

# Spatial, temporal and demographic patterns of cetacean strandings in the northcentral Gulf of Mexico

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## ABSTRACT

There are few published reports that summarise long-term stranding data for cetaceans in the Gulf of Mexico (GoM). Cetacean stranding data from Alabama (AL) in the northcentral GoM have been recorded since 1978, with dedicated stranding response beginning in 1987, providing an opportunity to characterise spatial, temporal and demographic stranding patterns for this unique area, which links the eastern and western northern GoM coastline. To elucidate long-term patterns, we analysed stranding records in AL from 1978–2018, taking into consideration the effects of periodic unusual mortality events (UMEs) on stranding demographics. During this 41-year period, 774 stranding events were recorded involving at least 13 different cetacean species. Common bottlenose dolphins (*Tursiops truncatus*) were the most reported stranded cetacean ( $n = 692$ , 89.4%), followed by *Stenella* spp. ( $n = 31$ , 4.0%). Spatial analysis revealed three stranding hotspots for bottlenose dolphins and an additional hotspot for non-bottlenose dolphin species that may represent a geographic influence in the northcentral GoM for strandings of offshore cetaceans. Because strandings were dominated by bottlenose dolphins, additional demographic analyses were possible for this species. Strandings occurred most often in the spring (March–May), and females and fetuses stranded more often in calving (January–April) than non-calving season, consistent with increased mortality related to birthing or reproductive-associated events. Subadults were found stranded more often than other size classes, indicating a potential stranding bias for this age class. These data are critical to guide future stranding response efforts, define vulnerable populations and help understand causes of mortality through time in the northcentral GoM. Our findings also highlight the importance of long-term, consistent stranding response to enable identifying spatial and temporal trends that are necessary to support meaningful, range-wide management and conservation.

**KEYWORDS:** STRANDING TRENDS; BOTTLENOSE DOLPHIN; UNUSUAL MORTALITY EVENTS; GULF OF MEXICO; SEASONALITY

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## INTRODUCTION

Cetacean stranding data are often the only cost-effective resource to provide insight into the ecology, biology, health and population demographics of these highly mobile and often elusive marine mammals (e.g., Evans and Hammond, 2004; McGovern *et al.*, 2016; IJsseldijk *et al.*, 2020; Coombs *et al.*, 2019). Hands-on research of live marine mammals is expensive, dangerous and there is a limited amount of invasive work that can be conducted humanely on most live, non-stranded marine mammals (Wells and Scott, 1990; Hodgson *et al.*, 2013; Schwacke *et al.*, 2014). Stranding datasets provide biologically important information, including life history and mortality

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demographics that may indicate the health of the greater aquatic ecosystem (e.g. Pitchford *et al.*, 2013; Peltier *et al.*, 2013; Liu *et al.*, 2018; Coombs *et al.*, 2019). Knowledge of historical spatial and temporal patterns of stranded marine mammals in an area is also important to plan location and timing of public awareness and outreach efforts for stranding networks (Olson *et al.*, 2020). Basic stranding data (e.g. species and location) are critical for setting baselines of cetacean mortality that are needed to define unusual mortality events (UMEs, as defined in the Marine Mammal Protection Act 16 U.S.C. 1421b) and identify stranding hotspots for enforcement and management. Although UMEs tend to be well-studied, the spatial, temporal and demographic attributes of strandings outside UMEs are less comprehensively documented, analysed and published. These baseline, non-UME data are important for identifying vulnerable population groups and informing management decisions (Ijsseldijk *et al.*, 2020).

Stranding networks exist worldwide to document and respond to live and dead marine mammal strandings (Hoffman, 1991). Although some stranding data are published for areas world-wide (e.g., McFee *et al.*, 2006, Auge *et al.*, 2018 and Ijsseldijk *et al.*, 2020), these data are lacking for the northcentral Gulf of Mexico (GoM) in the Southeastern United States of America (USA), an important location that links the eastern and western portions of the northern GoM. The waters off the coast of Alabama (AL) are considered a biogeographical suture-zone and a complex geological as well as environmental region for multiple species, including marine mammals (Portnoy and Gold, 2012; Vollmer and Rosel, 2017). A wide variety of taxa exist in the region, with differences in species composition, abundance and genetic structure across the suture-zone (Drymon *et al.*, 2012, Portnoy and Gold, 2012, Hollenbeck *et al.*, 2019). These differences are potentially due to the proximity to major sources of freshwater discharge and other physical environmental attributes, particularly related to temperature and salinity, which create zones of distinct species and subpopulations (Portnoy and Gold, 2012, Hollenbeck *et al.*, 2019). This area is also a hypothesised biogeographic break with little genetic overlap between bottlenose dolphin populations (Vollmer and Rosel, 2017; Vollmer *et al.*, 2021). There are 21 species of cetaceans that live within the northern GoM, including two species of large whales [Rice's (*Balaenoptera ricei*) and sperm (*Physeter macrocephalus*) whales] listed as endangered under the Endangered Species Act and many elusive species such as Gervais' beaked (*Mesoplodon europaeus*) and pygmy sperm (*Kogia breviceps*) whales (Maze-Foley and Mullin, 2006; Darnell 2015; Wursig 2017; Rosel *et al.*, 2021).

