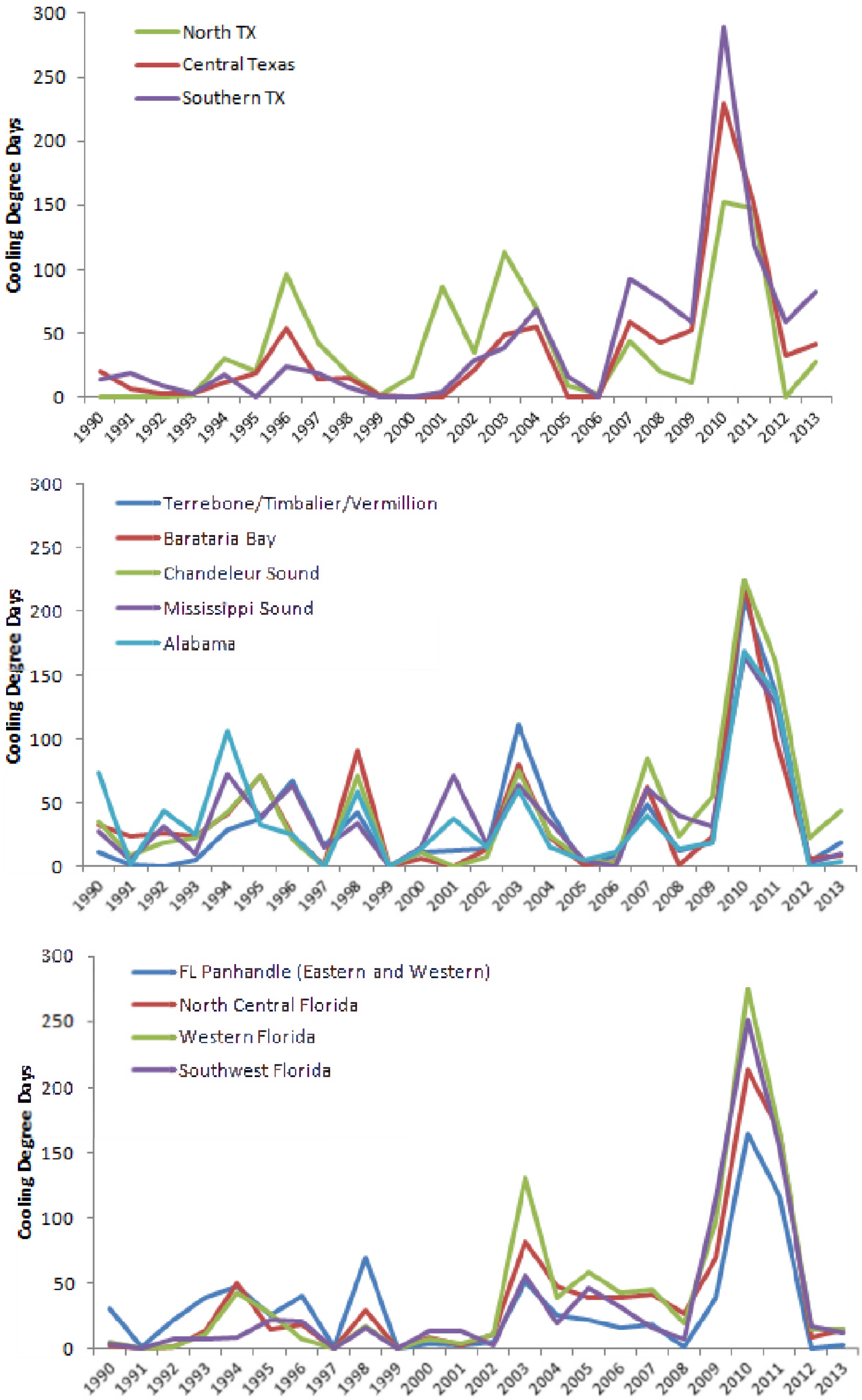


**Figure 11. Regional cooling degree days: 1990-2013 in regions of the Gulf of Mexico.**

Log-linear regression models for each region using an annual trend, season, the annual cooling degree days, and the cumulative oiling index as explanatory variables and the monthly total number of strandings as the response variable. Initial models were fit with categorical terms for the month explanatory variable, and then months were aggregated into simpler (binary) seasonal effects where there were no significant differences between the monthly terms. In most cases, stranding rates were significantly elevated during spring months (February – May) compared to other times of the year.

For months prior to May 2010, the cumulative oiling index had a value of zero. For May and June 2010 the oiling index was the cumulative value for April 25-May 31 and April 25-June 30, respectively. From July 2010 through December 2013, the total cumulative oiling index (April 25 – 28 July) was applied.

Potential serial correlation between successive observations was examined using an auto-regressive model: Generalized Linear Autoregressive Moving Average (GLARMA) model. The negative binomial error structure was used with a 1-month lag correlation. If there was no evidence of autocorrelation in the residuals, then a standard negative binomial Generalized Linear Model (GLM) was used. In general, the GLARMA models were an improvement over models that did not include autocorrelation terms as indicated by likelihood ratio tests.

The linear annual trend term was significant and negative in regression models for most regions with the number of strandings declining over the time series examined (Table 8). Seasonal effects were also consistently significant with higher stranding rates during spring months; however, the months with elevated strandings varied among the regions. Cooling degree days had a positive effect on stranding rates in Terrebonne/Timbalier Bay, Barataria Bay, Chandeleur Sound/Mississippi River Delta, and Alabama suggesting that the cold weather during 2010 may have contributed to elevated stranding rates during the period just prior to the DWH oil spill. This effect was strongest relative to other factors in Terrebonne/Timbalier Bay and Mississippi Sound.

**Table 8. p-values for significance of parameter values in each region. Brackets indicate non-significant tests (p>0.05).**

| Area | Annual Trend | Season | Cooling Degree Days | Cumulative Oiling Index |
|---|---|---|---|---|
| Terrebonne/Timbalier Bay[1] | NA | NA | 0.004 | [0.270] |
| Barataria Bay | < 0.001 | < 0.001 | 0.013 | < 0.001 |
| Chandeleur Sound/ Mississippi River Delta | 0.004 | < 0.001 | 0.008 | 0.004 |
| Mississippi Sound | < 0.001 | < 0.001 | 0.007 | < 0.001 |
| Alabama | 0.006 | < 0.001 | [0.571] | 0.025 |
| Florida Panhandle[2] | 0.009 | < 0.001 | [0.459] | [0.558] |

[1]The annual and monthly terms in the initial model for Terrebonne/Timbalier were not significant, therefore annual and seasonal terms were not included in the final model.

The cumulative oiling index had significant positive effects on stranding rates in Barataria Bay, Chandeleur Sound/Mississippi River Delta, Mississippi Sound, and Alabama. Of these, Alabama had the weakest overall increase in strandings correlated with oil exposure. The oil effects were strongest in Chandeleur Sound/Mississippi River Delta followed by Barataria Bay, where the relative impact of the cooling degree days effect was small. In Mississippi Sound, the oil effect was strong, but cooling degree days also had a strong effect on stranding rates. Figure 12 shows the output of the regression models in the four regions with significant oiling effects. The red line in these plots indicate the predicted number of strandings from the baseline model in the absence of exposure to DWH oil while the black line represents the modeled number of strandings based on the observed data in the presence of DWH oil. The difference between these two lines provides a metric of "excess" strandings associated with oil exposure.

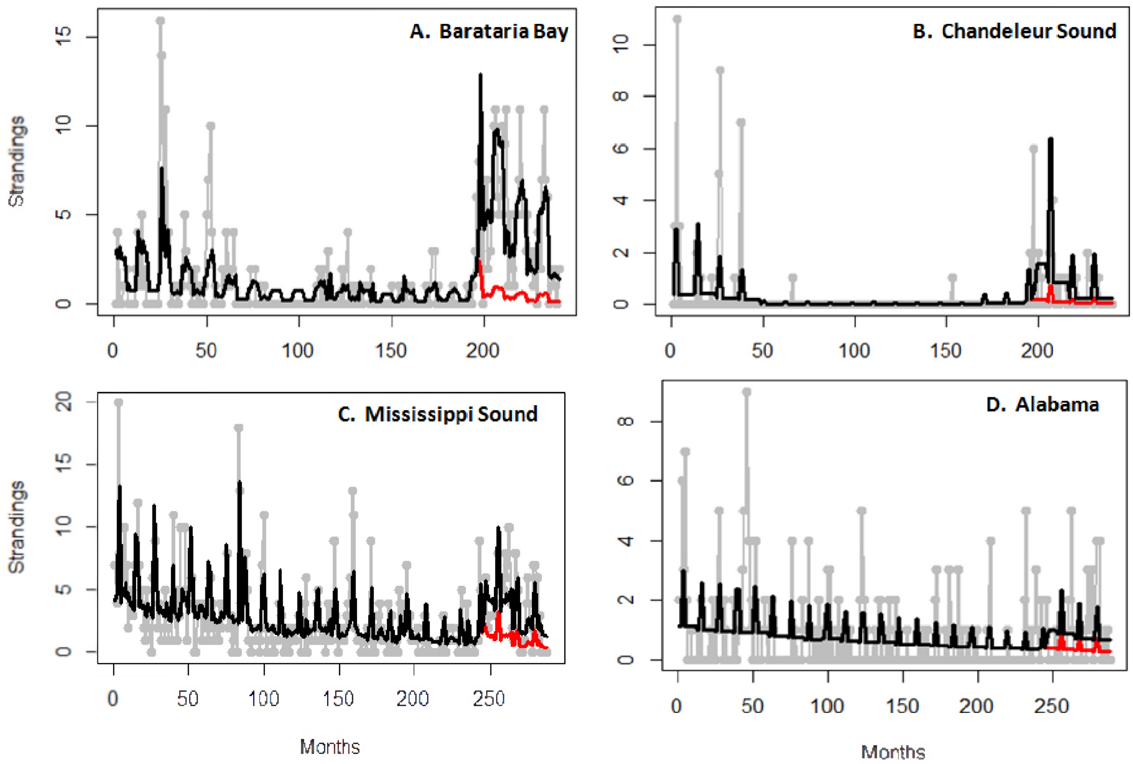


**Figure 12. Observed (gray) line and predicted (black line) strandings by month from the log-linear regression models. The red line indicates the predicted number of strandings in the absence of DWH oil. For the Barataria Bay and Chandeleur Sound/Mississippi River Delta regions, the time series is from 1 January 1994 to 31 December 2013. For the Mississippi Sound and Alabama regions, the time series is from 1 January 1990 – 31 December 2013.**

In Terrebonne/Timbalier Bay, the oil effect was not significant while the cooling degree days effect was significant and positive. The observation of elevated strandings in Terrebonne/Timbalier Bay was restricted to intermittent increases in the spring-summer of 2010 and 2011, and the elevated strandings did not persist in this area beyond 2011 as they did in other regions. As a result, any potential oil effect in Terrebonne/Timbalier Bay is confounded with the potential effect of cold water temperatures during the winters of 2010 and 2011. It is not possible to determine if this short term increase in strandings in Terrebonne and Timbalier Bays was associated with oil exposure, cold temperatures, or both. It is also possible that this is due to changes in effort since these areas are characterized by sparsely populated marshes.

Based upon the uncertainty in parameter estimates, a parametric bootstrap approach was used to quantify the variability in the excess strandings estimate from the regression models (Table 9).

**Table 9. Summary statistics for bootstrap distributions of number of excess strandings derived from regression models.**

| Region | Model Estimate | Bootstrap Median | Bootstrap Lower 95% Confidence Interval | Bootstrap Upper 95% Confidence Interval |
|---|---|---|---|---|
| Barataria Bay | 189 | 192 | 132 | 270 |
| Chandeleur Sound/ Mississippi River Delta | 39 | 39 | 13 | 78 |
| Mississippi Sound | 102 | 101 | 59 | 145 |
| Alabama | 24 | 25 | 4 | 48 |

### 2.5.4. Scaling Excess Strandings to Estimate Total Number of Excess Mortalities

The number of stranded animals counted on the shoreline is a function of several processes that vary in both time and space. The first process is the generation of carcasses which is a function of both population size and mortality rate. The next process is the transport of a floating carcass to shore, which is a function of the location where the animal died, wind and wave patterns that dictate onshore transport, and the time the carcass spends floating in the water column. Finally, once the carcass makes it to the shore, it must be detected and recorded which is dependent upon multiple factors including human use of the shore and the activity level and resources of the local stranding network. The goal of this analysis is to estimate each of the required parameters in order to scale the calculated excess strandings to the number of excess mortalities (i.e., the total number of individuals that died, whether or not a carcass was recovered). The analysis is conducted by geographical stranding region to determine the number of excess mortalities in each region, and then the mortalities are apportioned to the appropriate stock (BSE or coastal) based on probabilities determined by the Bayesian framework described in Section 2.5.2. The analysis is conducted in two steps: estimate probability of a carcass making it to shore (beaching), and (2) estimate the probability that a carcass that makes it to shore is detected.

