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Abstract

Sea turtle stranding events provide an opportunity to study drivers of mortality, but causes of strandings are poorly understood. A general turtle carcass oceanographic drift model was developed to estimate likely mortality locations from coastal sea turtle stranding records. Key model advancements include realistic direct wind forcing on carcasses, temperature driven carcass decomposition and the development of mortality location predictions for individual strandings. We applied this model to 2009-2014 stranding events within the Chesapeake Bay, Virginia. Predicted origin of vessel strike strandings were compared to commercial vessel data, and potential hazardous turtle-vessel interactions were identified in the southeastern Bay and James River. Commercial fishing activity of gear types with known sea turtle interactions were compared to predicted mortality locations for stranded turtles with suggested fisheries-induced mortality. Probable mortality locations for these strandings varied seasonally, with two distinct areas in the southwest and southeast portions of the lower Bay. Spatial overlap was noted between potential mortality locations and gillnet, seine, pot, and pound net fisheries, providing important information for focusing future research on mitigating conflict between sea turtles and human activities. Our ability to quantitatively assess spatial and temporal overlap between sea turtle mortality and human uses of the habitat were hindered by the low resolution of human use datasets, especially those for recreational vessel and commercial fishing gear distributions. This study highlights the importance of addressing these data gaps and provides a meaningful conservation tool that can be applied to stranding data of sea turtles and other marine megafauna worldwide.

Keywords: Sea turtle strandings; Sea turtle mortality; Chesapeake Bay; Drift simulations; Fisheries and vessel interactions; Endangered species; Marine conservation; Protected species management

1. Introduction

Many of the world's marine megafauna are highly threatened by a mixture of anthropogenic pressures (Learmonth et al. 2006, Crain et al. 2009, Wallace et al. 2013, Lewison et al. 2014) and natural threats (George 1997, Gulland and Hall 2007, Heithaus et al. 2008). Among these species are marine sea turtles, of which six out of the seven species worldwide are listed on the IUCN Red List of Threatened Species (http://www.redlist.org). For sea turtles and other marine megafauna, a better understanding of the impacts of anthropogenic activities on these species is essential to assessing risk of population extinction and identifying effective conservation strategies. Sea turtle strandings provide an important opportunity to study turtle mortality and identify threats for future mitigation and conservation actions, however, identifying potential causes of mortality of stranded sea turtles can be extremely challenging due to state of carcass decomposition and the lack of physical evidence of the cause of mortality (Hart et al. 2006, Koch et al. 2013). In particular, interactions with some fishing gears often do not leave marks on turtles, due to a combination of gear type and sea turtle anatomy (i.e. hard parts), thus solely using injuries noted at time of stranding to attribute cause of death has been suggested to grossly underestimate fisheries-induced mortality (Barco et al. 2016). Fishing activity has been noted as a large driver of anthropogenic sea turtle mortality worldwide, with lethal interactions documented in gear types including longlines, trawls, gillnets, pound nets, dredges, seines and pots (Lewison et al. 2004, Zollett 2009, Wallace et al. 2010, Finkbeiner et al. 2011). Despite the current vulnerability of sea turtle species and known interactions with recreational and commercial fishing gear, as well as commercial and recreational vessel traffic, management actions are still frequently hindered by lack of specific information on where and when human-turtle interactions occur.

The Chesapeake Bay (Bay) and its surrounding coastal waters are critical foraging and developmental habitats for thousands of sea turtles that use these waters seasonally (Musick and Limpus 1997, Mansfield 2006). However, hundreds of deceased turtles are found stranded on Virginia's coastline each year. The Virginia Sea Turtle Stranding and Salvage Network (VAQS), currently led by the Virginia Aquarium & Marine Science Center, has been responding to strandings throughout the state since the 1970s, documenting 100-300 events annually in the past decade (Swingle et al. 2016). Strandings are observed throughout the year, although the majority of annual strandings usually occur during a strong spring peak in May and June when turtles are first entering the Bay (Lutcavage and Musick 1985, Coles 1999). Mortality continues at a relatively high level throughout the summer, followed in some years by a small fall peak in strandings associated with turtles migrating out of the Bay to avoid cold winter temperatures (Mansfield et al. 2009). Juvenile loggerheads are the most commonly reported sea turtles found within Virginia's waters, followed by the critically endangered Kemp's ridley (http://www.redlist.org) (Lutcavage and Musick 1985, Coles 1999, Barco and Swingle 2014). Importantly, Virginia's waters provide crucial habitats for loggerheads from several different western Atlantic distinct population units (Conant et al. 2009, Mansfield et al. 2009, NMFS 2011, Ceriani et al. 2017), thus local mortality could lead to detrimental impacts among multiple loggerhead subpopulations (Mansfield et al. 2009). . Strandings likely represent a minimal measure of actual at-sea mortality, with some studies in open ocean environments estimating stranding events to represent only 10-20% of total deaths (Epperly et al. 1996, Hart et al. 2006; note, however, that these stranding percentages may be higher in the semi-enclosed Bay). Given the important role the Bay plays in regional sea turtle life cycles, detailed information on the times, places and causes of mortality are essential to maintaining and increasing these populations.