Cetacean strandings have been recorded in the northern GoM since the late 1970s, with multiple stressors known to affect these cetacean populations, including freshwater exposure, infectious diseases, negative human interactions, harmful algal blooms and oil exposure following the *Deepwater Horizon* oil spill, among others (e.g., Carmichael *et al.*, 2012, Wilkin *et al.*, 2017; Balmer *et al.*, 2018; Collins *et al.*, 2019; Deming *et al.*, 2020). From 1978 to 2018, there were three UMEs involving common bottlenose dolphins (*Tursiops truncatus*; herein referred to as bottlenose dolphins) in the northcentral GoM that included AL waters, primarily attributed to infectious disease and environmental effects (e.g., Carmichael *et al.*, 2012; Litz *et al.*, 2014; Colegrove *et al.*, 2016). While the cetacean stranding data collected during these UMEs have been extensively studied, there have been no published findings of basic historical stranding demographics in AL waters within the northcentral GoM to put these UMEs into the broader context of local or regional cetacean mortality.

The purpose of this study is to contribute to the limited body of knowledge on trends in strandings of cetaceans globally by analysing more than four decades (1978–2018) of stranding data from coastal AL in the northcentral GoM. We define spatial, temporal, and demographic patterns of cetacean strandings in this region. These analyses will establish baselines necessary to enhance population management, identify vulnerable population groups and analyse future stranding data in the GoM, including UME investigations.

## METHODS

### Study setting

Alabama is situated along the northcentral GoM coast (Fig. 1) and has 75km of coastal shoreline with another 490km of shoreline within the Mobile Bay estuary, one of the largest estuaries in the USA (Alexander *et al.*, 2001). The shoreline is made up of sandy beaches, tidal marshes and developed shorefront property. The study area

encompasses southern AL, including Mississippi Sound (MSS), Mobile Bay (MB), Perdido Bay (PB) and its connected waterways (Wolf Bay, Bayou Saint John, Cotton Bayou and others) and nearshore waters of the GoM (Fig. 1). The average air temperature range for the area is 8–31°C with temperatures rarely falling below 1°C or exceeding 33°C (Christmas *et al.*, 1973). The watersheds that discharge to the AL coast typically convey the highest volume of freshwater from February to May when the areas that drain into the watershed experience snow and ice melt from the central USA (Schroeder *et al.*, 1990). As a result, MB is one of the largest freshwater drainage areas in the USA (Alexander *et al.*, 2001). Due to this influx of freshwater, there are drastic salinity variations seasonally and annually in MB and adjacent AL waters (Schroeder *et al.*, 1990; Carmichael *et al.*, 2012).

## Stranding response and data collection

Cetacean strandings along the AL coast have been recorded since 1978, with regular response beginning in 1987. Individuals from area agencies and organisations responded to strandings in the early years of stranding response (1978–1987) with little consistency. Following this period, stranding agreements with the National Oceanic and Atmospheric Administration National Marine Fisheries Service (NOAA/NMFS) for response in AL were held by a series of agencies in AL and/or Mississippi (MS) until 2011 when the Dauphin Island Sea Lab (DISL) officially formed the Alabama Marine Mammal Stranding Network (ALMMSN) and began continuous, dedicated response to marine mammal strandings in the state.

In the USA, standardised marine mammal stranding data, including location, date, species, sex, total length and other information are provided to the NOAA/NMFS Marine Mammal Health and Stranding Response Program (MMHSRP) National Database within 30 days of data collection. For this study, data for cetaceans stranded in AL from 1978–2018 were obtained from the Database, with data prior to 1991 obtained by a data request to the MMHSRP manager and all other data downloaded from the online database on 7 March 2019. Repeat entries and unconfirmed strandings were not included in analyses.

## Spatial analyses

Cetacean stranding locations from 1978 to 2018 were mapped and analysed using ArcGIS version 10.6 (ESRI, Redlands, CA, 2020). Location data with unknown coordinates were excluded or estimated based on descriptions of locations when available. The Kernel Density tool from the Density toolbox was used to execute the kernel density function for point features (Silverman, 1986). Bottlenose dolphin strandings were separated from other known cetacean species, and stranding densities were analysed with a search radius of 4km and an output cell size of 50m, similar to McFee and Burdett (2007). Non-bottlenose dolphin species strandings were examined using a search radius of 6 km and a cell output size of 25m, due to the smaller number of overall strandings of these species. The density of the point features around each output cell was totaled and divided by the area of the cell, with each point datum weighted the same throughout the cell. The output values were sorted into four classes using Jenks natural breaks optimisation to display areas where cetacean strandings occurred in the highest densities, or hotspots (Wilkinson and Friendly, 2009; Augie *et al.*, 2018). As a result, these density analyses underestimate actual stranding numbers and show stranding patterns as the relative likelihood of finding a stranded cetacean in a given location, not exact densities of past strandings (Wilkinson and Friendly, 2009; Augie *et al.*, 2018).

## Temporal analyses

We analysed the total number of strandings annually for all species and seasonally for bottlenose dolphins. Outliers to mean annual strandings through time were detected as standardised residuals  $> 2$  or  $< -2$ , with the level of sensitivity set to be consistent with the NOAA/NMFS definition of UMEs. Traditional seasons were defined as Spring (March–May), Summer (June–August), Fall (September–November) and Winter (December–February) as in other marine mammal stranding publications in the GoM including Hubbard (2004), Miller and Baltz (2010) and Pitchford *et al.* (2013). We also examined stranding data according to calving (or birthing) season, a biologically relevant time of the year that may cause an increase in strandings. Calving season was defined as January–April, the time of year when most inshore bottlenose dolphin populations in this region give birth to their offspring

(Fernandez and Hohn, 1997; Venn-Watson *et al.*, 2015; Colegrove *et al.*, 2016) and was considered for statistical analyses in this study. Welch's Two Sample t-test was used to compare calving and non-calving season for both sexes. Pearson's chi-squared test was used to analyse frequencies of calving and non-calving season strandings of female bottlenose dolphins in size classes defined below.