Ultimately, the scaled excess strandings for Barataria Bay and Mississippi Sound were not used for injury metrics because more direct measures were available (*i.e.*, reduced survival estimates from longitudinal photo-id studies, Table 2); however, the analysis for these two stocks are still reported for completeness.

#### 2.5.4.1. Estimating Beaching Probability (Pb) with the ADCIRC Model

The first parameter to estimate is the probability of beaching (Pb) which varies both spatially and temporally. The primary tool used to accomplish this was the ADCIRC hydrodynamic model (Wirasaet et al. 2015). ADCIRC is a two-dimensional finite element hydrodynamic model that is designed for coastal inundation studies in nearshore environments. The model includes very high resolution bathymetry and shorelines along with input data on winds and freshwater flow inputs from a broad variety of sources. The model has undergone validation by comparisons between observations and predicted flows and transport fields (Wirasaet et al. 2015). Because it is explicitly designed for nearshore transport studies, it is particularly suitable for the current application.

The ADCIRC model implements particle tracking through the modeling of neutrally buoyant Lagrangian particles with approximately 16 million tracked elements within the model domain.

Simulations were initiated by "seeding" virtual particles throughout the model domain within defined regions. The movements of particles were tracked for two-week periods and daily outputs were stored recording the total number of particles in each region indexed by the source of the particles. As particles moved through the spatial domain, if they intersected with a region designated as a "beach" or "land", then the particle does not undergo additional movement, thus simulating the beaching of a stranded carcass. In addition to the forcing due to water currents, an additional forcing was applied to simulated particles to account for the effect of surface winds that tend to increase the velocity of floating objects. In this case, an additional velocity component equal to 1% of wind forcing was added to the particles (Wirasaet et al. 2015). Finally, it was assumed that floating dolphin carcasses would be transported for up to 5 days before sinking and undergoing no additional transport. Simulations were reset and run every week between 1 June 2010 and 15 June 2011.

The mean probability of beaching from various sources averaged across all simulated weeks was used as an estimate of the probability of a carcass beaching in a given location. In general, the probability that a carcass generated in waters greater than 20m depth would beach was extremely low, typically <1%. The probability of beaching for carcasses generated from coastal stocks was on the order of 5-10% for most regions, and this probability was heavily influenced by wind fields. Carcasses generated within nearshore coastal waters (< 2km from shore) or inside estuaries had a high probability of beaching, typically greater than 50% in most areas. For each stranding region of interest, the average probability of beaching from appropriate source regions was used as the estimate of Pb.

#### 2.5.4.2. Estimating Recovery Rate (Pr) and Scaling to Mortality

For a given stranding region, there are *n* possible carcass source regions that are identified from the ADCIRC model runs. Each of those regions is designated as a potential source of carcasses from either an estuarine or coastal stock. If $N_i$ is the expected abundance of animals within a source region derived from either capture-mark recapture density estimates or aerial survey data as appropriate for the source region and M is the mortality rate (1- survival rate) then the expected number of carcasses (C) that will arrive on the beach is:

$$C = \sum_{i=1}^{n} Pb_i N_i M,$$

where $Pb_i$ is the probability that a carcass starting in source *i* will intersect with the beach. The ratio between the annual average observed number of strandings and *C* provides an estimate of the probability of recovery (Pr) of carcasses given that they intersect with the beach. It is expected that mortality rates increased following the DWH event, and it is also possible that increased awareness and sampling effort in some regions may have changed the recovery rate of beached carcasses. Both of these factors could contribute to the "excess" strandings observed in the regression models. Therefore, a simple population model was applied that assumed that the survival rate was equal to 0.95 for the years prior to the spill based on a previous reported rate for a BSE stock (Speakman et al. 2010) and then decreased to an annual rate of 0.83 during 2010-2013 (average of rates for Barataria Bay in Table 2). These decreased survival (increased mortality) rates were used to calculate the annual expected number of carcasses for each region. Given the observed number of strandings during 2010-2013, the post-spill recovery rate (Pr) of beached carcasses was estimated as the ratio of observed strandings to C.

To quantify the excess mortality due to DWH oil by stock, the estimate of excess strandings from the chosen log-linear regression model for the given region was apportioned to the coastal vs. estuarine stock based upon the hierarchical Bayesian model, then divided by the product of the beaching probability (Pb) and the estimated recovery rate (Pr). The result was summarized by year to provide an estimate of the annual mortalities for each stock. Finally, a bootstrap distribution of excess mortality was derived by resampling from the distributions representing uncertainty in estimates of excess strandings, abundance, and the probability of a carcass being from the coastal or estuarine stock (assuming each distribution was independent of each other).

Parameter estimates and resulting estimates of excess mortality for each stranding region are shown in Table 10. The probability of beaching for mortalities from coastal strata was relatively low, ranging from 0.054 – 0.107. In contrast, the probability of beaching from estuarine sources ranged from 0.254 – 0.370. The calculated post-spill recovery rate varied strongly among areas. For the Barataria Bay region, the estimated post-spill recovery rate was very high at 0.462 (Table 10). It should be noted that the estimated recovery rate of carcasses prior to 2010 was extremely low. The level of effort of the local network along with the presence of response staff and other human use of the region may have contributed to higher recovery rate post-spill. Similarly for the Chandeleur Sound/Mississippi River Delta region, prior to the spill there were very few observed strandings due to the remote location, few occupied beach areas, and limited stranding network coverage. The estimated post-spill recovery rate for this area remained very low, but was estimated to be nearly twice as high as that prior to the spill. The resulting excess mortality estimates thus reflect the contribution of both increased efficiency at recovering carcasses and increased mortality rates associated with exposure to DWH oil.

**Table 10. Estimated excess mortality for coastal and estuarine dolphins by stranding region. Pb = probability of beaching, Pr = probability of recovering carcasses that beach. 95% confidence intervals (CI) are from the bootstrap distribution reflecting uncertainty in estimates of excess strandings and the probability of beaching.**

| Stranding Region | Pb (Coastal) | Pb (Estuarine) | Pr (Post-spill) | Excess Strandings | Excess Coastal Mortality (95% CI) | Excess Estuarine Mortality (95% CI) |
|---|---|---|---|---|---|---|
| Barataria Bay | 0.100 | 0.338 | 0.462 | 189 | 238 (107 – 469) | 1142 (774 – 1728) |
| Chandeleur Sound/ Mississippi River Delta | 0.054 | 0.254 | 0.078 | 39 | 1928 (601 – 4215) | 1549 (663 – 3266) |
| Mississippi Sound | 0.098 | 0.370 | 0.192 | 102 | 1112 (599 – 1877) | 1118 (691 – 1675) |
| Alabama/ Mobile Bay | 0.107 | 0.302 | 0.296 | 24 | 162 (19 – 368) | 216 (27 – 456) |

Excess strandings were apportioned from the defined stranding regions to the appropriate coastal and estuarine bottlenose dolphin stocks. The coastal animals stranding in the Barataria Bay region most likely come from the Western Coastal Stock of bottlenose dolphins while the remaining regions are adjacent to the Northern Coastal Stock of bottlenose dolphins. In addition, the Chandeleur Sound/Mississippi River Delta stranding region includes the Mississippi River Delta Stock and part of the Mississippi Sound Stock. Estuarine animal strandings in marshes in the northern portion of Chandeleur Sound are likely sourced from the Mississippi Sound Stock. Based upon animal densities and beaching probabilities, 65.1% of the beached estuarine carcasses were expected to come from sources within the boundaries of the Mississippi Sound Stock while 34.9% were expected to come from sources within the Mississippi River Delta Stock. These proportions were used to apportion the estimated excess mortalities between the two stocks.

Of the two coastal stocks, the Northern Coastal Stock experienced the greatest number of excess mortalities with a total estimate of 3,202 animals. This is a result of the low probability of recovering carcasses in the Chandeleur Sound/Mississippi River Delta region, the high estimated probability of stranded animals being from the coastal stock in this region, and the contribution of strandings in multiple regions associated with the Northern Coastal Stock. The majority of the Northern Coastal Stock's range intersected with DWH oil, therefore it is reasonable to expect a relatively large effect on this stock. In contrast, a relatively small proportion of the Western Coastal Stock was impacted by DWH oil, and the relatively low number of estimated mortalities is consistent with this level of exposure.

**Table 11. Estimated excess mortality by stock (95% confidence intervals from the bootstrap distribution)**

| Stock/Area | 2010 | 2011 | 2012 | 2013 | Total |
|---|---|---|---|---|---|
| Barataria Bay | 234<br>(134 – 414) | 411<br>(269 – 625) | 262<br>(177 – 411) | 234<br>(159 –360) | 1142<br>(774-1728) |
| Mississippi River Delta | 126<br>(37 – 301) | 259<br>(80 – 589) | 76<br>(22 – 192) | 78<br>(24 – 187) | 540<br>(179 – 1126) |
| Mississippi Sound – Marsh | 236<br>(69 – 564) | 484<br>(150 –1103) | 143<br>(41 – 359) | 147<br>(46 – 350) | 1010<br>(335 – 2108) |
| Mississippi Sound | 224<br>(133 – 356) | 455<br>(284 – 677) | 226<br>(135 –352) | 212<br>(129 – 326) | 1118<br>(691 – 1675) |
| Mobile Bay | 33<br>(4 – 75) | 71<br>(9 – 155) | 57<br>(7 – 127) | 55<br>(6 – 117) | 216<br>(27 – 456) |
| Western Coastal | 49<br>(19 – 106) | 86<br>(39-168) | 55<br>(23 – 112) | 49<br>(21 – 100) | 238<br>(107-469) |
| Northern Coastal | 699<br>(350 – 1406) | 1431<br>(731 - 2725) | 540<br>(300 – 987) | 532<br>(300 – 980) | 3202<br>(1784 – 5558) |

There are multiple sources of uncertainty in these estimates as it is difficult to disentangle the multiple factors that may contribute to changes in stranding rates. The bootstrap distributions of the estimates reflect this uncertainty as they account for the underlying variation in the estimates of excess strandings, population size, and beaching rate. Each of these factors contributes a degree of uncertainty to the model. However, the resulting modeled values control for the various factors that contribute to elevated stranding rates to the extent possible and constrain estimated additional mortalities to be consistent with both the size of the exposed populations and the observed decrease in survival rates for effected stocks.

## 2.6. Mortality, Reproductive, and Adverse Health Effects Metrics

Injury metrics express the excess mortality (or for Barataria Bay and Mississippi Sound, reduced survival), excess reproductive failure, and excess adverse effects as a proportion of the population. Injury metrics for the excess proportion of the population killed and the excess proportion of females with reproductive failure due to the DWH-oil spill were estimated based on findings of increased mortality reported in Sections 2.1 (Barataria Bay and Mississippi Sound) and 2.5 (all other BSE and coastal stocks), and findings of reproductive failure reported in Section 2.4. In addition, the proportion of each population with adverse health effects due to the DWH spill was estimated based on the prevalence of disease conditions such as lung disease and abnormal adrenal function in Barataria Bay and Mississippi Sound stocks that was much higher than the prevalence of disease conditions from a reference stock in Sarasota Bay, Florida (Schwacke et al. 2014, Smith et al. 2015a).

### 2.6.1. Excess Proportion of Population Killed

Estimated post-spill survival rates (Table 2) were compared with a previously reported survival estimate from a reference site (Speakman et al. 2010) to calculate excess mortality (*i.e.*, excess proportion of population killed) for Barataria Bay and Mississippi Sound stocks. A bootstrap percentile method was used to calculate confidence intervals. Specifically, beta distributions were fit to survival estimates (Table 2) and also fit to the survival estimate with 95% confidence intervals from a reference site (Speakman et al. 2010). Ten thousand simulations were then run, randomly drawing from the fitted distributions for the reference site and injured sites, and taking the difference between the two proportions. Differences were summed across 3 years for Barataria Bay, but for the Mississippi Sound stock only the single year survival estimate was available (Table 2). The 50th (median2.5th and 97.5th percentiles were calculated (Table 12).