When stranded sea turtles are recovered as fresh dead carcasses, cause of death can often be determined by conducting a thorough necropsy and submitting tissues to a veterinary pathologist for histopathology. Barco et al. (2016) summarized cause of death for 70 fresh carcasses recovered in Virginia and North Carolina from 2004-2013. Nearly half of the turtles (n=31; 44%) died from acute vessel (n=15) or fishery interaction (n=16) and most of these were apparently healthy prior to death with no significant pathology and good body condition, suggesting they were not already compromised in any way prior to mortality (Barco et al. 2016). Of those turtles that were categorized as drowning from fishery interaction, few, if any, lesions were present on the carcasses (Barco et al. 2016), which is similar to some fishery interaction cases in cetaceans (Moore et al. 2013). This lack of injuries has importance for the majority of dead stranded sea turtles observed in Virginia, which are in a moderate to advanced state of decomposition at time of discovery. Though some causes of death, such as drowning due to underwater entrapment in fishing gear, are impossible to definitively assess in these more extensively decomposed cases, they often share several of the characteristics of fishery interactions, such as a lack of lesions or obvious pathology. Collectively, these results suggest that vessel and fishery interactions are important sources of human-induced mortality in the Bay, but more information is needed on the locations of mortality to help pinpoint the gears or vessels likely responsible. Turtles in this region have been documented caught or entangled in pound net leader hedging, gillnets, trawl nets, crab pot lines and whelk pot lines (Bellmund et al. 1987, Keinath et al. 1987, Mansfield et al. 2001, Barco et al. 2016). Although there is no concrete evidence of the Chesapeake Bay’s menhaden purse seine fishery causing sea turtle mortality, other purse seine fisheries in the region are known to kill turtles (Silva 1996) and there is no state-run observer program for this and many other fisheries in the Bay (Barco et al. 2015). Narrowing down this list of potential causes for sea turtle mortality in the Bay to the most prevalent causes, locations and time periods is essential to developing targeted conservation strategies for these threatened species.

Mitigating sea turtle mortality (especially when fishery observer data are limited) requires investigation into the location of mortality in order to assess potential causal mechanisms and identify hotspots for negative human-turtle interactions. After sea turtles die, their bodies sink until decomposition gases causes the body to bloat and float to the surface (if not entangled). Partially submerged and acting as drifting objects, carcasses are transported by winds and currents. Landfall may occur if conditions are favorable to onshore transport and the turtle carcass does not decompose and sink before reaching a coastline. Santos et al. (2018) found that sea turtle carcass drift time is highly dependent on water temperature due to decomposition rates and that winds make an important contribution to the net transport of turtle carcasses. Oceanographic modeling and drift studies have been used in the past to understand mechanisms for larval release and dispersal (Garavelli et al. 2012), as well as to predict trajectories of drifting human bodies (Carniel et al. 2002) and cetacean carcasses (Peltier et al. 2012). A limited number of recent studies have applied this approach to sea turtle carcasses in other geographic regions (Hart et al. 2006, Nero et al. 2013, Koch et al. 2013), providing valuable insight on stranding causes and likelihood. Santos et al. (2018) conducted preliminary investigations into sea turtle carcass drift patterns within the Chesapeake Bay area specifically, however strandings were not assessed at the individual level, with potential mortality hotspots based on fairly general areas of historically high stranding rates. Furthermore, only stranding locations during June, the peak month of sea turtle strandings in Virginia, were assessed.

In this study, we construct an oceanographic drift model for the lower Chesapeake Bay to predict the probable location of mortality for individual coastal sea turtle strandings in Virginia based on the location of stranding, state of carcass decomposition and environmental conditions. We simulated the drift patterns of dead turtles prior to stranding and identified likely locations of sea turtle mortality using the starting points of particle trajectories arriving at the stranding location at the correct time and decomposition state. Empirical results from Santos et al. (2018) were used in the drift model to parameterize the probable oceanic drift time as a function of temperature and the impact of direct wind forcing on carcass drift. We applied this adjusted model to individual sea turtle stranding observations in coastal areas of Virginia and most probable mortality locations within the region were identified for specific classes of strandings with similar characteristics (e.g., probable cause of death, state of carcass decomposition).

Overall, this study provides a basis for quantitative and qualitative comparisons with spatial distributions of potential causes of sea turtle mortality in the Bay. Our previous work parameterized the characteristics of drifting sea turtle carcasses and found general areas of likely sea turtle mortality in the Bay (Santos et al. 2018). Here, we build upon that preliminary study to predict the trajectories and mortality locations of individual strandings, aggregating results over many events and making comparisons with available information on potential causal mechanisms. The model constructed in this paper also includes a number of methodological improvements to the methods outlined in Santos et al. (2018), including the incorporation of winds, currents, temperature and carcass condition on carcass drift, that can be applied to stranding data for sea turtles and other marine megafauna around the globe to better understand and mitigate mortality events.

2. Material and Methods

A model simulating the drift of dead sea turtles prior to stranding was developed using the offline Lagrangian drift simulation tool Ichthyop version 3.3 (Lett et al. 2008, Santos et al. 2018). The model was configured to release 20,000 pseudo-particles (i.e. simulated particles) throughout the oceanographic domain every three hours and run forward in time based on transport estimates from a wind reanalysis product and an ocean circulation model (Fig. 1a). Pseudo-particles arriving at stranding locations at the appropriate time (i.e. probable date of landfall based on reported stranding date) and having a desired set of conditions (see below) were considered to represent potential turtle carcass drift trajectories. The release points for many such trajectories were aggregated to create a probability distribution representing likely mortality locations of stranding events.

Water circulation information was derived from an implementation of the Regional Ocean Modeling System (ChesROMS; version 3.6) for the Chesapeake Bay area (Feng et al. 2015, Irby et al. 2016, 2017, Luettich et al. 2017, Moriarty 2017, Da 2018) and wind forcing was obtained from the North American Regional Reanalysis (NARR) (Mesinger et al. 2006). The horizontal grid cell size for ChesROMS and NARR varied over space, but was on average 1.7 km and 32 km, respectively. ChesROMS included tidal fluctuations and fresh water inputs from major rivers in the region. ChesROMS,NARR data and Ichthyop output timesteps were all 3 hours. Ichthyop's internal timestep was set to 20 minutes.