### Demographic analyses of bottlenose dolphin strandings

Because the majority of strandings were bottlenose dolphins, further demographic and statistical analyses were limited to this species. Data for bottlenose dolphins were further separated by sex and size classes to assess demographic patterns. Data on sex (male or female) for bottlenose dolphins were available for the entire study period. Welch's Two Sample t-tests were used to compare the number of strandings per year between females and males. Data on size classes for bottlenose dolphin males and females in AL were available from 2008 and 2009, respectively, to 2018. Size classes for this study were defined from the total length of intact carcasses, categorised based on size at estimated-age class (Wells *et al.*, 1987; Hohn, 1989; Mattson *et al.*, 2006), including < 95cm (foetus), 95–115cm (perinate), 116–169cm (calf), 170–230cm (subadult female), 170–250cm (subadult male), > 230cm (adult female), > 250cm (adult male). A Kruskal-Wallis rank sum test was used to compare the average number of strandings per year among size classes, and post-hoc analysis was performed with Dunn's Test with Bonferroni corrections to account for multiple comparisons. All data analyses were performed using R version 4.0.2 (R Core Team, 2020).

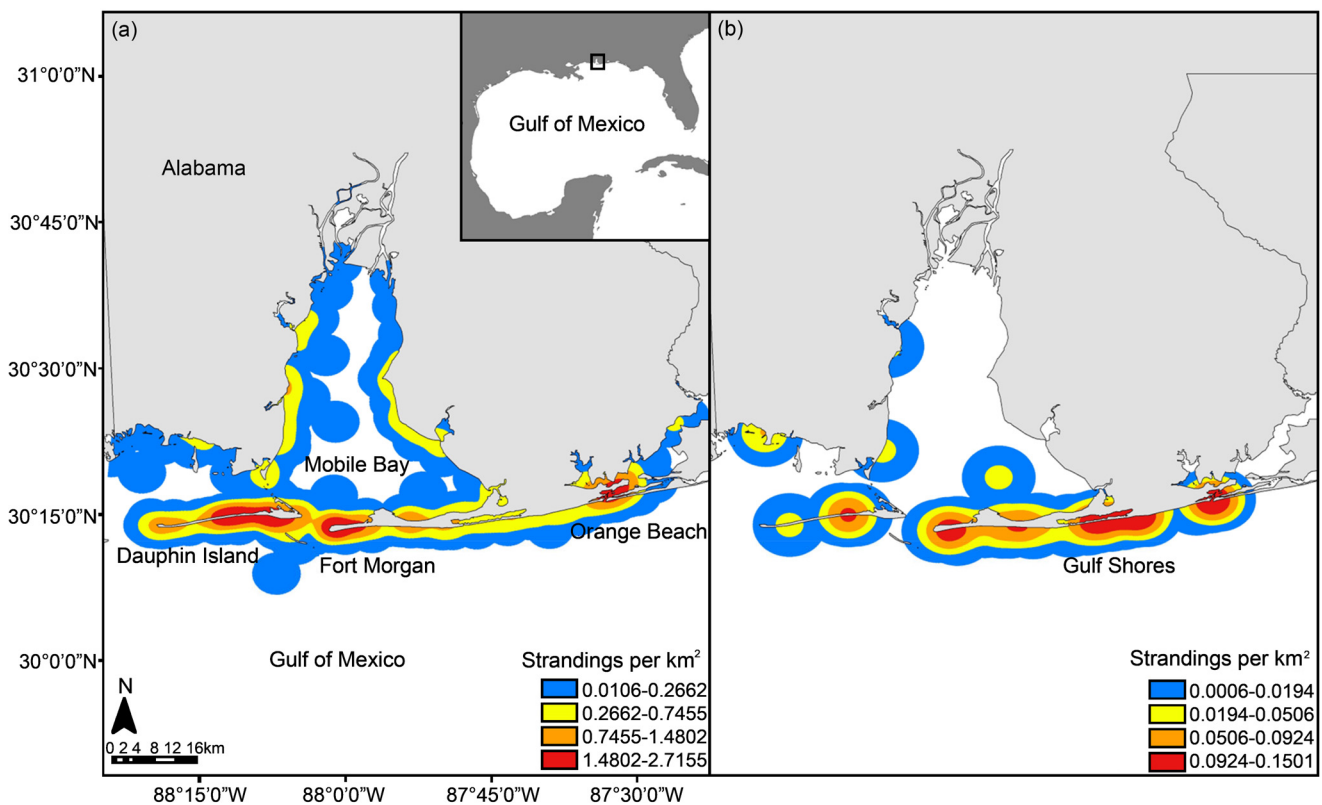


Fig. 1. Kernel density estimation of stranding density along the Alabama coast for (a) *Tursiops truncatus* and (b) all other known species from 1978 to 2018 (note the difference in magnitude of scaling between panels to account for the lower numbers of other known species stranded, shown in panel b).