**Table 12. Proportional excess mortality, reproductive failure, and adverse health effects in BSE and coastal bottlenose dolphin stocks due to the DWH oil spill with 95% confidence intervals (CI). Barataria Bay and Mississippi Sound values were estimated from survival rates derived directly from those populations (Section 2.1). For the remaining stocks, values were derived through scaling excess strandings (Section 2.5)**

| Stock | Excess Mortality | | Excess Reproductive Failure | | Excess Adverse Health Efffects | |
|---|---|---|---|---|---|---|
| | Proportion of Population | 95% CI | Proportion of Females | 95% CI | Proportion of Population | 95% CI |
| Barataria Bay | 0.35 | 0.15-0.49 | 0.46 | 0.21 - 0.65 | 0.37 | 0.14 - 0.57 |
| Mississippi River Delta | 0.59 | 0.29-1.00 | 0.46 | 0.21 - 0.65 | 0.37 | 0.14 - 0.57 |
| Mississippi Sound | 0.22 | 0.13-0.29 | 0.46 | 0.21 - 0.65 | 0.24 | 0.00 - 0.48 |
| Mobile Bay | 0.12 | 0.05-0.20 | 0.46 | 0.21 - 0.65 | 0.24 | 0.00 - 0.48 |
| Western Coastal | 0.01 | 0.01-0.02 | 0.10 | 0.05 - 0.15 | 0.08 | 0.03 - 0.13 |
| Northern Coastal | 0.38 | 0.26-0.58 | 0.37 | 0.17 - 0.53 | 0.30 | 0.11 - 0.47 |

Excess proportion of population killed for Mississippi River Delta and Mobile Bay BSE stocks, and the Northern and Western Gulf of Mexico coastal stocks was calculated based on the bootstrap distribution of excess mortalities (Table 11) as a proportion of total population size (Table 6). Again, a bootstrap percentile method was used to calculate confidence intervals.

### 2.6.2. Excess Proportion of Females with Reproductive Failure

Excess proportion of females with reproductive failure was calculated as the reproductive success rate estimated for the reference site (0.65, Section 2.4) minus the reproductive success rate estimated for the DWH exposed sites (0.19, Section 2.4). The difference represents the proportion of females with reproductive failure that was attributable to exposure to the DWH oil; confidence intervals for the difference in two proportions were estimated following the method described by Miettinen and

Nurminen (1985). As reproductive failure was observed consistently across both surveyed sites (Barataria Bay and Mississippi Sound), it was assumed that this metric is applicable across all injured BSE stocks (Mississippi River Delta and Mobile Bay), which were similarly exposed. Likewise, the metric is applicable to the proportion of coastal stocks with cumulative oiling classified to be as severe, or worse, than Barataria Bay.

### 2.6.3. Excess Proportion of Population with Adverse Health Effects

For Barataria Bay and Mississippi Sound stocks, the excess proportion of population with adverse health effects was estimated as the prevalence of dolphins given a guarded or worse prognosis [0.48 and 0.35 for Barataria Bay and Mississippi Sound, respectively (Smith et al. 2015b)] minus the prevalence of dolphins given a guarded or worse prognosis for reference site [0.11 (Smith et al. 2015b)]. The metric for the Mississippi River Delta Stock was based on the estimated proportion for Barataria Bay, and the metric for the Mobile Bay Stock was based on the proportion for Mississippi Sound. For coastal stocks, the Barataria Bay metric was applied for the exposed proportion of the stock, 0.23 and 0.82 for the Western Coastal and Northern Coastal Stock, respectively (method for estimation of exposed portion is described in Sections 3.1 and 3.2.5).

## 2.7. Population Model for Estimating Long-term Population Injury

Bottlenose dolphins are long-lived and slow to mature, and as such, it is difficult for populations to recover from the loss of reproductive adults, particularly when combined with persisting reproductive failure. A population model allows consideration of long-term impacts resulting from individual losses (mortality), adverse reproductive effects, and persistent impacts on survival rates among exposed animals (e.g., continuing reduced survival rates due to chronic disease).

To predict future population trajectories, estimates of survival and fecundity rate are required. Section 2.7.1 describes the approach used to estimate baseline survival rates (*i.e.*, in the absence of oil). Section 2.7.2 describes the population model structure and implementation of the density dependent fecundity that drives population growth, and Section 2.7.3 describes the incorporation of injury measure to assess long-term population effect.

### 2.7.1. Bayesian Model to Estimate Baseline Age-specific Survival Rates

To estimate age-specific survival rates, a Siler 5-parameter competing hazard model (Siler 1979) was fit to age-at-death data for bottlenose dolphins from 5 sites along the U.S. southeast coast using a Bayesian framework. The Siler functional form was selected to model age-specific survival (in 1-yr intervals) due to its previous broad application to long-lived species, and particularly to marine mammals (Barlow and Boveng 1991, Stolen and Barlow 2003, Moore and Read 2008). The data were combined from multiple BSE stocks (Table 13) to obtain an adequate sample size. Although the data were from genetically different stocks, life history characteristics are believed to be similar enough that survivorship pattern would not differ significantly across the stocks. This is supported by previous studies of bottlenose dolphins from several BSE sites along the southeast US coast that have suggested a common basis to biology, behavior, ecology and health (Wells and Scott 1999, Reynolds et al. 2000).

**Table 13. Summary of sources for bottlenose dolphin age-at-death data.**

| Sampling Area | Time Period | Number Males | Number Females | Reference |
|---|---|---|---|---|
| Texas | 1981-1990 | 83 | 83 | Fernandez and Hohn (1998) |
| Mississippi Sound | 1986-2003 | 69 | 42 | Mattson et al. (2006) |
| Sarasota Bay, Florida | 1993-2014 | 51 | 52 | Wells, unpublished |
| Indian River Lagoon, Florida | 1978-1997 | 118 | 72 | Stolen and Barlow (2003) |
| South Carolina | 1991-2012 | 228 | 237 | McFee et al. (2010); McFee, unpublished |

Previous studies have demonstrated differing survival rates for males and females; therefore sex was included as a factor for each of the Siler model parameters. The sex-specific survivorship function of age, $l_s(x)$, and its credible interval were estimated using prior distributions previously suggested for other small cetaceans (Moore and Read 2008). Although it was assumed that survival among the various sites is similar, it was unknown whether the stocks could be experiencing different rates of population growth. Therefore, geographic location was included as a factor for growth rate, $r$. The growth rate for the Sarasota Bay stock was recently estimated as 0.018 based on a long-term photo-id study documenting births and losses (RS Wells, unpublished). Therefore, $r$ for the Sarasota Bay stock was fixed to be 0.018. The growth rates for remaining stocks were simultaneously estimated with the Siler model parameters, with maximum annual growth rate constrained to be within reasonable limits (≤ 5%) as previously proposed for dolphin species (Reilly and Barlow 1985, Slooten and Lad 1991, Mannocci et al. 2012).

The resulting survival curves (Figure 13) were used as input for the age- and stage-structured population model to represent expected age-specific survival rates under baseline conditions (i.e., in the absence of the oil spill).

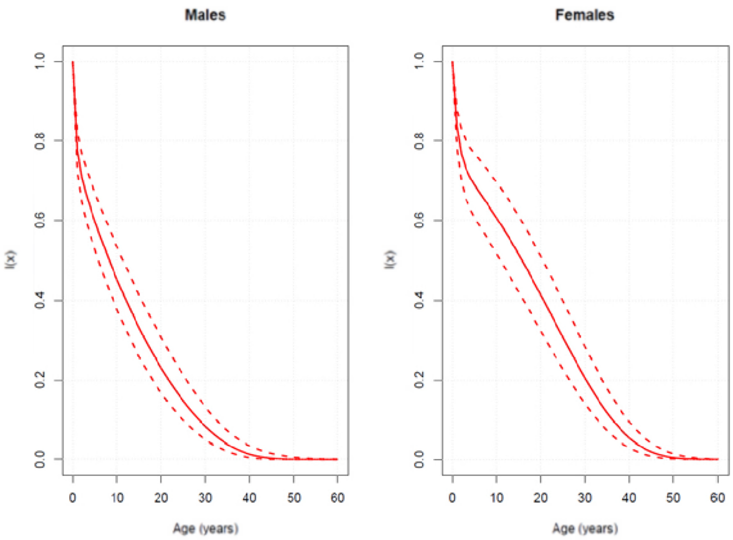


**Figure 13. Sex-specific survivorship l(x) as function of age. Solid line is posterior median, dashed lines represent 95% credible interval.**

### 2.7.2. Age-, Sex- and Class-structured Population Model Structure

An age-, sex- and class-structured deterministic model (matrix model) of population growth was used to characterize the population loss and recovery trajectory following the DWH oil spill. Distributions for model input parameters were constructed to represent uncertainty in parameter values. The same population model framework was used for all BSE and coastal stocks, but a separate model was constructed and parameterized for each stock. One-year age classes were defined and projected separately for females and males, with survival rates based on the estimated sex- and age-specific survival rates as described above (Section 2.7.1).

A density dependent fecundity (DDF) function was derived from an extended Beverton-Holt form previously applied for grey seal populations (Thomas and Harwood 2005). The fecundity rate at time *t* is calculated as:

$$F(t) = \frac{F_{max}}{1+(\beta \times N_t)^{\rho}},$$

where $F_{max}$ is the maximum achievable fecundity rate, $N_t$ is the population size at time *t*, and $\rho$ is a shape parameter. The fecundity rate will approach $F_{max}$ when the population size is low (approaching zero). The $\beta$ parameter is related to carrying capacity, and is defined as:

$$\beta = \frac{1}{N_{nominal}} \times \left(\frac{F_{max} - F_{nominal}}{F_{nominal}}\right)^{\frac{1}{\rho}}.$$

$\beta$ is parameterized based on an assumption about $F_{max}$, and observations of fecundity and population size at a particular point in time, which are deemed $F_{nominal}$ and $N_{nominal}$, respectively. In this case, $N_{nominal}$ was set to the initial population size for each stock and it was assumed the pre-spill fecundity rate for all stocks was 0.24 (range 0.13-0.33) based on previously documented rates from other bottlenose dolphin populations (Wells and Scott 1999, Mann et al. 2000, Haase and Schneider 2001, Thayer 2008) as well as recent data from Sarasota Bay dolphins (Wells, unpublished). $F_{max}$ was estimated as 0.34 (range 0.33-0.41) based on fecundity rates previously documented for dolphin populations impacted by fisheries bycatch (Myrick et al. 1986), high mortality related to epidemic disease (Thayer 2008), or research takes (Kasuya et al. 1997).

It is commonly believed that population growth for dolphins, and more generally cetaceans, is at least partially controlled by density dependent factors, and a relationship among prey abundance, body condition, and pregnancy rate has been documented for cetaceans (Williams et al. 2013). While this suggests that an assumption of density dependent fecundity is reasonable, an actual functional form for the density dependent response is still undefined. Therefore, a range of values for the DDF shape parameter, $\rho$, were evaluated. A distribution for $\rho$ was constructed that provides a range of possible density dependent functional forms believed to be reasonable for cetaceans based on range of previously observed fecundity rates and current BSE stock densities. The same DDF function was applied for all females between the ages of 8 to 48 years, inclusive. This age range was based on previous reports for female age at sexual maturity (Wells and Scott 1999), and observations from Sarasota Bay where the oldest female bottlenose dolphin documented to reproduce was 48 years of age.

### 2.7.3. Incorporation of Quantified Injuries into the Population Model

For each stock's model, two classes, exposed and unexposed, were used to represent the collective population following the spill. For the four BSE stocks (Barataria Bay, Mississippi River Delta, Mississippi Sound, and Mobile Bay), all individuals were considered exposed immediately following the spill, with no members in the unexposed portion of the population. For the two coastal stocks, the proportion of the stock exposed to oil was determined using a cumulative oiling footprint model and densities documented by aerial surveys, as 82.0% and 22.6% for the Northern Coastal and Western Coastal stocks, respectively (see Table 16). All offspring were contributed to the unexposed class with the assumption of no transgenerational effects. The baseline survival and fecundity rates were applied to the unexposed class, while an adjusted survival and/or fecundity rate was applied to the exposed class. Adjusted survival and fecundity rates were based on estimated injuries for the given stock and are detailed in sections that follow. The model for each stock was projected for 150 years under a baseline scenario using the aforementioned baseline parameters to predict what the population trajectory would have been if the DWH spill had not happened (*i.e.*, with all members part of the unexposed class), and then under an injured scenario for the same number of years using the estimated reduced survival and reduced reproductive success for the exposed class. Model outputs (Figure 14) were then calculated as:

- **Years to recovery (YTR)** - the number of years until the DWH-injured population trajectory reaches 95% of the baseline population trajectory
- **Lost cetacean years (LCY)** – difference between the baseline and DWH-injured population size, summed over all 150 years
- **Maximum proportional decrease** – the difference between the two population trajectories when the DWH-injured trajectory is at its lowest point, divided by the baseline

It should also be noted that the years to recovery metric reflects recovery to 95% of population size, whereas lost cetacean years reflects recovery back to 100% of the baseline. Therefore, lost cetacean years will reflect a longer duration than the years to recovery metric. Uncertainty in model output was evaluated by drawing from the distributions for model input parameters to execute 10,000 simulations, producing distributions for each of the model outputs.