The amount of direct wind forcing on the surface transport of turtle carcasses is estimated to be 1-4% of wind speed (Nero et al. 2013, Santos et al. 2018). Wind forcing was thus added to the ChesROMS currents at 0%, 2% and 4% of wind speed to assess sensitivity of estimates to wind forcing levels over the range of experimentally observed levels. Resulting particle trajectories therefore represent the combined impacts of wind and currents on carcass movements. When presenting model results, 2% wind forcing will be used unless otherwise indicated because it is closest to experimentally observed values in Santos et al. (2018). A comparison of drift trajectories from modelled pseudo-particles to experimentally-observed data can be found in the supplement materials.

2.1 Stranding data

Sea turtle stranding data collected by VAQS during 2009-2014 were analyzed. Strandings include dead and live animals, but the potential for active swimming of sick turtles found alive can complicate the simulation of their movements. In this study, we focus only on deceased individuals found washed ashore and refer to these as "stranded turtles" with the understanding that we are excluding live turtle strandings. All stranding data were reviewed and each event was consistently assigned a stranding date (date of report, not date of examination, if different), carcass condition (at time of report, if available) and probable cause of death (based on gross external and internal examinations). Carcass condition was determined on a qualitative scale of 1 (freshly dead) to 5 (bones) as per the National Oceanic and Atmospheric Administration's Sea Turtle Stranding and Salvage Network guidelines (https://www.sefsc.noaa.gov/species/turtles/strandings.htm). Causes of death included: vessel strike, disease, cold-stunning, pollution/debris, entanglement, no apparent injuries and unable to assess. When moderately and severely decomposed turtles were examined, but no injury or disease was observed, the probable cause of death was listed as "no apparent injuries". Thus, turtle carcasses classified as "no apparent injuries" includes turtles that appear to have been healthy prior to death. The category “unable to assess” was comprised of stranding events with insufficient information (i.e. evaluated by an unqualified observer, necropsy was not performed, etc.) to assign a probable cause of death category.

The developed model depends on the assumptions that stranded turtles died at-sea, were able to float freely (i.e. not entangled), and the stranding event was reported and documented shortly after beaching on land. Carcass decomposition state at time of discovery on the beach was recorded on a condition code scale from 1 to 5, with lower condition codes indicating a “fresher” carcass that likely died more recently, and, thus, drifted for a shorter amount of time. Based on experimental results that turtles are positively buoyant and capable of drifting only until code 3 (Santos et al. 2018), stranding events with condition codes 4-5 were omitted from analyses as beach time to decay to these states was difficult to determine and open ended. Thus, analysis of stranding data was limited to turtles within the model domain that were classified as condition codes 1-3 (n=1023).

We also limited analyses to strandings documented on the coastlines within identified regions of relatively high human population densities, with the assumption that strandings in these areas were discovered and reported in a timely manner (n=751; 73%). This included stranding events documented along the coasts of Virginia Beach, Norfolk, and bayside Northampton County (Fig. 1b). Virginia Beach and Norfolk are highly inhabited areas and popular summer vacation spots, where water front areas are frequently visited in the warmer months (Virginia Tourism Corporation 2015). Strandings in these areas were assumed to be observed and reported by a member of the public at least every 24-hours. Although Northampton has a lower population density, visitors frequently walk the beaches during the popular summer months, particularly along the southern bayside of the peninsula where most strandings were reported. The ocean-facing coastline of Northampton County is made up of uninhabited barrier islands that are difficult to access, thus strandings in these areas were omitted from analyses (n=22; 2.2%). We also excluded strandings located in small tributaries and other waterways, for these water areas were not well represented in the oceanographic model and the assumption of observation within 24-hours likely did not hold true (n=20; 2.0%).

From the remaining subset of strandings, we focused on those occurring during the spring, summer and fall (May-October/November) (n=651; 87%). Due to lethal water temperatures, turtles are not typically present in the Bay during the winter when temperatures fall below 18^{o}C (Lutcavage and Musick 1985, Coles 1999). Turtles that strand during this non-residency period likely either died long before being observed or drifted over long times and distances, both of which complicate estimating their probable mortality locations.

2.2 Criteria for a "successful" stranded particle

Three basic conditions were established to determine which particle trajectories potentially correspond to the drift pathways of a stranded turtle, including: 1) arriving within the stranding target area, 2) arriving within a 24-hour time period around the documented stranding event, and 3) having the appropriate state of decomposition (Fig. 2).

A target zone was created around the geographic location of each stranding event. Stranding coordinates were snapped to the coastline of the model domain and a target zone with a water area of 28.3 km^2 was created around each stranding location. This area is equal to the area of a 3 km radius circle, but the actual offshore extent of the buffer around each stranding location was varied so that the water area was constant across strandings after taking into account differences in coastline morphology. Carcass drift simulations were run targeting these specific individual target zones before and up to the date of the corresponding strandings.

It was assumed that beaches in Virginia Beach, Norfolk, and bayside Northampton County were observed for turtle strandings once a day, ranging from approximately 6am to 6pm EST (local time) (Nero et al. 2013). Therefore, we assumed that the actual beaching event in these areas could have occurred anytime from 6pm the night before to 6pm the day of the reported stranding. This 24-hour duration was used as the stranding window for simulations, with “competent” particles (described below) arriving in the stranding target zone during this time period considered to have “successfully” stranded.

Particle tracking times were based on results of a recent sea-turtle carcass decomposition study (Santos et al. 2018) that used modeled water temperatures along particle trajectories and carcass condition codes to determine drift duration. Here, we limited turtle carcass drift duration to the interval of positive buoyancy (i.e., after the turtle had bloated sufficiently to float to the surface, but before decomposition released internal gases causing the carcass to sink again to the bottom). Linear regressions were used on buoyancy and condition code results from Santos et al. (2018) to determine the minimum and maximum duration a floating carcass spent in each condition code at a given water temperature (Fig. 3). As turtles in condition 1 were not observed buoyant in the study, condition code 1 turtles were assigned a maximum drift duration of 1 day (similar to Nero et al. 2013), and drift duration for turtles with condition codes 2 and 3 were increased by 24 hours relative to raw results from Santos et al. (2018).