## RESULTS

A total of 774 reported cetacean strandings occurred in Alabama from 1978 to 2018 (Table 1). Thirteen different cetacean species were documented during this time, of which the majority were bottlenose dolphins ( $n = 692$ , 89.4%), followed by *Stenella* spp. ( $n = 31$ , 4.0%). All recorded cetaceans that stranded in AL during this period

were members of the suborder Odontoceti (toothed whales), and no Mysticeti (baleen whale) strandings are known. More male cetacean strandings were recorded ( $n = 350$ , 45.0%) than female ( $n = 271$ , 35.0%) and unknown sex ( $n = 153$ , 20.0%). The majority of strandings recorded were dead ( $n = 720$ , 93.0%). Known non-bottlenose dolphin species were more often stranded live ( $n = 14$ , 32.6%) than bottlenose dolphins ( $n = 20$ , 2.9%). No mass strandings, defined as two or more non-mother/calf pairs, occurred.

## Spatial analyses

Cetacean strandings occurred along the full coastline of AL from 1978 to 2018. Bottlenose dolphin strandings occurred in highest numbers at three locations during the 41 years of this study, including (from west to east) Dauphin Island, Fort Morgan and Orange Beach, AL (Fig. 1a). All other known, non-bottlenose dolphin species stranded in similar locations to bottlenose dolphins along the southern AL coast, with hotspots in Fort Morgan and Orange beach, but with an additional hotspot on the GoM side of Gulf Shores (Fig. 1b). Bottlenose dolphin strandings occurred along both the eastern ( $n = 380$ , 56.0%) and western shoreline of AL ( $n = 298$ , 44.0%). Bottlenose dolphins were less likely to strand inside MB ( $n = 240$ , 35.4%) than outside MB ( $n = 438$ , 64.6%). Non-bottlenose dolphin species stranded three times more often on the eastern shoreline ( $n = 31$ , 77.5%) of AL than on the western side ( $n = 9$ , 22.5%). These species also stranded more often outside MB ( $n = 38$ , 92.3%) than inside MB ( $n = 3$ , 7.7%).

## Temporal analyses

Stranding counts varied through time during this study, with a greater number of unknown species reported in recent years (Table 1, Fig. 2). The mean ( $\pm$  SD) annual stranding count for all years was  $19 \pm 16$  (range 0–61). Three years were statistical outliers and had higher than usual stranding numbers, including 1990 ( $n = 61$ ), 1993 ( $n = 48$ ), and 2011 ( $n = 59$ ), which correspond with UMEs officially declared by NOAA/NMFS for bottlenose dolphins in the northern GoM (Lipscomb *et al.*, 1996; McFee and Lipscomb, 2009; Litz *et al.*, 2014). When these outlier years were excluded, the mean annual stranding count for all remaining years was  $16 \pm 12$  (range 0–42), increasing to  $34 \pm 6$  (range 25–42) during recent years (2012–2018 alone). To detect and account for potential UME-specific effects on stranding demographics, subsequent analyses were performed with and without the outlier years.

Bottlenose dolphin strandings with known stranding date ( $n = 691$ ) occurred most often in March ( $n = 137$ , 19.8%) and least often in June ( $n = 26$ , 3.8%). Strandings were highest in spring (March–May) and winter (December–February), followed by fall (September–November) and summer (June–August) (Fig. 4). When all years were analysed, the number of strandings per year was not statistically different in the calving (January–

Table 1  
Total number of cetaceans stranded in Alabama waters from 1978–2018, separated by species, sex, and stranding condition.

| Species                        | Total      | Sex        |            |            | Stranding Condition |            |           |
|--------------------------------|------------|------------|------------|------------|---------------------|------------|-----------|
|                                |            | Male       | Female     | Unknown    | Live                | Dead       | Unknown   |
| <i>Tursiops truncatus</i>      | 692        | 321        | 262        | 109        | 20                  | 664        | 8         |
| Unknown spp.                   | 39         | 3          | 0          | 36         | 4                   | 27         | 8         |
| <i>Stenella frontalis</i>      | 13         | 8          | 4          | 1          | 3                   | 10         | 0         |
| <i>Stenella longirostris</i>   | 7          | 5          | 1          | 1          | 2                   | 5          | 0         |
| <i>Stenella coeruleoalba</i>   | 4          | 2          | 1          | 1          | 0                   | 4          | 0         |
| <i>Stenella</i> spp.           | 4          | 1          | 1          | 2          | 1                   | 3          | 0         |
| <i>Peponocephala electra</i>   | 4          | 2          | 1          | 1          | 3                   | 1          | 0         |
| <i>Stenella attenuata</i>      | 3          | 2          | 1          | 0          | 2                   | 1          | 0         |
| <i>Kogia breviceps</i>         | 1          | 1          | 0          | 0          | 1                   | 0          | 0         |
| <i>Kogia</i> spp.              | 1          | 0          | 0          | 1          | 0                   | 1          | 0         |
| <i>Pseudorca crassidens</i>    | 1          | 1          | 0          | 0          | 0                   | 1          | 0         |
| <i>Physeter macrocephalus</i>  | 1          | 0          | 0          | 1          | 0                   | 1          | 0         |
| <i>Mesoplodon europaeus</i>    | 1          | 1          | 0          | 0          | 0                   | 1          | 0         |
| <i>Feresa attenuata</i>        | 1          | 1          | 0          | 0          | 1                   | 0          | 0         |
| <i>Steno bredanensis</i>       | 1          | 1          | 0          | 0          | 1                   | 0          | 0         |
| <i>Mesoplodon densirostris</i> | 1          | 1          | 0          | 0          | 0                   | 1          | 0         |
| <b>Total strandings</b>        | <b>774</b> | <b>350</b> | <b>271</b> | <b>153</b> | <b>38</b>           | <b>720</b> | <b>16</b> |