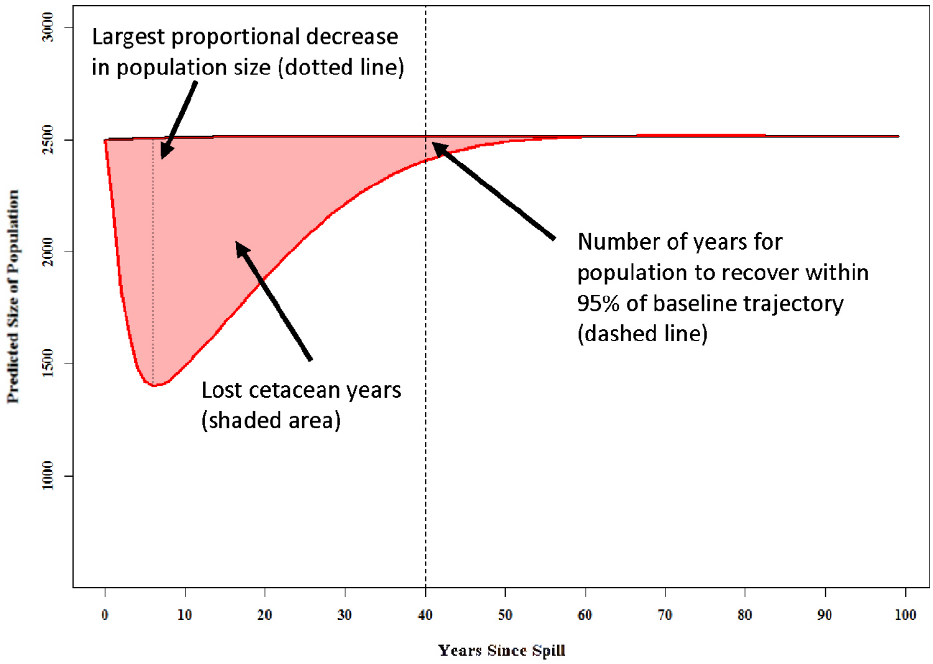


**Figure 14. Population model outputs. Shaded area represents the difference between the baseline and DWH-injured population size, summed over all years (lost cetacean years). The dashed line represents the number of years post-spill until the injured population trajectory reaches 95% of the baseline population trajectory.**

#### 2.7.3.1. Reduced Survival and Reproductive Success Factors for Exposed Class

A reduced survival factor for the exposed class was calculated as the ratio of the observed annual post-spill survival rate to an expected annual baseline survival rate (0.95) previously reported for another non-oiled BSE bottlenose dolphin stock using capture-recapture methods (Speakman et al. 2010). The post-spill survival rate was calculated separately for each stock:

- **Barataria Bay stock**: post-spill survival was taken from the Barataria Bay capture-recapture model results (0.85, 0.83, and 0.80, with associated estimates of uncertainty) for the three years post-spill (Section 2.1)
- **Mississippi Sound stock**: post-spill survival was taken from the Mississippi Sound capture-recapture model result (0.73, with uncertainty estimates) for the first year following the spill (Section 2.1)
- **Mississippi River Delta, Mobile Bay, Northern Coastal, and Western Coastal stocks**: post-spill survival was calculated as the ratio of the estimated number of mortalities for each stock over the three years following the spill (based on full bootstrap dataset for scaled carcasses, Section

2.5, Table 11) to the stock's total population size with associated uncertainty estimates (Section 2.2).

Note that each stock's reduced survival factors were estimated separately for the first three years following the spill (the period over which data were available). The exception was the Mississippi Sound stock for which survival was only estimated for one year due to the limited number of photo-id surveys.

A reduced reproductive success factor for the females in the exposed class was estimated as the ratio of observed reproductive success for the combined observations within the DWH oil footprint (0.19, see Section 2.4) to baseline reproductive success (0.65, see Section 2.4). The same reduced reproductive success factor was applied for all stocks. Note that a separate reproductive success factor was not calculated for different years; observations were pooled to calculate a single value due to the limited number of observations. However, the reproductive success factor was applied for the same number of years as described for the reduced mortality factors above (one year for Mississippi Sound, three years for all other stocks). The application of the reproductive success factor for three years is somewhat conservative given that reproductive success rates were still low in Barataria Bay during 2015.

Insufficient data were available to estimate reduced survival and reproductive success in the exposed classes that likely have continued beyond three years (or beyond one year in Mississippi Sound). However, an assumption that survival and reproductive rates would return to normal (i.e., baseline values) immediately after the observation period was also not supported. Therefore, to estimate future reduced survival (lingering effects), opinion was elicited from six veterinary experts familiar with dolphin health and physiology, as well as the specific dolphin disease conditions following the DWH spill. All experts hold a Doctor of Veterinary Medicine (DVM) degree, have specific experience through their practice and/or research with disease in bottlenose dolphins, and were either part of the NRDA dolphin health assessment studies or participated in the cetacean UME investigation following the DWH spill.

The experts were asked the following question:

> ***Given the observed disease conditions and current evidence for changing/improving condition over time, how many years do you believe it will be before the dolphins with these conditions return to a pre-spill health state?***

The experts were asked to respond to the question separately based on the documented injuries in Barataria Bay and in Mississippi Sound. The responses from the experts were used to calculate two estimates for the number of years that it would take for the reduced survival factor to be equal to 1.0 (i.e., until survival rate became the same as baseline survival rate); one estimate for Barataria Bay, and one estimate for Mississippi Sound. A linear function (straight line) was then fit between the final year of an observed survival rate and the estimated number of years until the reduced survival factor equaled 1.0; this linear function then was used to estimate the reduced survival factor for remaining years. A similar approach was applied to all other stocks.

Future reduced reproductive success was also estimated based on elicitation of expert opinion as described above. The same six experts were asked the following question:

> ***Given the observed disease conditions and current evidence for changing/improving condition over time, how many years do you believe it will be before the female dolphins with these conditions return to a pre-spill reproductive state?***

The responses were used to estimate the number of years it would take for the reduced reproductive success factor to be equal to 1.0, and a linear function was fit between year three (or year one in Mississippi Sound) and the estimated number of years until the reduced reproductive factor equaled 1.0.

Of the six experts, two indicated that they believed that the injuries to dolphins exposed to DWH oil in Barataria Bay would continue as chronic disease conditions, and that these exposed dolphins would never return to a normal (baseline) health state. The remaining four experts indicated that they believed the dolphins would recover to a baseline health state within 10-12 years (mean=10.7 years).

For Mississippi Sound dolphins, opinions were slightly more optimistic but still similar to those for Barataria Bay dolphins. The same two experts again indicated that they did not believe the exposed individuals would ever return to a baseline health state. The remaining four experts indicated that they believed that the Mississippi Sound dolphins would recover to a baseline health state within 8-12 years (mean=9.8 years).

To account for the two of six experts that stated the exposed dolphins would never return to a baseline health state, approximately one third of the population model simulations were executed with reduced survival and reproductive success factors remaining constant for the exposed cohort. The remaining expert results were modeled as a gamma distribution, and for each of the remaining simulations, the number of years until the reduced survival factor was equal to 1.0 and the number of years until the reduced reproductive success factor was equal to 1.0, were randomly drawn from the fitted distributions. The linear functions were then fitted between the year three (or year one in Mississippi Sound) and the randomly drawn number of years.

Lingering effects estimated for Barataria Bay and Mississippi Sound were extrapolated to other stocks. Effects for Mobile Bay were extrapolated from the Mississippi Sound values due to the close proximity of the two sites. The proportion of the coastal stock populations included as part of the exposed class was based on the cumulative oiling footprint model, and specifically on oiling that was estimated to be "as bad as Barataria Bay". Therefore, lingering effects for the coastal stocks were based on Barataria Bay measures. Similarly, due to the significant oiling within much of the Mississippi River Delta, lingering effects for this stock were also extrapolated from Barataria Bay measures.

### 2.7.4. Population Model Results

Population trajectories under both baseline and injured scenarios show high variability (Figures 15 and 16) reflecting the uncertainty on input parameter values as described above. However, the DWH-related population injury was calculated as the difference in paired baseline and injured scenarios (Figure 17) that were executed using the same input parameters with the exception of the estimated reduced survival and reproductive success factors for the injured scenario. Therefore, this variability related to input parameter uncertainty is reflected in the width of the confidence intervals for Lost Cetacean Years (LCY), Years to Recovery (YTR), and maximum proportional decrease (Table 14).

The number of lost cetacean years varied across stocks in relation to the magnitude of injuries as well as the size of stock, or in the case of the coastal stocks, the proportion of the stock exposed. For example, lost cetacean years was highest (LCY=92,069) for the Northern Coastal Stock, which is a relatively large stock (N=7,185) and has a distribution that overlapped significantly with the DWH oil (estimated 82% of the stock exposed). Mississippi Sound, the largest of the BSE stocks (N=4188), also had a high number of lost cetacean years (LCY=78,266). Lost cetacean years was relatively low for the Mississippi River Delta Stock (LCY=20,065) due to its small abundance estimate (N=820). However, this stock is in an area that was heavily oiled, and the estimated number of excess mortalities was relatively high. Due to the high magnitude of injuries, the population model predicted that the Mississippi River Delta stock will take the longest number of years to recover (52 years), and have the greatest proportional change in population size (-0.71). The least impact was indicated for the Western Coastal Stock that had limited overlap with the cumulative oiling footprint (estimated 22.6% exposed), with a proportional (negative) change in the population of only 5%.

**Table 14. Injury metrics output from population models for BSE and coastal bottlenose dolphin stocks. NA for Years to Recovery (YTR) occurred when the model determined that a stock did not decline by more than 5%. Note that Lost Cetacean Years is a metric specific to each stock.**

| Stock | Lost Cetacean Years (LCY) | Maximum Proportional Change in Population Size | Years to Recovery (YTR) |
|---|---|---|---|
| Mobile Bay | 9,362 | -0.31 | 31 |
| Mississippi Sound | 78,266 | -0.62 | 46 |
| Northern Coastal | 92,069 | -0.50 | 39 |
| Mississippi River Delta | 20,065 | -0.71 | 52 |
| Barataria Bay | 30,347 | -0.51 | 39 |
| Western Coastal | 19,041 | -0.05 | NA |