Each model pseudo-particle had a minimum and maximum drift time during which the particle was considered to be buoyant and to have the observed condition code for the corresponding stranding. If temperatures were constant over space and time, then the minimum and maximum drift times would be given by the results from Santos et al. (2018) at fixed temperature (Fig. 3; for example, at 20^{o}C, we would predict a code 3 turtle would have been drifting for approximately 7 to 12 days). However, as temperatures vary, the advancement of a particle towards the minimum and maximum drift duration over a model timestep was assessed as equal to the fraction of the minimum and maximum drift times that the timestep represents for the temperature at the particle location. These fractions were cumulatively summed over timesteps until the total fraction for minimum drift time was >1, but the total for maximum drift time was <1. This defined a "competency" window for each particle trajectory during which the carcass was considered to be of the appropriate decomposition state to strand. Particles were then assessed to see if they were within the stranding target zone during this time interval.

Simulations were run targeting each stranding zone individually and starting points of "successful" stranding particles were mapped on a 5 km x 5 km grid. For each stranding, a relative particle density was calculated for each grid cell representing the estimated probability that the turtle died in that grid cell. For each release event (occurring every 3 hours), the number of particles released in each grid cell that successfully landed in the stranding zone at the appropriate time was divided by the total number of particles released in that grid cell to get the relative probability of "successful" stranding. These relative probabilities were then summed over all release events for that stranding and the resulting sum for each grid cell was further divided by the sum over all grid cells so that the total probability of mortality over all grid cells for a given stranding event was 1.

2.3 Analyses

Probable mortality locations for individual stranding events were aggregated over the six-year study period by time of year and/or stranding type to develop synthetic maps of recurrent mortality locations. Probability maps for groups of strandings were added together and then the total was divided by the number of strandings to obtain a final synthetic normalized probability map for the group of strandings. Stranding events having a low number of particles that met all the stranding criteria (defined as <100 particles in total) were omitted from these syntheses to prevent skewing results in specific cases where fine-scale coastal movements may not have been accurately represented in the model (n=13 for 0% wind forcing, n=23 for 2%, and n=48 for 4%)

Strandings occurring during the spring peak in May and June and throughout the rest of the summer stranding period were analyzed separately to investigate potential differences in mortality locations and sources between these two time periods. The timing of the spring peak period was independently assessed for each year by plotting the number of strandings per week and visually identifying the sharp peak in strandings in May, indicating onset, followed by a sharp drop off during June, representing the end of the peak period approximately 3-5 weeks later. The duration of the remaining summer and early fall foraging season was defined in a similar manner to encompass the time period after the end of the spring peak until the frequency of stranding events greatly diminished around October or November. This period varied by year from 19 to 23 weeks, occasionally including an irregular second fall peak in strandings. The fall peak was not separately analyzed as it was hard to consistently define across years and represented a relatively small total number of strandings (Fig. A.1).

Probability maps of turtle mortality locations were further categorized by probable cause of death as determined by necropsy results and external visual observations of the stranded turtles. Categories examined included vessel strike (n=250; 38%), no apparent injuries (n=163; 25%), and unable to assess (n=199; 31%). The remaining 6% of strandings (n=39) included carcasses with death attributed to disease, cold-stunning, pollution/debris or entanglement. Due to low sample size and diversity surrounding potential causes of mortality, these strandings were excluded from analyses. Combining the two stranding "seasons" (spring peak and remainder of the summer/fall) and these three probable cause of death categories yielded a total of six possible synthetic maps, of which only five were produced because there were no code 1 strandings that met all our criteria during the non-peak stranding period in the “unable to assess” category.

Spatial overlap between predicted mortality locations of vessel strike turtles and U.S. Coast Guard shipping lane data were evaluated to assess model validity and identify areas of high mortality due to vessel traffic. Vessel location data from the Automatic Identification System (AIS) for non-federal vessels over 65 ft in length were obtained during the 2009-2014 time period at 1-minute intervals (https://marinecadastre.gov/ais/). We limited data to vessels traveling faster than 4 km/hr, the reported speed at which turtles cannot actively avoid being struck by watercraft (Hazel et al. 2007). Vessel density was computed for each year-month strata and rasterized on the 5 km x 5 km grid used to predict turtle mortality. As AIS position data is limited to larger, non-federal vessels, it does not include many vessels that could be responsible for boat strikes. Therefore, we chose to use a wider, monthly temporal resolution to better capture general boat traffic in the bay. Relative probability of vessel activity for each year-month was computed by dividing the number of AIS data points in each grid cell by the total number of points over all grid cells for that strata. The predicted mortality location map for each stranding record was multiplied cell-by-cell with the corresponding year-month relative vessel activity layer, resulting in a joint probability distribution map, with each grid cell representing the probability that both vessel activity occurred and the turtle died in that location. This joint probability map was summed over all grid cells to develop a single indicator of the overlap between predicted mortality locations and AIS-tracked vessel activity. AIS data from September to November 2014 were incomplete, so vessel strike turtles that stranded during this time period were omitted from analyses (n=18).