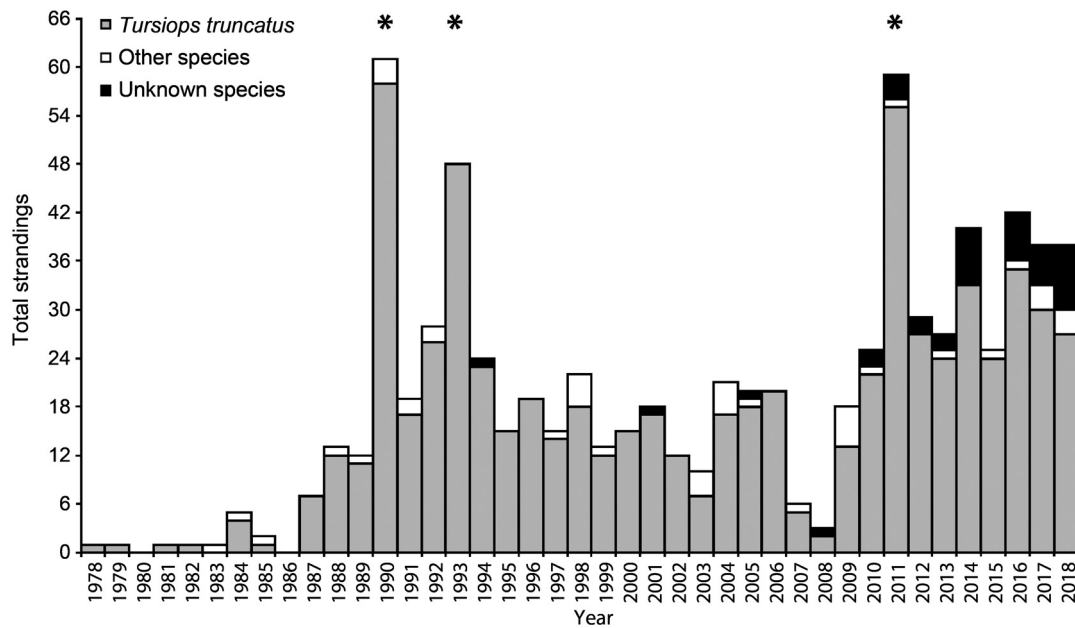


Fig. 2. Cetacean stranding counts per year in Alabama from 1978–2018, including bottlenose dolphin (*Tursiops truncatus*) and other known and unknown species. \* = statistical outliers corresponding to Unusual Mortality Events that occurred in the northern Gulf of Mexico.

April) compared to non-calving season for females (Welch’s t-test,  $t = 1.96$ ,  $df = 73.27$ ,  $p = 0.05$ ) or males ( $t = 0.18$ ,  $df = 75.86$ ,  $p = 0.85$ ) (Fig. 5a). When outlier years were excluded, however, females were more likely to strand during calving season ( $t = 2.05$ ,  $df = 70.49$ ,  $p = 0.04$ ), while males were not ( $t = 0.95$ ,  $df = 73.99$ ,  $p = 0.35$ ). When female strandings were separated by size class, fetuses stranded significantly more often during the calving season than non-calving season (2009–2018: Pearson’s Chi-squared test,  $\chi^2 = 27.42$ ,  $df = 4$ ,  $p < 0.001$ ; Chi-squared Post hoc test,  $p = 0.001$ ; Fig. 5b), regardless of outliers (2009–2018, excluding 2011: Pearson’s Chi-squared test,  $\chi^2 = 17.43$ ,  $df = 4$ ,  $p = 0.002$ ; Chi-squared Post hoc test,  $p = 0.02$ ).

### Demographic and statistical analyses of bottlenose dolphin strandings

The number of bottlenose dolphins that stranded per year did not differ between sexes for the study period (1978–2018: Welch’s t-test,  $t = 1.03$ ,  $df = 68.30$ ,  $p = 0.31$ ; Fig. 3a) or when outlier years were excluded (Welch’s t-test,

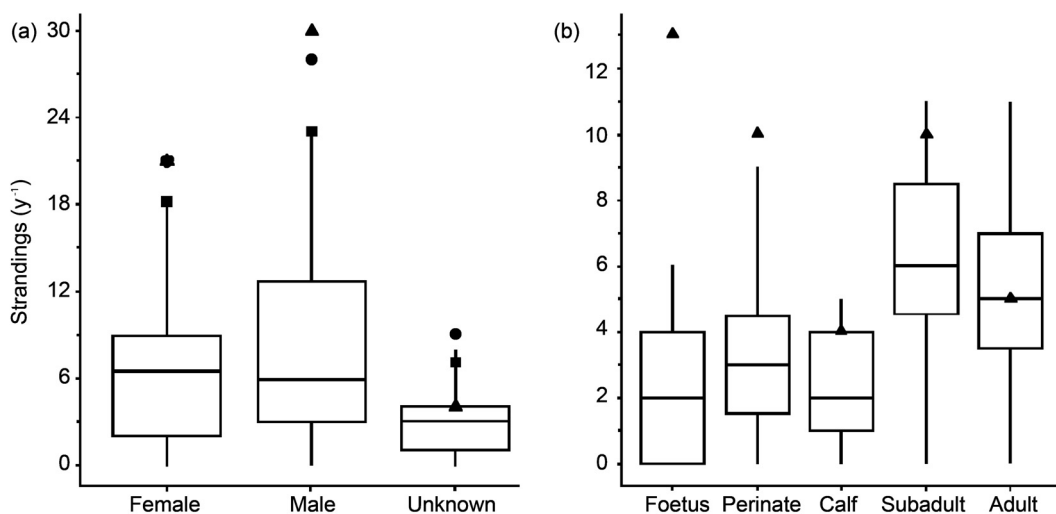


Fig. 3. The number of *Tursiops truncatus* strandings per year separated by (a) sex from 1978–2018 and (b) size class from 2008–2018. Outlier years are shown as shapes (circles = 1990, squares = 1993, triangles = 2011, corresponding to known UMEs shown in Fig. 2).