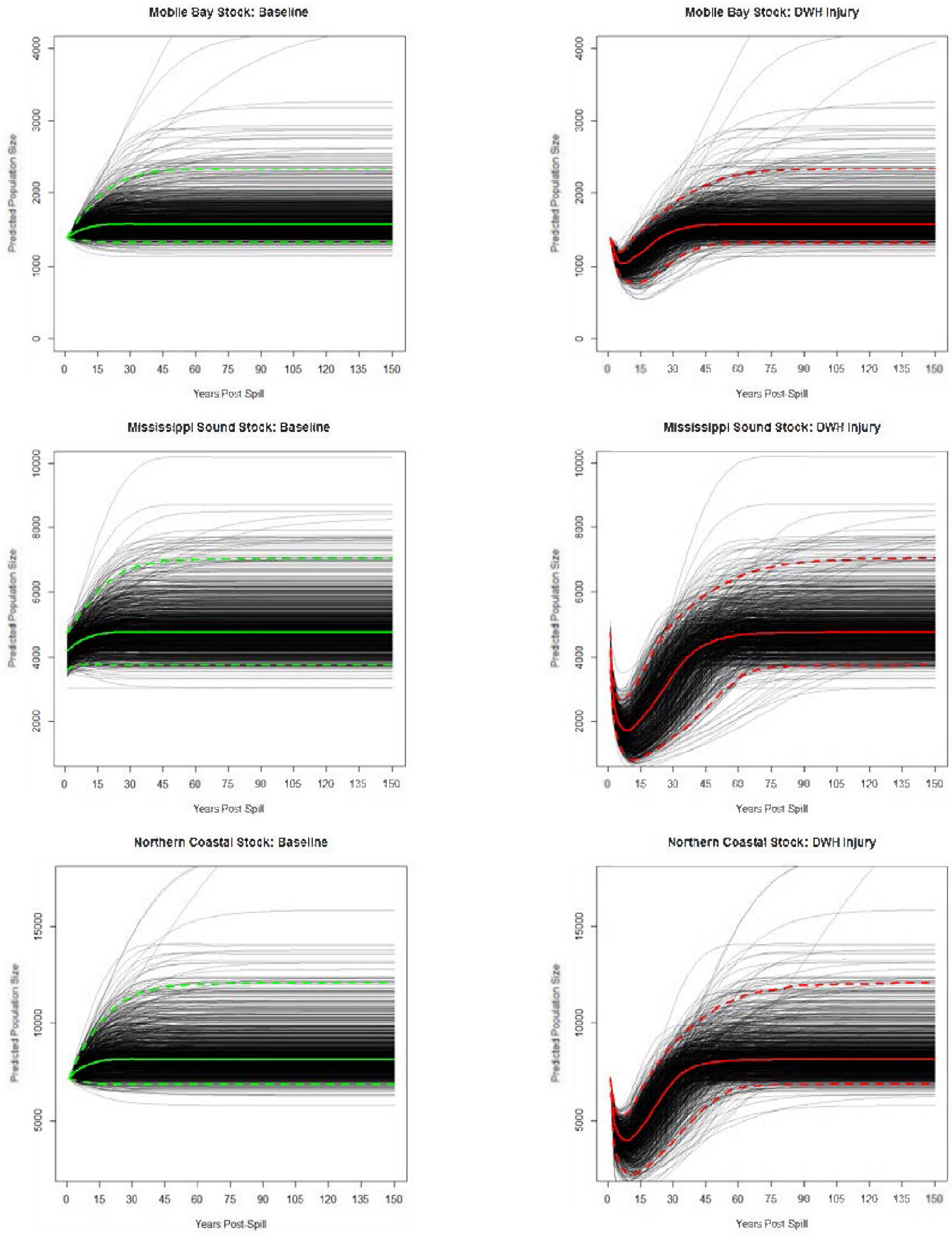


**Figure 15. Simulated population trajectories for Mobile Bay, Mississippi Sound, and Northern Coastal stocks under baseline conditions, and with DWH injury. Each black line represents result from one simulated trajectory; trajectories were thinned by a factor of 10 for graphing. Solid and dashed green/red lines represent median and 95th percentiles for trajectories .**

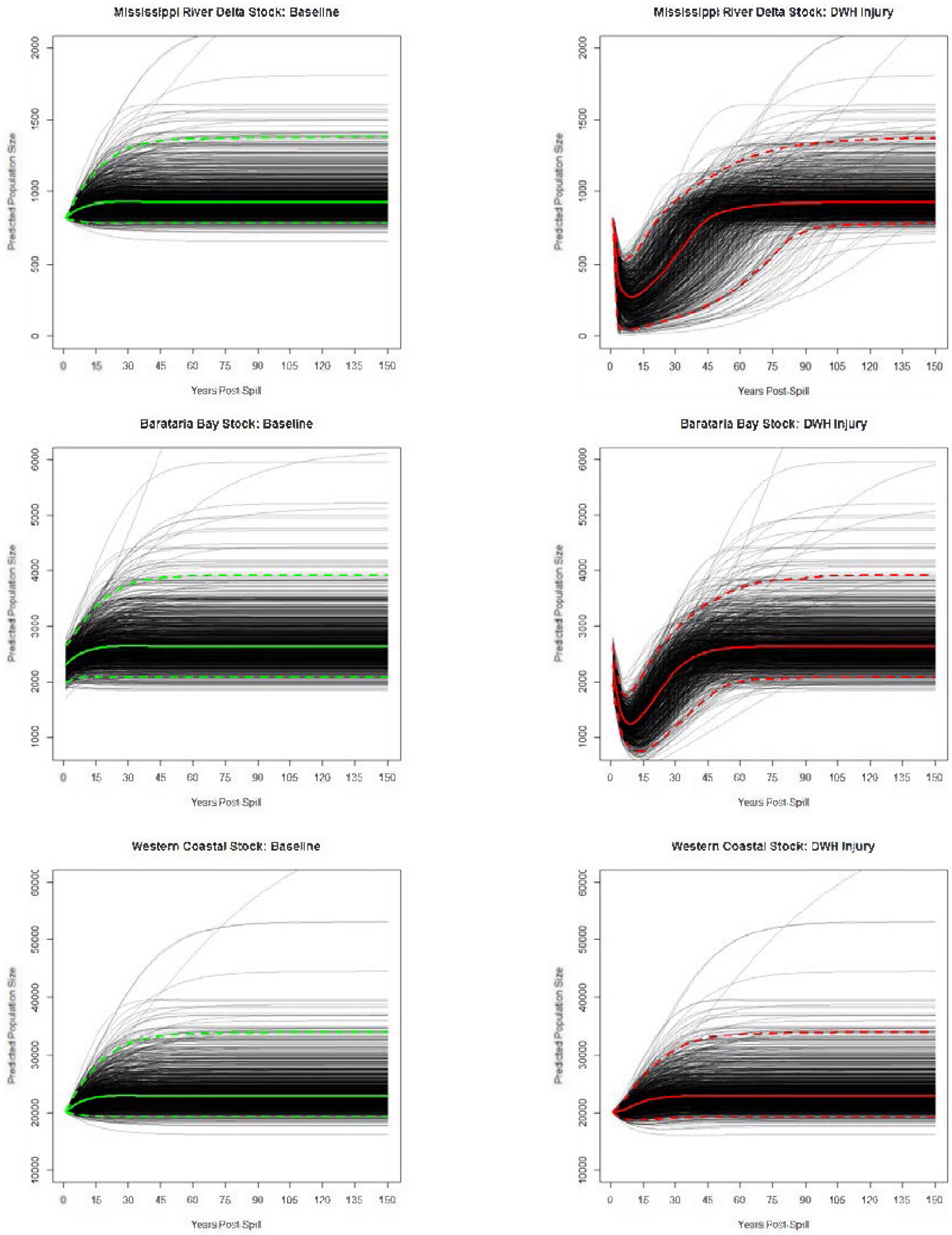


**Figure 16. Simulated population trajectories for Mississippi River Delta, Barataria Bay and Western Coastal stocks under baseline conditions, and with DWH injury. Each black line represents result from one simulated trajectory; trajectories were thinned by a factor of 10 for graphing. Solid and dashed green/red lines represent median and 95$^{th}$ percentiles for trajectories .**

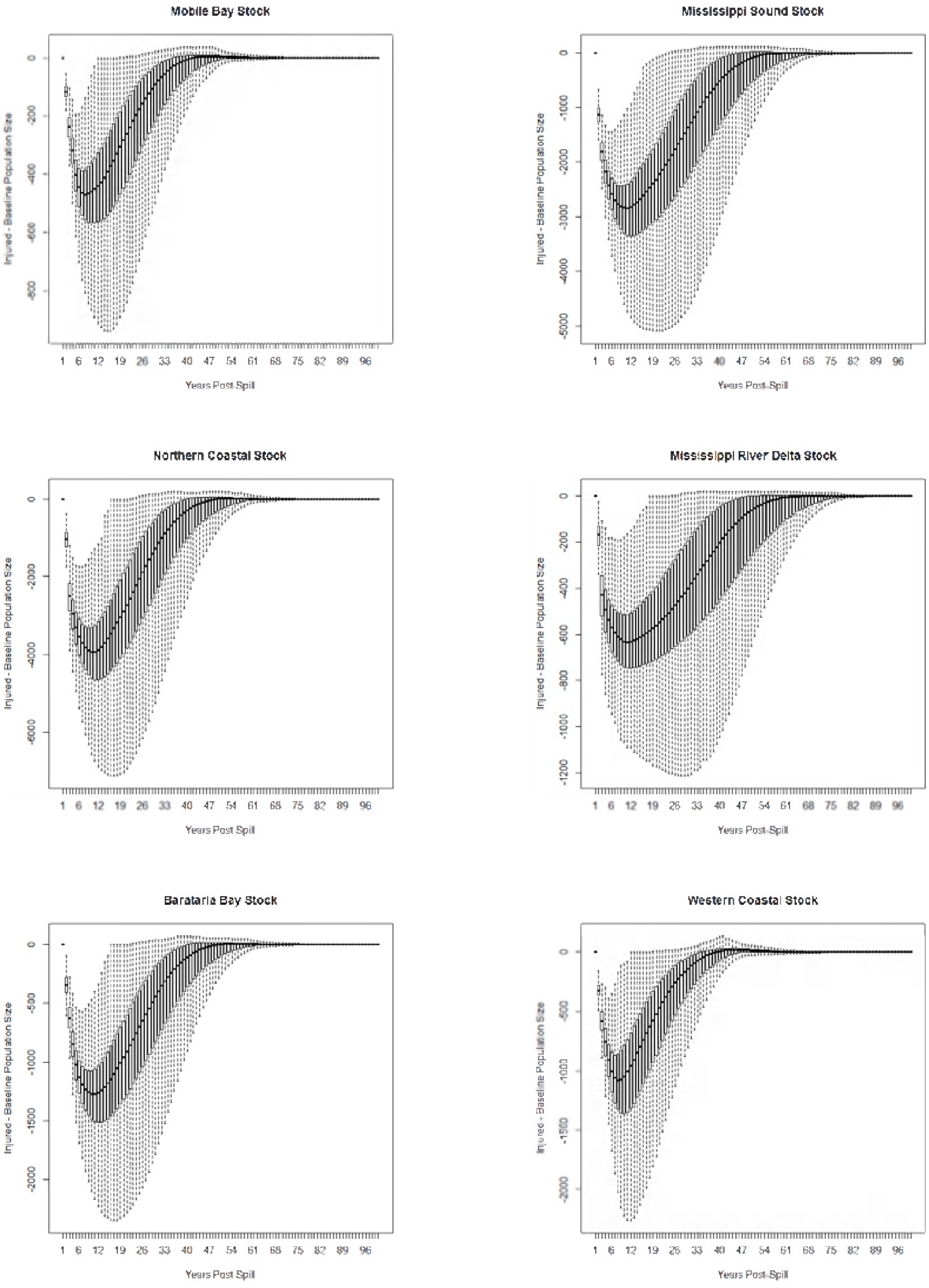


**Figure 17. Box-plots of difference in predicted population size over time due to DWH oil spill (injured - baseline). Boxes represent 25th and 75th percentiles and whiskers are the 1.5 times the interquartile range. Note differing range of values on y-axes.**

## 3. Quantification of Injury to Shelf and Oceanic Cetacean Stocks

Twenty-one cetacean species regularly inhabit shelf and oceanic waters of the Gulf including a diverse group of tropical and sub-tropical species (Waring et al. 2013, Dias and Garrison 2015, Rosel and Mullin 2015). The two primary species that occur over the continental shelf are the Atlantic spotted dolphin and the shelf stock of common bottlenose dolphin (Fulling et al. 2003). The majority of cetacean taxa occupy the oceanic waters of the northern Gulf in waters deeper than 200m. These oceanic species include the endangered sperm whale, a small resident population of Bryde's whales, several species of beaked whales, and a number of delphinids (Rosel and Mullin 2015). Cetacean spatial distribution is strongly influenced by oceanographic and bathymetric features with some species strongly associated with areas of high bathymetric slope and frontal zones (Baumgartner et al. 2001). Bottom depth and bottom depth-gradient are two of the features influencing cetacean distribution in oceanic waters of the Gulf (Davis et al. 1998).

The majority of oceanic and shelf cetacean species within the northern Gulf of Mexico occur in areas that were impacted by DWH oil. In these waters, cetaceans experienced exposure to high concentrations of fresh oil both at the surface and sub-surface, high concentrations of volatile gasses that could be inhaled at the surface, and response activities including increased vessel operations, dispersant applications, and oil burns. Visual observations and photographs from aircraft and vessels documented both the occurrence of cetaceans within the spill area during the summer of 2010 and interactions between individual animals and oil at the surface. Observations included sightings of over 1,100 individuals from at least 10 species occurring in oil from the DWH event (Dias and Garrison 2015). Animals were observed in the region impacted by DWH oil throughout the summer of 2010 and thus likely experienced exposure to oil through inhalation, ingestion, and dermal exposure and exposure to response activities.