In order to assess whether or not the model was successfully predicting the mortality locations of known vessel strike stranding records, a Monte Carlo randomization analysis was performed to compare overlap between vessel activity and the predicted mortality locations of these strandings with the overlap for a randomized mortality location probability map. For each individual stranding event, the model-predicted probability map was randomly reshuffled over the area of all possible mortality locations of turtles for the corresponding year, resulting in a randomly distributed probability map. Similar to the model predicted maps, the randomly generated mortality grids were multiplied by the vessel activity map and summed over all grid cells to obtain an indicator of the overlap between these data sets. This process was repeated 5,000 times for each individual stranding event. A pseudo-p-value was calculated as the fraction of these 5,000 trails for which the model predicted map had a lower overlap with vessel activity than the randomly distributed null maps. These pseudo-p-values were then aggregated by stranding condition code and plotted as a density function.

Predicted mortality locations for stranding records with probable cause of death classified as "unable to assess" and "no apparent injury" were identified and spatially compared to data on anthropogenic activities. Total harvest for different gear types throughout the Chesapeake Bay were obtained from the Virginia Marine Resource Commission (VMRC) for the 6-year study period. Spatio-temporal maps of fishing effort are not generally available for fisheries in the Chesapeake Bay, so instead we used total harvest as a rough indicator of extraction intensity in general regions. Due to privacy and data resolution issues, harvest was only available as an aggregate over the entire study period and for individual "waterways", marine areas defined by VMRC and used for harvest reporting by fishermen (Fig. A.2). Gear types that are thought to pose particular threats to sea turtle, including gillnets, haul seines, and pots and traps were subset and mapped by waterway. To ensure confidentially in cases where the number of harvesters per gear-waterway combination was low, results for certain water areas were grouped together by "water system" (a larger area defined by VMRC to include multiple nearby waterways). In the 10% of instances where this occurred, total pounds harvested per gear-waterway strata was estimated by dividing the gear-water system total among the number of waterway represented within the grouping. Fine scale pound net and stake gillnets locations were obtained from the VMRC website for 2017, the current license year at the time of the study (https://webapps.mrc.virginia.gov/public/maps/chesapeakebay_map.php). Point locations were extracted and plotted on the 5 km x 5 km grid by length of net per unit area. Although fine-scale information on staked gillnets and pound nets locations were only available for 2017, these are stationary, semi-permanent fishing gears that likely remain in the same general area over many years. In addition, this point license location information matches relatively well with available broader-scale information on aggregated 2009-2014 harvest (Fig. A.3). Therefore, the gridded 2017 stake gillnet and pound net locations were deemed appropriate to use for comparisons with the 2009-2014 data. Location of purse-seine sets by Omega Protein vessels from 2011-2013 were obtained from the 2015 Atlantic Menhaden Stock Assessment Report (SEDAR 2015). Images of set locations were georeferenced and digitized in ArcGIS, and presence/absence of purse seines noted on a 5 km x 5 km grid.

To assess changes in carcass drift duration throughout the stranding season, timespan and distance from point of release to the first timestep for which all three stranding criteria were met was recorded for each "successful" stranding particle for all stranding events. Given the variability in drift criteria across condition code, we limited this analysis to strandings classified as condition code 3 to observe trends at the maximum range (results for condition code 2 strandings were qualitatively similar). Average drift times and distances per stranding were binned by week of the year and averaged together over the 6-year study period.

3. Results

Possible drift time for strandings classified with condition codes 2 and 3 decreased with warming water temperature (Fig. 3). The effect of temperature was found to be statistically significant on the maximum drift time for code 2 turtles ($p<0.001$, $R^2 = 0.7495$) as well as the minimum ($p<0.01$, $R^2 = 0.7947$) and maximum ($p<0.001$, $R^2 = 0.8932$) drift times for code 3 turtles (Table 1).

Average drift times and straight-line distances for pseudo-particles successfully arriving at condition code 3 stranding target zones decreased throughout the late spring (May-late June), reached minimal values of ~2-5 days and ~15-30 km, respectively, during the summer months (late June-late September) before increasing again in the fall (late September-November) (Fig. 4a-b). The minimum in both drift times and distances occurred in July, shortly after the spring peak period. A significant relationship was noted between drift time and drift duration (Fig. 4c; $p<0.001$, $R^2 = 0.2746$).

Although predicted mortality locations differed among probable cause of death categories, as well as between spring peak and summer, non-peak stranding time periods, high probability zones for mortality were consistently identified in areas within the main channel of the lower Bay, as well as the James River which includes the port of Hampton Roads (Figs. 1c, 5-7). Mortality locations for strandings where vessel strike was the probable cause of stranding were largely concentrated in the southwest portion of the Bay, while most probable locations for strandings classified as having no apparent injuries or where responders were unable to assess cause of stranding were generally more dispersed and also included areas in the southeast quadrant of the Bay. In all cases, mortality was less likely to occur in tributaries of the Bay, with a notable exception of the James River.

3.1 Vessel strikes

Analyses of commercial vessel density data highlighted high vessel activity during months with observed stranding data in the lower Chesapeake Bay, particularly along shipping channels of bayside areas of Norfolk and Virginia Beach and within the lower James River (Fig. 5a). Overall predicted mortality locations of stranded sea turtles with evidence of vessel strike were concentrated in the lower, southwest portion of the Chesapeake Bay (Fig. 5b). In particular, high probability was noted near the mouth of the James River and the bayside coast of Norfolk in the vicinity of both the commercial and military ports. Mortality was also moderate to high near the bayside coast of Northampton County, near the mouth of the Bay, and in the northern oceanic-coast of Virginia Beach. A combined probability map depicting overlap of both vessel activity and probable vessel strike turtle mortality was very heavily weighted towards the immediate vicinity of the Lynnhaven Inlet and Elizabeth River (Figs. 1c, 5c).

Results from the Monte Carlo randomization analyses showed a strong distribution of low p-values across all condition codes, indicating that the model was doing considerably better than random at predicting vessel-strike mortality event locations (Fig. 8). Actual predicted mortality locations derived from the model were better ($p<0.05$) at predicting overlap with vessel activity than expected by random chance for approximately 67% of code 1 turtles (4 out of 6 strandings), 73% of code 2 turtles (115 out of 156), and 46% of code 3 turtles (30 out of 71).