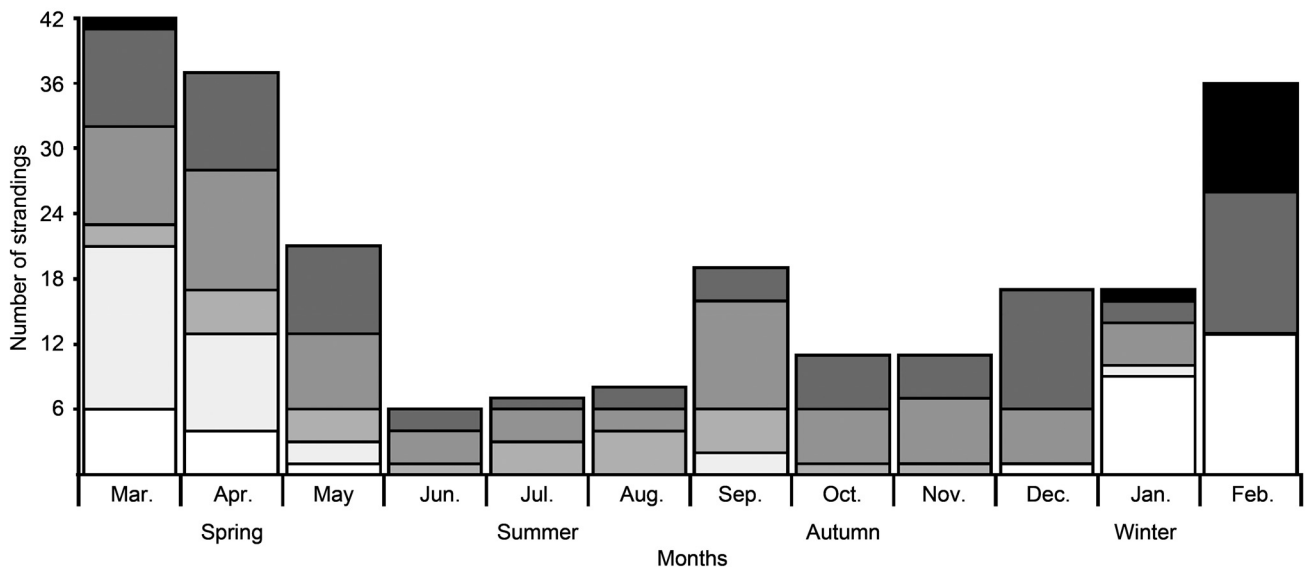


Fig. 4. The total number of *Tursiops truncatus* strandings in Alabama waters from 1978–2018 for each size class, separated by month within each season.

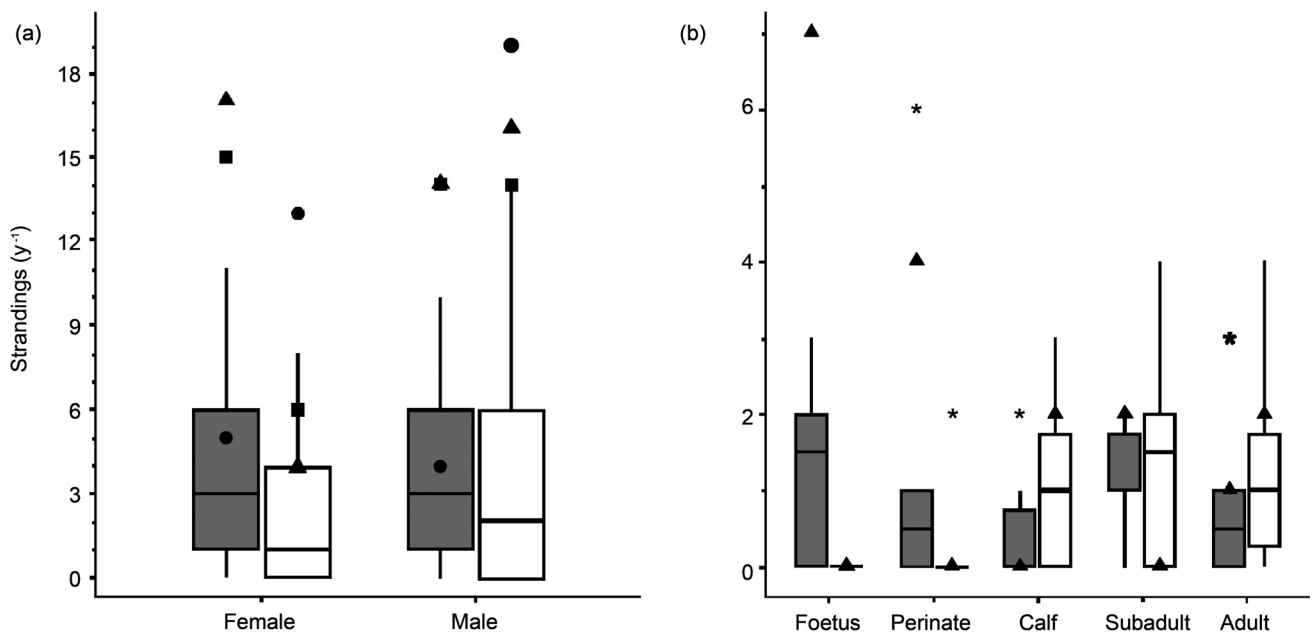


Fig. 5. The number of *Tursiops truncatus* strandings per year during calving season (Jan.–Apr.; dark grey) and non-calving season (May–Dec.; white) for (a) females and males from 1978–2018 and (b) females in each size class from 2009–2018. Outlier years are shown as shapes (circles = 1990, squares = 1993, triangles = 2011, corresponding to known UMEs shown in Fig. 2). \* = statistical outliers that were not also UME years.