It is not possible to sample directly individual animals in oceanic waters and nor to conduct direct assessments of injury using the methods applied to bottlenose dolphins in estuarine waters. In addition, the extremely low probability that animals dying far from shore will eventually strand on beaches limits the ability to collect data on injuries from stranded animals from the offshore species and also makes stranding rates unreliable indicators of elevated mortality for these species, unlike nearshore stocks. Therefore, the observed injuries and mortality rates observed in estuarine and coastal bottlenose dolphins (Section 2.6) were used as a model to infer these rates for the shelf and oceanic cetaceans of the Gulf. These species can be reasonably expected to experience injuries at least as severe as those observed in bottlenose dolphins occupying estuarine waters, in particular those in Barataria Bay. This is likely a conservative estimate of impacts for several reasons. First, shelf and oceanic species experienced exposures of up to 90 days to very high concentrations of fresh oil and a diverse suite of response activities. In contrast, estuarine dolphins were not exposed until later in the spill period and to weathered oil products at lower water concentrations. However, there is a higher likelihood of accumulation of oil in estuarine habitats, and the relative toxicity of fresh vs. weathered oil products is not clear. Second, since oceanic cetaceans dive longer and to deeper depths than shallow water dolphins (Piscitelli et al. 2010, Ponganis 2011), it is possible that the types of lung injuries observed in estuarine dolphins may be more severe in oceanic cetaceans. Third, cetaceans in deeper waters were exposed to very high concentrations of volatile gas compounds at the water's surface near the wellhead. Therefore, the approach in this analysis was to quantify the number of individuals from shelf and oceanic species that experienced exposure to surface oil at levels at least as high as those in Barataria Bay. These individuals were then inferred to have reduced survival and reproductive rates similar to those estimated for estuarine dolphins. These rates were then applied within population models to quantify changes in population sizes, population time to recovery, and lost cetacean years.

## 3.1. Cumulative Oiling Index

In order to identify the area of offshore waters that experienced surface oil at levels at least as high as those in Barataria Bay, an index of cumulative oil exposure in surface waters was derived using the same methods as those described in Section 2.5.3 using a gridded analysis of percentage of oil coverage at the surface within 5x5 km grid cells derived by the NRDA oil on water technical working group (Graettinger et al. 2015). The resulting cumulative oiling index "footprint" is shown in Figure 18. This metric thus allowed a direct comparison of relative exposure level across estuarine, shelf, and oceanic environments.

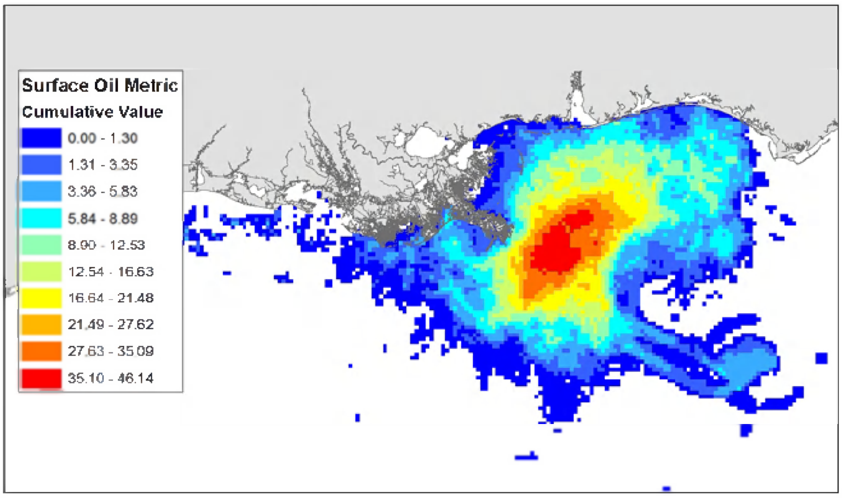


**Figure 18. Cumulative oil exposure index.**

Grid cells that experienced a cumulative oil exposure index equal to or greater than that for Barataria Bay (see Section 2.5.3, Figure 18) were identified and a smoothed polygon was created that outlined these cells. This polygon was then split at the 20 and 200m isobaths to identify regions where high oil exposure likely occurred for shelf and oceanic stocks (Figure 19). These "oil exposure polygons" were used as regions to quantify the expected density and number of animals from each cetacean species that likely experienced exposure to DWH oil at levels sufficient to cause injury as documented in estuarine dolphins (see Section 2.6).

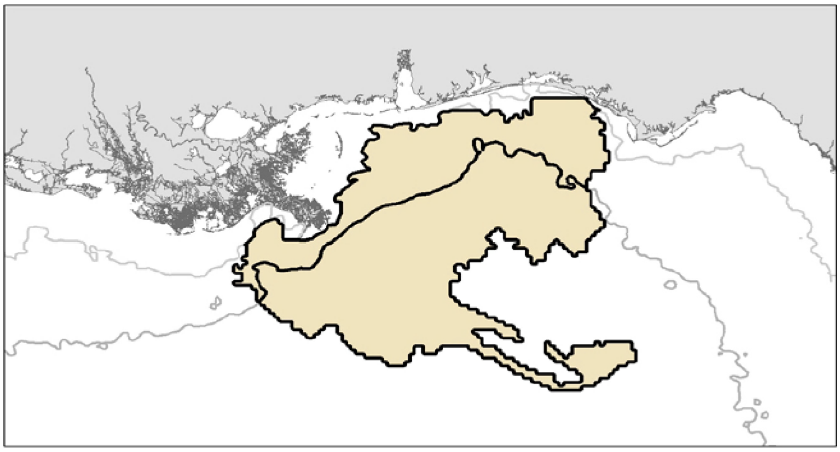

**Figure 19. Smoothed "oil exposure polygons" for regions with a cumulative oil index at least as high as that for Barataria Bay based upon the gridded map of the cumulative oil exposure index. Gray lines denote 20m and 200m isobaths.**

## 3.2. Sources of Density Estimates and Distribution of Populations

Data on cetacean distributions in the Gulf of Mexico have been collected by the Southeast Fisheries Science Center using large vessel and aerial surveys since the early 1990's. These surveys are most typically line-transect surveys that use visual observations to quantify density and abundance and to characterize cetacean spatial distribution and habitat preferences (Dias and Garrison 2015). The most recent available vessel and/or aerial surveys were used to assess the density and spatial distribution of shelf and oceanic cetacean species for this analysis.

### 3.2.1. Shelf Dolphins: 2011-2012 Aerial Surveys

The 2011-2012 aerial surveys (see Section 2.3) were used to quantify the combined abundance of Atlantic spotted dolphins and bottlenose dolphins within continental shelf waters between the 20 and 200m isobaths from Texas to Florida. These two species were combined into a single abundance estimate due to uncertainties in species identification from the air. A total of 30,340 km of trackline was flown in the "shelf" stratum between the 20m and 200m isobaths across the four seasonal surveys (Figure 20). There were 358 groups of dolphins sighted totaling 3,442 animals. Of these sightings, 64 were observed within the oil exposure polygon totaling 587 animals. Distance analysis employing independent observer approaches to account for incomplete detection on the trackline (see Section 2.3) was used to estimate both the total abundance of shelf dolphins and the abundance of animals within the area covered by the oiled polygon.

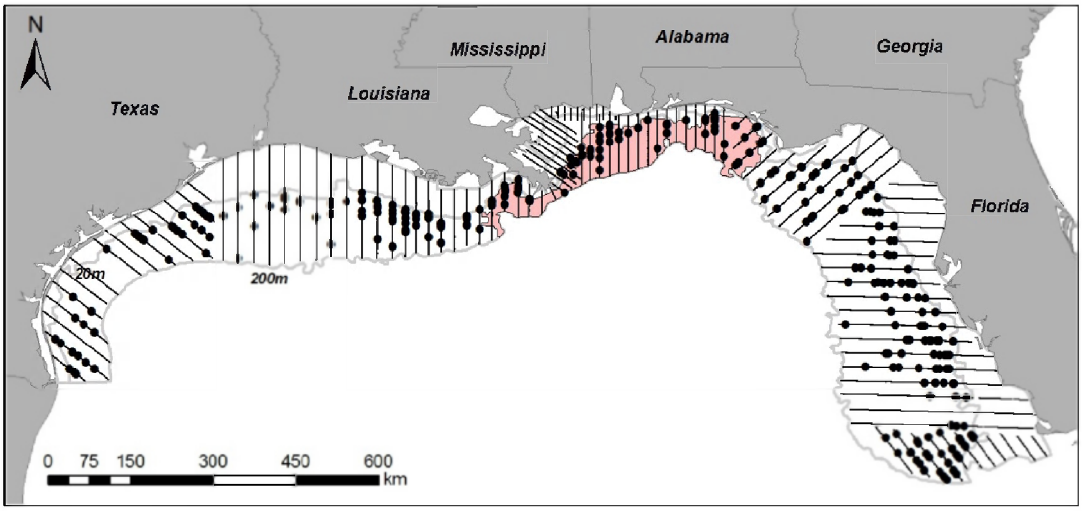


**Figure 20. 2011-2012 aerial survey tracklines (lines) and sightings of Atlantic spotted and bottlenose dolphins (points) within the continental shelf stratum (20-200m). The portion of the oil exposure polygon located over the continental shelf is shown in red. Sightings for the coastal stocks in nearshore waters are not depicted in this figure, see Figure 7.**

### 3.2.2. Oceanic Stocks: 2003, 2004, and 2009 Large Vessel Surveys

Line-transect vessel survey data during large-scale, systematic, visual line-transect surveys conducted during summer 2003, spring 2004, and summer 2009 were used to estimate the abundance of oceanic marine mammal stocks (Table 15, Figure 21). The surveys used during this analysis were conducted aboard the NOAA Ship *Gordon Gunter* and followed similar survey procedures and design. Briefly, each survey was conducted along a “double saw-tooth” survey trackline pattern with tracks oriented to cross roughly perpendicular to bathymetry gradients. Data were collected by a team of three visual observers stationed on the flying bridge of the vessel. Continuous data were recorded on survey effort status and visual conditions (e.g., sea state, swell height, visibility, etc.).

Distance analysis methods were used to estimate the detection probability of marine mammals during each of these three surveys (Dias and Garrison, 2015). The probability of detection was modeled incorporating the effects of covariates on the sighting function. For each sighting, covariates evaluated for the detection model included sea state, swell height, and horizontal visibility. Sequential deletion of terms and Akaike’s Information Criterion (AIC) were used to select the most parsimonious model for the detection function. Separate detection functions were estimated for taxonomic groups that reflect differences in availability and sighting probability including large whales, small whales, dolphins, and cryptic species (beaked whales and pygmy/dwarf sperm whales). These analyses assume that detection probability on the trackline is equal to 1; it is known to be less than 1, therefore the resulting abundance estimates are negatively biased.