3.2 No apparent injuries and unable to assess

Predicted mortality locations for strandings classified as “no apparent injuries” or “unable to assess” generally occurred throughout the lower Bay, with noted differences in probable mortality locations between the spring peak in strandings and the rest of the summer stranding period (Figs. 6-7). Turtles classified as condition code 1 originated in nearshore areas relatively close to stranding locations. Although sample size was low, elevated mortality probability for these strandings were noted near the bayside coasts of Virginia Beach and Northampton. As cause of death was easier to determine in fresher carcasses, there were no documented code 1 "unable to assess" strandings that met all stranding data conditions during the non-peak stranding period. During the spring peak, predicted mortality locations for turtles classified as either condition code 2 or 3 were heavily concentrated at the mouth of the James River and along the Northampton County lower bayside coast. Additionally, there was a strong likelihood of mortality near Hampton County (Fig. 1c) for condition code 3 turtles classified as "no apparent injuries" that was not present in any of the other images, with elevated mortality probability concentrated in a region spanning across the lower main-stem of the Bay. Non-peak stranding mortality locations were generally more diffuse in space, with high probability near the bayside coast of Northampton County.

3.3 Wind forcing

Although major areas of predicted mortality remained the same between 0%, 2%, and 4% of wind forcing on carcass drift, increasing winds had a general tendency towards increasing the spread and geographic range of predicted mortality locations (Fig. 9). For example, the three concentrated regions of high predicted mortality locations for turtles classified as condition code 2 with no apparent injuries during the spring peak, including, the James River, the southern bayside coast of Northampton County, and the Virginia Beach Oceanfront, were most obvious with 0% wind forcing and became slightly smaller at 2% and 4% (Fig. 9). However, an area of high mortality remained constant within the lower southwest portion of the Bay and the James

River across all three wind speed percentages. The high likelihood of mortality occurring in this area across all wind conditions was highlighted in a map depicting the mean of these three probability images (Fig. 9d).

3.4 Fishing data

Focusing primarily on those gears and fisheries that were most active in the lower Bay and James River and were predicted to be associated with turtle mortality that could lead to strandings (Figs. 6-7), we found that areas of activity of sink/anchor gillnets (as well as drift gillnet to a lesser extent; Fig. 10a-b), haul seines (Fig. 10c), crab pots and traps (Fig. 10e), and the purse-seine fishery for Menhaden (Fig. 11a) overlapped extensively with areas of predicted sea turtle mortality. Nevertheless, the limited spatial and temporal resolution of the data made quantitative assessments of overlap impossible. Of the fixed gears, only pound net locations (Figs. 11c) corresponded with some of the predicted turtle mortality locations along the bayside of Northampton County. Whelk pots and traps (Fig. 10d) and sink gillnets (Fig. 11b) were located in regions of the upper Bay or oceanic waters outside the Bay, areas that generally did not greatly overlap with predicted turtle mortality locations.

4. Discussion

In this study, we developed the first model for predicting mortality locations of individual stranded sea turtles in Virginia, USA, using a methodology that is widely applicable to stranding data for sea turtles and other megafauna around the world. The novel approach used in our model incorporated wind, current, and temperature effects on carcass drift to stranding locations. We identified probable mortality locations for different cause of stranding categories for sea turtles in the Chesapeake Bay, making comparisons between high-probability areas with available information on fisheries activity and commercial vessel traffic. Identified hotspots during the peak (May-June) and non-peak (July-October/November) stranding season suggest that much of the mortality leading to sea turtle strandings in the lower Chesapeake Bay occurs in two distinct regions: 1) near the vicinity of the James River and 2) near the lower bayside coast of Northampton County. These results are in line with those of Santos et al. (2018), who identified probable mortality hotspots during the peak month of strandings (June) within the lower Bay.

4.1 Vessel strikes

Combined probability maps of vessel density and predicted mortality locations for turtle likely to have stranded due to a vessel strike suggests that watercraft interactions leading to mortality occur primarily in the lower Chesapeake Bay just north of Virginia Beach in the vicinity of the Lynnhaven Inlet, as well where the James and Elizabeth rivers meet (Fig. 5c). Given the importance of the Norfolk and Virginia Beach areas for commercial, recreational and military maritime traffic, turtle-vessel interactions were to be expected. Sea turtles are susceptible to interactions with vessel activity throughout their entire range, with vessel strikes identified as an important mortality factor in several nearshore turtle habitats worldwide (Orós et al. 2005, Chaloupka et al. 2008, Casale et al. 2010). In a Florida study, nearly all injuries consistent with vessel strike on stranded sea turtles occurred antemortem or perimortem, regardless of the level of carcass decomposition. These results suggest that vessel strikes seldom occur with moderately to severely decomposed turtles which float above the water line (Brian Stacy, personal communications). In Virginia, loggerheads appeared to be particularly affected by vessels and rarely survived severe propeller trauma, especially if the trauma occurred in the cranial two-thirds of the carapace (Barco et al. 2012a, Barco and Swingle 2014). Barco et al. (2016) noted that the majority of loggerheads that stranded in the Bay with vessel damage represented normal, healthy turtles prior to interactions, which suggests that mortality occurs as a direct result of lethal vessel-turtle contact. Our results complement this information by providing precise target areas for mitigation efforts to reduce probability of lethal vessel-turtle interactions.