$t = 0.98$ ,  $df = 64.61$ ,  $p = 0.33$ ). For the period from 2008–2018 when data were available for size class comparisons, there were no differences in the number of strandings among size classes with Bonferroni corrections (Fig. 3b). Without corrections, more adults and subadults stranded than either calves or fetuses (2008–2018: Kruskal-Wallis chi-squared = 12.24,  $df = 4$ ,  $p = 0.02$ ; Dunn's Post hoc test  $p < 0.04$  for all significant comparisons with and without Bonferroni corrections; Supp. Table 1). The highest number of strandings in any individual year occurred for fetuses during 2011, the outlier year during this time. When this outlier year was excluded, more subadults stranded than fetuses with and without Bonferroni corrections. Without corrections, we also found more adults stranded than calves or fetuses (2008–2018, excluding 2011: Kruskal-Wallis chi-squared = 13.82,  $df = 4$ ,  $p < 0.01$ ; Dunn's Post hoc test  $p < 0.03$  for all significant comparisons with and without Bonferroni corrections; Supp. Table 1).

## DISCUSSION

Thirteen species of cetaceans stranded in AL waters along the northcentral GoM coast during the 41 years of this study, accounting for more than half of the 21 cetacean species that live in the northern GoM (Maze-Foley and Mullin, 2006; Darnell, 2015; Wursig, 2017). We found distinct spatial, temporal and demographic patterns in strandings of cetaceans in this area, which previously had been poorly characterised. Strandings of bottlenose dolphins accounted for the majority of AL strandings from 1978–2018 and had the greatest influence on the long-term dataset. This finding was not surprising given that bottlenose dolphins are the most common marine mammal in nearshore waters of the northern GoM (Mullin and Fulling, 2004; Waring *et al.*, 2015; Pitchford *et al.*, 2018). Because the bottlenose dolphins that strand in AL most likely lived and then died close to the AL shoreline, stranding demographics of these animals are likely to be more useful to defining current and monitoring future status of local populations than data from populations in disparate locations, which have been relied on in the past (Connor *et al.*, 2000).

While non-bottlenose dolphin species were fewer in number, stranding counts of these species should not be ignored. Deceased offshore cetaceans may be undercounted because individuals that die offshore may not be close enough inshore to strand or be reported and instead may be washed out to sea, sink, or be scavenged, similar to findings with offshore sea turtles (Epperley *et al.*, 1996; Williams *et al.*, 2011; Peltier *et al.*, 2012, 2013, 2014). More *Stenella* species stranded than other non-bottlenose dolphin species, likely correlated to the presence of *S. frontalis* on the continental shelf and *S. longirostris* on the continental slope (Fulling *et al.*, 2003; Maze-Foley and Mullin, 2006). Other non-bottlenose dolphin species are oceanic and exist farther away from the coast (Fulling *et al.*, 2003; Maze-Foley and Mullin, 2006). Although they account for little effect on historical stranding trends in the northcentral GoM, strandings of less common species are important to document because they are often the only source of information on anatomy, physiology, life history, and health of these cetaceans and may be indicative of mortality events offshore.

During the study period, multiple distinct hotspots, where strandings of cetaceans were most likely to occur, were identified. These hotspots may be areas of increased documented strandings due to bathymetry and geography, human population density, or sources of mortality such as increased fisheries presence. Dauphin Island, Fort Morgan and Gulf Shores have large, sandy beaches with high public foot-traffic, where stranded animals are likely to be more visible and may be reported in a higher frequency than strandings in marshes or open water. Dauphin Island, Gulf Shores and Orange Beach are populated year-round and have large seasonal influxes of tourists who recreate on the water (Alabama Tourism Department, 2019), which also may contribute to greater year-round reported strandings (Auge *et al.*, 2018). In contrast, Fort Morgan has relatively limited year-round residency, fewer commercial water-based tourist activities than those in Gulf Shores and Orange Beach to the east and is largely made up of uninhabited federal land. The higher number of strandings documented in this area suggests that environmental parameters such as tides, currents and winds, which are known to influence stranding hotspots (Peltier *et al.*, 2012, 2013, 2014) likely affect stranding numbers here. Surface waters on the eastern half of MB travel south along the eastern shore to north of the Fort Morgan peninsula and then flush out of the bay closest to the stranding hotspot observed in Fort Morgan (Du *et al.*, 2018). It is possible, therefore, that some of the cetaceans that strand on Fort Morgan originally died inside MB and were transported by surface water flow to this area. The additional stranding hotspot discovered for non-bottlenose dolphin species on the GoM side of Gulf Shores may indicate a hotspot of strandings for offshore species. This area may be influenced by its proximity to DeSoto Canyon, a submarine canyon in the northern GoM approximately 100km south-southwest of Pensacola, Florida (Haribson, 1968). The canyon is approximately 41km long and 2,100m deep, with known marine mammal presence that may act as a corridor in which confused, sick, or injured offshore cetacean species are funneled toward land (Haribson, 1968; Sirovic *et al.*, 2014; Soldevilla *et al.* 2017). Supporting this idea, we found that offshore species more often stranded on the GoM shoreline of AL rather than inside MB and were more likely to strand alive than inshore bottlenose dolphins. Previous studies have described that changes in bathymetry near coastlines along with currents, riverine discharge, and wind speed and direction can influence strandings in specific locations (Carmichael *et al.* 2012, Forney *et al.*, 2012; Auge *et al.*, 2018), and this is an area of needed study to fully understand the spatial patterns for strandings in the northern GoM.