Table 15. Vessel based line-transect surveys used in the current analysis and the amount of survey effort within the oil exposure polygon.

| Survey | Dates | Total Effort (km) | Effort in Oil Polygon (km) |
|---|---|---|---|
| GU0302 | 6/14 – 8/17 2003 | 6,752 | 1,111 |
| GU0402 | 4/15 – 6/10 2004 | 6,214 | 992 |
| GU0903 | 6/18 – 8/09 2009 | 4,233 | 425 |

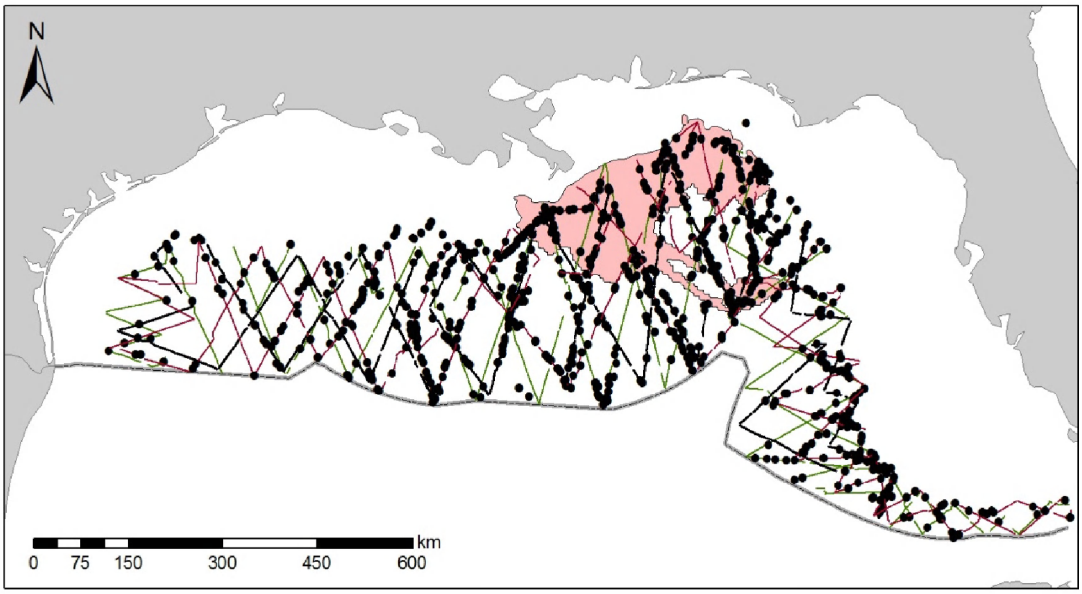


**Figure 21. Survey effort (lines colored by survey year) and cetacean sightings (points) for 2003, 2004, and 2009 vessel-based line-transect surveys used to estimate cetacean density and abundance in oceanic waters. The oil exposure polygon for oceanic waters is shown in red.**

### 3.2.3. Beaked Whales and Pygmy/Dwarf Sperm Whales (*Kogia* spp.): Acoustics Estimates

Both beaked whales (*Ziphius cavirostris and Mesoplodon* spp.*)* and the pygmy/dwarf sperm whales (*Kogia* spp.) are difficult to detect using visual surveys. These species groups are deep divers with long duration dives and are difficult to detect visually when they are at the surface. The available visually-based abundance estimates from the 2003, 2004, and 2009 vessel surveys are therefore severely negatively biased for these species because data are not available to correct for this bias. Passive acoustic monitoring units were deployed during 2010-2011 in the Mississippi Canyon region and analyses were conducted to estimate densities of both beaked whales and *Kogia* spp. based upon detections and identification of echolocation clicks (Hildebrand et al. 2012). The estimated densities from the analyses of passive acoustic data collected in the Mississippi Canyon region were used to estimate abundance within the oil exposure polygon in place of visual estimates for these species. The placement of passive acoustic buoys was non-random, and the estimated densities are uncertain for a number of reasons described by Hildebrand et al. (2012).

### 3.2.4. Bryde's Whales: Additional Large Vessel Surveys

The resident population of Bryde's whales in the Northern Gulf of Mexico occupies a relatively small region off the coast of northwestern Florida. Due to the narrow distribution and small population size, Bryde's whales are rarely observed during any single line transect survey. Therefore, in addition to the surveys referenced above, line transect surveys from summer 2007, summer 2010, fall 2010, and summer 2012 were included. The 2010 surveys and the 2012 survey included dedicated tracklines within the region where Bryde's whales are known to occur (Figure 22).

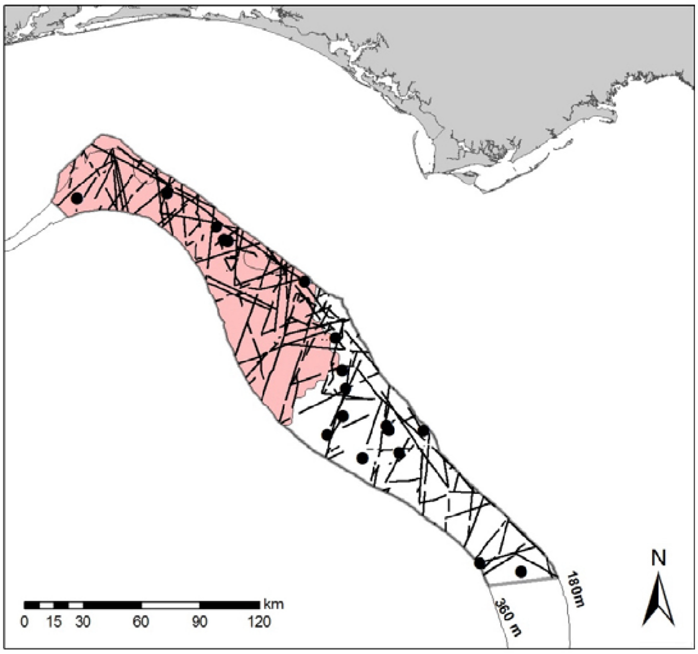


**Figure 22. Line transects (lines) and sightings (points) used to estimate Bryde's whale abundance. The red area represents the overlap between the oil exposure polygon and the Bryde's whale habitat boundary, which lies between the 180 and 360m isobaths south of the Florida panhandle.**

A total of 2,943 km of trackline within the Bryde's whale habitat was covered across these 7 line-transect surveys including 15 on effort sightings comprising a total of 37 individuals. Distance analysis was used to estimate the abundance for the surveyed region (Dias and Garrison 2015). The oil exposure polygon overlapped with 48% of this defined area.

### 3.2.5. Final Estimates of Proportion of Each Stock Exposed to Oil

The proportion of each stock exposed to oil was calculated as the ratio between the abundance estimate within the oil exposure polygon and the estimate for the entire northern Gulf stock. The resulting Gulf-wide stock sizes for each species and the proportion of the population intersecting with the oil exposure polygon are shown in Table 16. Estimates for the northern and western coastal stocks are based upon surveys and abundance estimates discussed in Section 2.3.

**Table 16. Total abundance estimates (in U.S. waters of the northern Gulf of Mexico) and proportion of each stock intersecting with the oil exposure polygons for shelf and oceanic marine mammals.**

| Species | Total Abundance | CV | 95% confidence interval | Proportion of the population in oil polygon |
|---|---|---|---|---|
| Western Coastal Stock Bottlenose Dolphin | 20161 | 0.17 | 14482-28066 | 0.226 |
| Northern Coastal Stock Bottlenose Dolphin | 7185 | 0.21 | 4800-10754 | 0.820 |
| Continental Shelf Dolphins | 63361 | 0.13 | 42898-87417 | 0.125 |
| Bottlenose Dolphin Oceanic | 8467 | 0.36 | 4285-16731 | 0.099 |
| Sperm Whale | 1635 | 0.19 | 1132-2359 | 0.161 |
| Bryde's Whale | 26 | 0.40 | 12-56 | 0.480 |
| Beaked Whales | 1167 | 0.31 | 643-2117 | 0.119 |
| Clymene Dolphin | 3228 | 0.39 | 1558-6691 | 0.071 |
| False Killer Whale | 316 | 0.52 | 121-827 | 0.183 |
| Melon-headed Whale | 1696 | 0.47 | 709-4060 | 0.150 |
| Pantropical Spotted Dolphin | 33382 | 0.14 | 25489-43719 | 0.196 |
| Short-finned Pilot Whale | 1641 | 0.45 | 710-3790 | 0.057 |
| Pygmy Killer Whale | 281 | 0.40 | 131-601 | 0.152 |
| Pygmy/Dwarf Sperm Whale | 6690 | 0.34 | 3482-12857 | 0.151 |
| Risso's Dolphin | 1848 | 0.26 | 1123-3041 | 0.076 |
| Rough-toothed Dolphin | 2414 | 0.49 | 964-6040 | 0.409 |
| Spinner Dolphin | 6621 | 0.35 | 3386-12947 | 0.465 |
| Striped Dolphin | 2605 | 0.27 | 1537-4415 | 0.132 |

## 3.3. Mortality, Reproductive, and Adverse Health Effects Metrics

Injury metrics for offshore species were calculated based upon the mortality, adverse health effects and reduction in reproductive success observed for bottlenose dolphins in Barataria Bay (Table 12).

### 3.3.1. Excess Proportion of Population Killed

The Barataria Bay bottlenose dolphin population experienced an estimated 35% increase in mortality in the three years after the spill (Table 12). However, the entire Barataria Bay population was exposed to DWH oil, while only a portion of the populations of the shelf and oceanic stocks was exposed. Therefore, the proportion of the population killed was calculated by multiplying the proportion of the population exposed (Table 16) by 0.35. Uncertainty in this parameter was calculated by multiplying the upper and lower confidence limits by the proportion of the population exposed (Table 14).

### 3.3.2. Excess Proportion of Females with Adverse Health Effects

Observations of bottlenose dolphins in Barataria Bay during health assessment captures in 2011 indicated an increase in the number of animals in poor health condition relative to an unexposed population in Sarasota Bay (Schwacke et al. 2014). The relative increase in animals with a guarded or worse prognosis based upon veterinary assessment of captured individuals was estimated to be 0.37 (Table 12). As with the proportion killed metric, this proportion was multiplied by the proportion of the population exposed (Table 16) to estimate the proportion of animals with adverse health impacts for oceanic and shelf stocks, and uncertainty was calculated based only on the uncertainty in the estimate of the relative increase in adverse health effects.

### 3.3.3. Excess Proportion of Population with Reproductive Failure

Barataria Bay dolphins also exhibited an estimated 46% increase in the number of reproductive failures for pregnant females relative to reference populations (Table 12). This proportion was applied to the exposed portion of the shelf and oceanic stocks (Table 16) to estimate a metric of reproductive failure, and uncertainty was calculated based only on the uncertainty in the estimate of the proportion of the population with reproductive failure.

## 3.4. Population Model for Estimating Long-term Population Injury

### 3.4.1. Methods and Sources of Input Values

The population level effects of reduced survivorship, adverse health effects and reproductive success associated with exposure to DWH oil were modeled using a stage-, sex- and class-structured matrix model for each stock. These models are similar to the age-, sex- and class-structured model applied to estuarine and coastal bottlenose dolphins (Section 2.7.2); however, the structure was simplified by grouping ages into life-history stages. Unlike bottlenose dolphins, there is little information available on the demographic structure and population dynamics of the offshore species, particularly for populations within the Gulf of Mexico. The available information for these species from other regions are summarized primarily by life-history stage (e.g., juvenile, adult, etc.), rather than by age categories, and hence it is more appropriate to derived stage-based models using these parameters. The structure of these models divided the life-cycle of female animals into five stages: dependent calf, juvenile, reproductively mature, mother with calf, and post-calving female. This structure is similar to models recently derived for sperm whales in the Gulf of Mexico (Chiquet et al. 2013) and those applied to North Atlantic right whales (Fujiwara and Caswell 2001) among other large vertebrate species. The "post-calving female" stage reflects a period of time after calving when females are not available to reproduce. The durations of the mother with calf period (associated with nursing a dependent calf) and the post-calving interval combine to determine the inter-birth interval. The models also included a male compartment consisting of calves, immature individuals, and mature individuals. As with the age-structured models for bottlenose dolphins, there are "exposed" and "unexposed" classes in the model, and in the case of the oceanic and shelf species, only a portion of the population is considered to be exposed. Animals in both the exposed and unexposed classes contribute calves to the unexposed class in the model (Figure 23). Density dependent fecundity was implemented in the same manner as that described for the bottlenose dolphin model (Section 2.7.2) with the scale parameter, **ρ**, set to the mean of the distribution used for bottlenose dolphins.

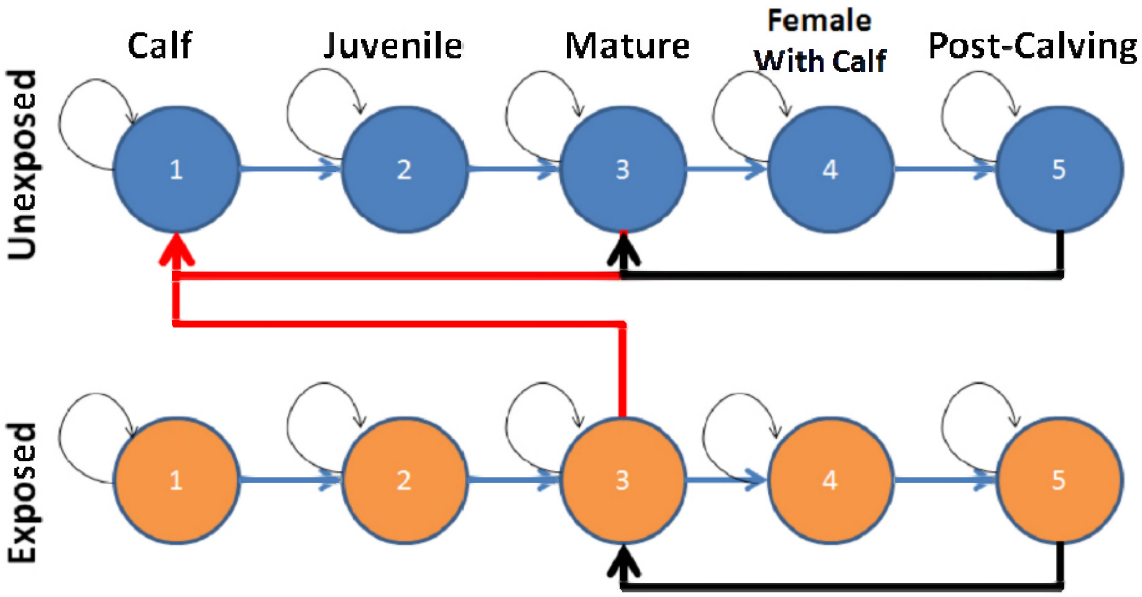


**Figure 23. Multi-class stage-structured population model used for oceanic cetacean stocks. Transitions between stages are indicated by arrows with red arrows indicating the production of calves by mature females. Only the female component of the model is depicted for all life stages.**

Bottlenose dolphins inhabit nearshore waters and as such are more accessible; therefore, their life history has been studied extensively relative to other shelf and oceanic cetacean species. While the relatively rich literature on bottlenose dolphin life history parameters allowed for the quantification of uncertainty in population model input parameter values, there was insufficient information on shelf and oceanic species to construct informed input parameter distributions. Therefore, unlike the bottlenose dolphin model, only a single model scenario was run using point estimates for input parameter values, and simulations have not been conducted to explore the effects of uncertainty in the model parameters.