Overall, analysis of vessel strike mortality location predictions suggested that our model was a good predictor of mortality locations for stranded turtles. Our Monte Carlo randomization analysis indicates that mortality location predictions overlap vessel activity maps far more than one would expect at random (Fig. 8). Based on the overlap with vessel activity, the drift model was better at predicting mortality locations for stranded turtles classified as condition codes 1 and 2 than code 3 turtles. This is as one would expect, for turtles found in fresher decomposition conditions likely had only a short amount of time to drift before stranding, leading to lower uncertainty in their drift trajectory.

Although the analysis of turtles with evidence of death by watercraft interaction provided a good proxy for assessing model accuracy, the nature of the AIS boat position data underrepresents, and may misrepresent, overall vessel activity in the Bay. AIS provided a vast amount of real-time vessel track data, but was only legally required for non-federal vessels 65 ft and larger, including large commercial vessels and industrial fishing vessels (Title 33, Code of Federal Regulations, Part 164). The data do not account for smaller commercial vessels and recreational vessels. Propeller lesions on stranded turtles in Virginia suggest that at least some portion of vessel strike mortality was due to smaller propellers that are common on smaller vessels (Barco et al. 2011). Furthermore, all vessels owned and operated by the U.S. government are legally exempt from AIS data reporting requirements (Title 33, Code of Federal Regulations, Part 164). The Chesapeake Bay has a significant number of military ports including the Norfolk Naval Base, which is the largest naval base in the world. In a study incorporating the use of AIS and RADAR data, researchers in southeastern Virginia found that military vessels had a distinctly different distribution than commercial vessels broadcasting AIS signals (Barco et al. 2012b). Therefore, identified regions of high vessel activity underestimate both the intensity and spatial distribution of vessel activity in the study area. These differences between available data and the real distribution of vessel traffic in the Bay likely explain the fact that model mortality location predictions for a small number of vessel strike turtle strandings did not extensively overlap vessel traffic data (e.g., if the strike was caused by a recreational vessel outside of normal shipping channels; see pseudo-p-values>0.5 in Fig. 8).

4.2 Potential fisheries interactions

This study highlights novel methodology that significantly improved our ability to identify possible locations of sea turtle mortality. However, a complete quantitative assessment of overlaps between anthropogenic activities and these turtle mortality location predictions was limited by the poor spatial and temporal resolution of fishing activity data, as well as the lack of true measures of fishing effort, available for comparisons. This study represents a case where our ability to model the biology (i.e., the drift and decomposition of turtle carcasses) exceeds our ability to interpret model results in light of available anthropogenic observations. For instance, data from VMRC at the waterway level were only accessible as an aggregation over the 6-year study period, prohibiting comparisons on a month-year level. Thus, although there are noted differences in mortality location for the spring peak compared to the remainder of the stranding period, lack of temporal fisheries information makes it impossible to assess differences in potential causes of mortality for the two different time periods. If data on anthropogenic activities, such as fishing effort, were available on spatial and temporal scales pertinent for interpreting individual stranding events (kilometers and a week to a month, respectively), then the overlap between these activities and mortality location predictions could be calculated and one could quantitatively assess which activities were most likely to be causing the mortality.

For some human activities, such as large commercial vessel traffic, detailed information were available and we were able to quantitatively compare and combine these data with mortality predictions. For others, such as the purse-seine menhaden fishery, detailed data exist, but were not publicly available due to industry confidentiality, public image and equity (among fisheries) concerns. OMEGA Protein has operated the sole menhaden reduction plant along the Atlantic coast since 2005 and controls all purse seine vessels (Kirkley 2011). Due to the single participant in this fishery, purse seine fishery location data were not available from VMRC. We requested data on purse seine fishing locations directly from OMEGA Protein, but our data request was denied due to confidentiality concerns and fear of negative repercussions on the image of the industry.

Ultimately, given these various data limitations, we could not definitively rule out any fisheries as a cause of sea turtle mortality. However, preliminary qualitative comparisons can be made between predicted mortality locations and the general distribution of Chesapeake Bay fisheries. The distribution of sink/anchor gillnets, crab pots, and purse seine fishing overlap with both distinct areas of high probability of sea turtle mortality: the lower James River region and bayside Northampton County (Figs. 6, 7, 10-11). Mortality of both loggerhead and Kemp's ridley turtles have been observed within Virginia's gillnet fisheries (Turtle Expert Working Group 2000, Mansfield 2006). Sink gillnets in the nearshore waters of the Bay may interact with bottom-feeding turtles as they forage for food. Crab pots pose a threat to turtles through entanglement with vertical lines, but a side scan sonar survey conducted during the 2006 spring peak of turtle strandings found no entanglements in any of the over 1,600 crab or whelk pots monitored (DeAlteris Associates Inc 2006). Menhaden purse seine effort overlaps with nearly all probable mortality locations, with the notable exception of the region of high mortality likelihood in the James River (Figs. 6-7, 11a). Although results from a 1992 study investigating bycatch in the mid-Atlantic menhaden fishery found no sea turtles captured or even observed during sampling, as well as particularly low bycatch within the Chesapeake Bay fleet, this study observed catch as it was unloaded at the processing plant and did not observe fishing in action (Austin et al. 1994). Measuring turtle interactions with these fisheries is an important avenue to consider for future investigations.

The concentration of haul seine effort almost exclusively in the southwest quadrant of the Bay aligns with predicted mortality locations near the James River and coastline of Hampton County (Figs. 6-7, 10c), while high drift gillnet activity in the southeast region of the Bay coincides with some of the probable mortality locations near Northampton County (Fig. 10a). Minimal overlap is noted between probable mortality locations with whelk pots and traps and staked gillnets (Figs. 6-7, 10d, 11b-c). Because of data pooling, we are unsure, however, if there is temporal as well as spatial overlap between some of these fisheries and sea turtle strandings. Although some likely mortality locations coincide with pound net usage in the northwest Bay, a number of regulatory changes relating to use of modified pound net leaders were made to this fishery in the mid-2000s specifically to reduce turtle mortality (67 FR 41196, 69 FR 24997, 71 FR 36024, 73 FR 68348). Research suggests that these regulations have resulted in a significant reduction of pound net turtle entanglements (DeAlteris and Silva 2007, Silva et al. 2011).