Seasonal stranding trends are typically analysed using the traditional method of dividing the year into Spring, Summer, Fall and Winter (e.g. Hubard, 2004, Miller and Baltz, 2010; Pitchford *et al.*, 2013). Strandings in AL were documented in every month of the year, but they were highest in February, March and April, months that spanned two of these traditional seasons (Winter and Spring). These months also correspond to bottlenose dolphin calving season (January–April) in the northern GoM (Fernandez and Hohn, 1997; Venn-Watson *et al.*, 2015; Colegrove *et al.*, 2016). The higher frequency of females stranding in calving season than non-calving season is consistent with negative pressures on pregnant females due to reproduction and the greater potential for susceptibility to other stressors such as water quality and disease during pregnancy (Carmichael *et al.*, 2012; Colegrove *et al.*, 2016). Foetuses were also more likely to strand in calving season than non-calving season. In this study, foetuses and perinates were separated based on size, however it is possible that overlap in these classes occurs (e.g. foetus > 95cm or perinate < 95cm). If foetus and perinate stranding numbers are considered together, the increase during calving season becomes even more apparent. This finding supports bottlenose dolphin stranding studies completed in Texas and Mississippi within the GoM, which also found increased perinate strandings during calving season (Fernandez and Hohn, 1997; Pitchford *et al.*, 2013). These patterns highlight the importance of using biologically relevant seasons when analysing stranding trends (Urian *et al.*, 1996).

Previous studies on cetacean mortality have upheld that mortality trends for cetacean size or age classes follow a similar pattern to many terrestrial animals, with highest mortality in the smallest (youngest) and largest (oldest) members of the population and fewer mortalities in the middle-age range (e.g., Wells and Scott, 1990; Fernandez and Hohn, 1997; McFee and Hopkins-Murphy, 2002; McFee *et al.*, 2006). Our findings of higher strandings among subadults is counter to this notion. It is possible that the smaller size classes (e.g. perinates and calves) are less likely to strand and be reported because they are less likely to be discovered, may decompose more rapidly, or are more fully consumed by scavengers compared to larger sized animals; however, this is not well documented and additional study is needed to determine carcass recovery rates specifically for smaller size classes in the northcentral GoM. It is also possible that subadults participate in riskier, more precocious behaviors, such as interacting with humans and fishing gear, than other age classes (Wells and Scott, 1994; McFee *et al.*, 2006). Human and fisheries interactions are a common cause of death in cetacean strandings in AL and adjacent northern GoM waters, and predominately affect the subadult size class (Collins *et al.*, 2019; Bloodgood, unpubl. data), supporting this hypothesis. An additional hypothesis is that subadults were the most abundant age class of bottlenose dolphins in the northcentral GoM during the study period, leading to proportionally higher subadult strandings seen in this study. Data for live populations of local bottlenose dolphins are needed to corroborate this hypothesis, but are limited due to the inability of abundance surveys to fully capture age classes of dolphins and the lack of study in many areas of the northern GoM (Blaylock and Hoggard, 1994).

The influence of UMEs on stranding patterns and baseline data was seen throughout the study period. Statistical analyses with and without the three UME outlier years (1990, 1993, 2011) in AL revealed that UME years had potential to bias or mask the baseline patterns in spatial, temporal and demographic data. To best identify vulnerable population components that may be disproportionately affected during a UME, it is paramount to remove UME data when defining baseline stranding patterns. For example, stranding data from the outlier year 2011, which was a part of the *Deepwater Horizon* Oil Spill (DWHOS) UME and a time of unusual environmental conditions (increased freshwater, prolonged cold temperatures) in the northern GoM, masked the higher number of subadult strandings and the increased number of female strandings during calving season during 2008–2018. This example highlights that when UME and outlier years are included in historical analyses of stranding datasets, we may miss strandings that are naturally elevated (e.g., females and foetuses during calving season). If this study had been completed and baseline stranding patterns defined prior to 2011, it may have been possible to predict that foetuses would be the age class most vulnerable to the combination of the DWHOS and other environmental stressors that occurred in the first half of 2010 and contributed to the 2011 UME (Colegrove *et al.*, 2016). This study underscores the importance of outlier analyses to assessment and application of long-term datasets to set baselines for management and conservation, particularly when UMEs have occurred in the past.

This analysis of a long-term dataset fills an important gap in data on cetacean strandings in a marine suture zone linking the eastern and western sides of the northern GoM (Portnoy and Gold, 2012). The 41 years of data

analysed in this study highlight the importance of long-term stranding datasets in establishing spatial, temporal and demographic stranding trends. This study also demonstrates the need for consistency in stranding response to accurately define baseline stranding data for cetacean populations. Important components of consistent stranding response include establishing and raising public awareness of a single authority for reporting, and provision of funding for staffing, equipment and training to meet the stranding response needs of the area (Wilkin *et al.*, 2017). With consistent stranding response and long-term data collection and analyses, baseline stranding patterns can be identified and used to define vulnerable population groups and inform meaningful, range-wide management and conservation.

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