The key demographic parameters for these models are: minimum inter-birth interval, age at sexual maturity, time to weaning/calf dependency, and stage-specific mortality rates. Survival rates were assumed to be similar for all "adult" life history stages. Input parameters were selected based upon an extensive review of available literature on marine mammal demographic parameters. These literature sources included both reviews and individual papers targeted at particular species (Ridgway and Harrison 1994, Taylor et al. 2007, Jefferson and Hung 2008). Species were grouped since, in most cases, detailed demographic information is available for only a subset of species within a given genus or other taxonomic grouping (Table 17).

**Table 17. Input model parameters by species group for stage structured models**

| Species (Group) | Min. Inter-birth Interval | Age at Weaning | Age at Sexual Maturity | Calf Survival Rate | Juvenile Survival Rate | Adult Survival Rate |
|---|---|---|---|---|---|---|
| Sperm Whale[a] | 3 | 2 | 9 | 0.91 | 0.94 | 0.98 |
| Bryde's Whale | 2 | 2 | 10 | 0.90 | 0.95 | 0.97 |
| Pygmy/Dwarf sperm whale | 1.5 | 1 | 6 | 0.85 | 0.90 | 0.95 |
| Stenellid dolphins[b] | 2.5 | 2 | 10 | 0.85 | 0.95 | 0.97 |
| Large dolphins[c] | 2 | 2 | 9 | 0.85 | 0.97 | 0.97 |
| Small whales[d] | 3 | 3 | 11 | 0.93 | 0.95 | 0.97 |
| Beaked whales[e] | 3 | 3 | 9 | 0.90 | 0.95 | 0.97 |

[a] sperm whale parameters based on Chiquet et al. 2013. For males, an added loss term was included in the adult survival rate to account for emigration of adult males and a resulting 0.72/0.28 female to male sex ratio observed in the Gulf of Mexico population (Engelhaupt et al. 2009)
[b] includes *Stenella attenuata*, *S. clymene*, *S. longirostris*, and *S. coeruleoalba*
[c] includes *Tursiops truncatus*, *Stenella frontalis*, *Grampus griseus*, and *Steno bredanensis*
[d] includes *Globicephala macrorhynchus*, *Peponocephala electra*, *Pseudorca crassidens* and *Feresa attenuata*
[e] includes *Mesoplodon* spp. and *Ziphius cavirostris*

With the exception of sperm whales, there is little information on survival rates for most species, and no direct information on survival rates specifically for the northern Gulf of Mexico. Our review of available literature suggested that calf survival rates range from 0.80 to 0.93 across multiple species. The survival rates for adults were typically higher than those for calves and juveniles and ranged from 0.95 to 0.98 for most species. Based upon the available information, default values for the current models were chosen as 0.90, 0.95, and 0.97 for calf, juvenile, and adult survival, respectively. In some species groups, positive population growth could not occur with these defaults even at the minimum inter-birth interval, so survival rates were adjusted upward with a maximum of 0.97. In the case of the *Kogia* spp., which are both short-lived and have high reproductive rates, survival rates were adjusted downward from these defaults (see Table 17).

The models were implemented with the assumption that the population sizes were constant (*i.e.*, zero population growth rate) prior to the spill. By using an average of surveys from 2003-2009 as the estimates of initial population sizes, we are implicitly assuming that there were no population changes over that time period. Therefore, following the selection of baseline survival parameters, the inter-birth interval was tuned to achieve zero population growth at the pre-spill population size (Section 3.2, Table 16). This value for inter-birth interval is the "nominal" value where one is assuming that reproduction and mortality are balanced to achieve a zero growth population. Due to the density dependent fecundity function, when the population drops below the baseline population size, the inter-birth interval will decrease (with a lower bound set by the input minimum inter-birth interval) which allows the population growth rate to increase so the population will recover to the baseline.

The starting conditions for the models are the baseline population size (species specific) with the exposed cohort size being estimated as the proportion of the population exposed to oil. The proportion of the population in each stage is solved for within the model structure, and a 50:50 sex ratio is assumed (except for sperm whales, see footnote in Table 17). To simulate the impacts of oil exposure, the exposed cohort experiences a 12% decrease in survival annually for 2010-2014 then for 10 years, this extra mortality decreases linearly. To simulate increased negative reproductive effects, the exposed cohort experiences a 45.5% annual decrease in calf production for 2010-2014, then this reduction decreases linearly for 15 years. These patterns reflect the observed reductions in survival and reproduction quantified in Barataria Bay and the "lingering effects" and time to recovery based on the elicitation of expert opinion (Section 2.7.3). The population models were projected for 150 years. Output metrics of injury are:

- **Years to recovery (YTR)** - the number of years until the DWH-injured population trajectory reaches 95% of the baseline population trajectory. Note that if the population is not decreased by at least 5%, then YTR will not be applicable.
- **Lost cetacean years (LCY)** – difference between the baseline and DWH-injured population size, summed over all 150 years
- **Maximum proportional decrease** – the difference between the two population trajectories when the DWH-injured trajectory is at its lowest point

### 3.4.2. Population Model Results

Outputs from the population models for each species are shown in Table 18 and example population trajectories are in Figure 24. The proportional decrease in population sizes ranged from 0.03 – 0.23 with the highest decreases being experienced by Bryde's whale (0.22) and spinner dolphins (0.23). Both of these species, along with rough-toothed dolphins, had over 40% of their population overlapping with oil, and thus there was the greatest proportional impact. In the case of species with very large population sizes (e.g., shelf dolphins and pantropical spotted dolphins), the number of lost cetacean years was very large even though the proportional decrease in population size was relatively small. This reflects the combined effects of the number of animals and the time to recovery that contribute to the lost cetacean years metric. Smaller populations (e.g., Bryde's whales) will have low numbers of lost cetacean years even with greater proportional declines in population sizes.

**Table 18. Injury metrics output from population models for shelf and oceanic stocks. NA for Years to Recovery (YTR) occurred when the model determined that a stock did not decline by more than 5%. Lost cetacean years are species specific and cannot be combined**

| Stock | Lost Cetacean Years (LCY) | Maximum Proportional Change in Population Size | Years to Recovery (YTR) |
|---|---|---|---|
| Shelf Dolphins | 359,996 | -0.03 | NA |
| Bottlenose Dolphin, Oceanic | 37,668 | -0.04 | NA |
| Sperm Whale | 13,197 | -0.07 | 21 |
| Bryde's Whale | 705 | -0.22 | 69 |
| Beaked Whales | 7,838 | -0.06 | 10 |
| Clymene Dolphin | 12,167 | -0.03 | NA |
| False Killer Whale | 3,422 | -0.09 | 42 |
| Melon-headed Whale | 14,887 | -0.07 | 29 |
| Pantropical Spotted Dolphin | 363,780 | -0.09 | 39 |
| Short-finned Pilot Whale | 5,304 | -0.03 | NA |
| Pygmy Killer Whale | 2,501 | -0.07 | 29 |
| Pygmy/Dwarf Sperm Whale | 49,100 | -0.06 | 11 |
| Risso's Dolphin | 6,258 | -0.03 | NA |
| Rough-toothed Dolphin | 50,464 | -0.17 | 54 |
| Spinner Dolphin | 188,713 | -0.23 | 105 |
| Striped Dolphin | 18,647 | -0.06 | 14 |

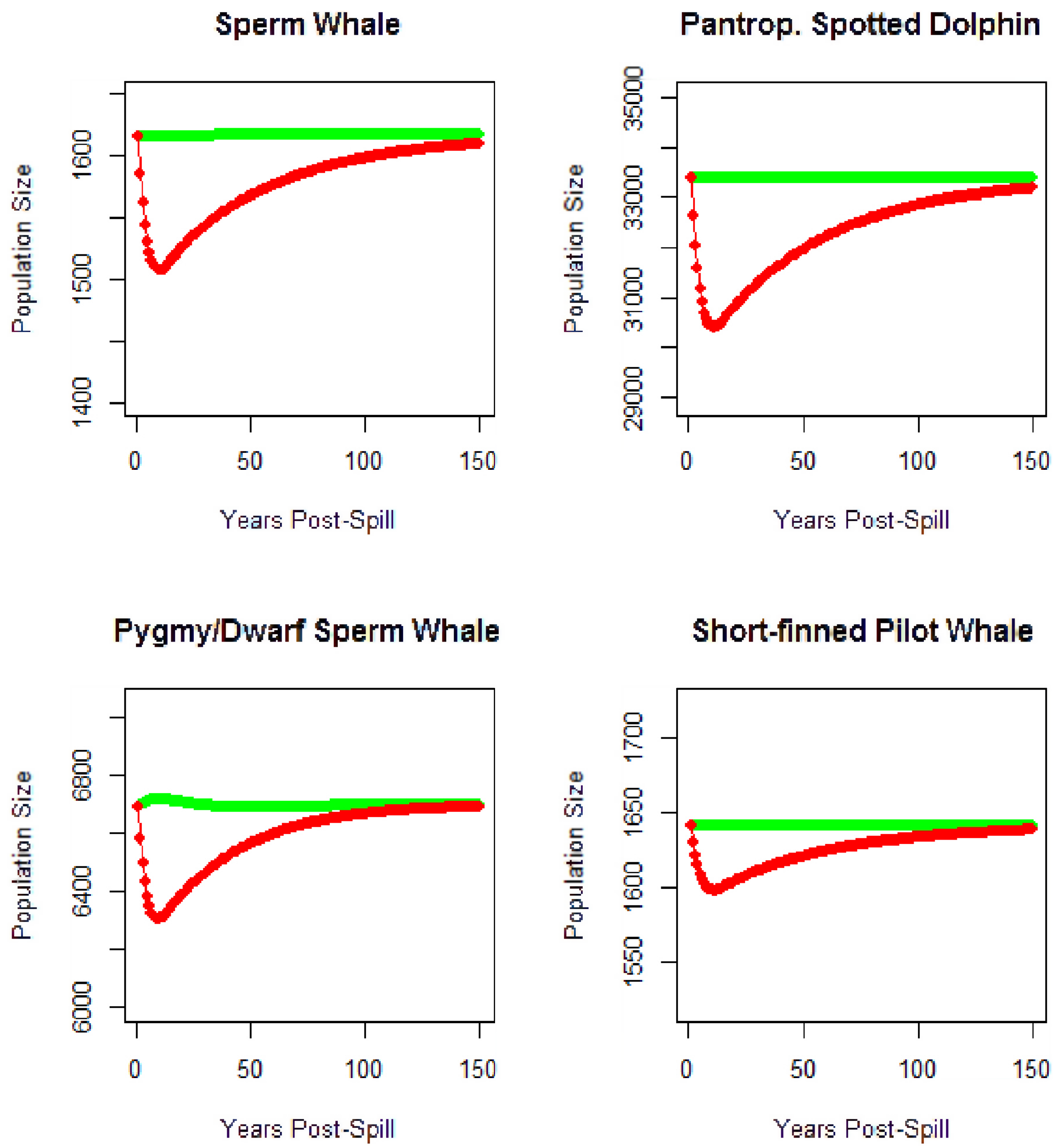


**Figure 24. Example population model outputs. The green line is the baseline population trajectory (assuming no change in survival or reproductive rates) and the red line shows the trajectory of the population after exposure to DWH oil.**

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