Given the endangered status of sea turtles and potential societal and environmental benefits of addressing threats in a timely fashion, data barriers surrounding the accessibility or collect of fishing data should be lifted. Such information would allow for more complete assessments of potential drivers of sea turtle mortality in Virginia based on the predicted mortality locations highlighted in this study. A combination of state fishery observer coverage, vessel monitoring systems, and increasingly cheap tracking technologies will help address these data deficiencies if funds are made available and fishery engagement can be achieved. Additionally, although observer programs can provide direct evidence of sea turtle-fishery interactions, the state of Virginia lacks an observer program capable of gathering these data for most commercial state fisheries. The availability of data on anthropogenic activities on a finer spatio-temporal scale is key to the ability to conduct more robust identifications of drivers that threaten local sea turtle populations, as well as populations of other protected species.

4.3 Future Research and Mitigation

One study limitation lies in the model assumption that turtle carcasses are freely floating at sea prior to beaching. Sea turtle carcasses in this area have been found entangled within fishing gear, violating this assumption and thus limiting our ability to accurately predict the drift trajectories of these individuals. Several fisheries (i.e. pound net, crab pot) are not checked daily and gear soaks continuously, thus carcasses entangled within these gears can be discovered in a fairly decomposed state and may represent individuals that have been omitted from analyses based on their late condition code. It is also likely that some species and size classes of turtles are more susceptible to types of fishing gears than others. In this study, stranding data of all species and size classes were considered together, thus future research may consider investigating these characteristics separately.

Sea turtle populations in the Chesapeake Bay have increased over time (Mansfield et al. 2004), thus the potential for interactions with fisheries may also increase as turtles become more abundance in Virginia’s waters. Limited information is available on the distribution of foraging sea turtles in the bay, but turtles are typically more abundant in the lower Bay (Mansfield 2006), coinciding with many of the areas of high predicted sea turtle mortality locations. Research suggests that loggerheads tend to stay primarily along channel edges and at river mouths, while Kemp's ridleys are typically found in shallower waters, including seagrass beds (Keinath et al. 1987, Byles 1988). Additional research and information on the distribution of sea turtles in the bay could be useful in further correlating the co-occurrence of sea turtles and human activities.

In addition, analyses in this study greatly relied on temperature-dependent carcass decomposition at the sea surface. As bottom temperatures are lower than surface temperatures, it is possible that cooler conditions will increase bottom time and cause carcasses to decompose less quickly than modeled. This would result in a greater spread in the predicted area of mortality and is an important avenue for future research. Finally, it is also worthwhile to note that the coarse domain of the ChesROMS model may cause inaccurate simulation of pseudo-particles in the coastal area. Using a model with higher horizontal resolution and/or an irregular grid that better represents the complex coastline, such as the Semi-implicit Cross-scale Hydroscience Integrated System Model (Ye et al. 2016, 2018), will be an important improvement to this analysis in the future.

Overall, the ability to quantitatively assess overlap between the predicted sea turtle mortality locations highlighted in this study with anthropogenic activities was severely limited by the lack of fine-scale temporal and spatial resolutions of human use datasets. Nonetheless, the spatio-temporal mortality information obtained from this study provided a starting point for future research and mitigation. Slower vessel speeds are noted as the primary tool to reduce vessel damage to sea turtles (Hazel et al. 2007), as well as marine mammals (Laist and Shaw 2006, Calleson and Frohlich 2007). However, using the results from this study, managers can consider strategies for boaters to reduce speeds in predicted areas with a high likelihood of vessel-strike sea turtle mortality (Fig. 5b) and/or high probability of vessel-turtle interactions during the times of year where turtles are abundant in these waters (Fig. 5c). A finer scale analysis of vessel strikes based on propeller wound size could also be an area of future research. Turtles that were completely bisected were likely to have interacted with larger vessels, and those with multiple, parallel chop wounds were more likely to have interacted with smaller vessels similar to many recreational vessels (Barco et al. 2011). Conducting separate large and small vessel overlap analyses that include locations of marinas and boat ramps popular with recreational vessels as a proxy for location may provide interesting insight into interaction by vessel size.

Similarly, management regulations on commercial fisheries (i.e. time-area closures, limited soak time) or gear modifications should be prioritized in time and space where there is an increased likelihood of interaction with sea turtles. Energetic demands from spring migrations cause turtles to be weaker and in poor health upon entering the Bay, and thus they may be at a greater risk of negative interaction with fishing gear if caught in strong currents (Bellmund et al. 1987, Byles 1988). In addition, it is possible that turtles stranding during the spring peak are weakened from predisposed conditions or cryptic mortality occurring during their migration into the Bay. The cooler water temperature at this time of the year may also slow the metabolism of the migrating turtles, further weakening them. However, by the time mortalities drop near the end of June, water temperatures have increased and turtles are able to forage and move around nets with minimal threat (Lutcavage and Musick 1985, Byles 1988). Therefore, from a temporal standpoint, management efforts may choose to prioritize implementing regulations during this vulnerable spring peak time period.

Predicted mortality locations for turtles classified as having no apparent injuries or where cause of stranding was unable to be assessed were noted to differ within the spring peak compared to the rest of the stranding season, generally shifting from the southwest portion of the Bay to southeastern waters near the bayside of Northampton County (Figs. 6-7). Some maps also show a shift in mortality locations from the lower Bay to more northern Virginia areas of the Bay, consistent with movement of turtles higher into the Bay as the foraging season progresses. Thus, rolling regulations taking into account turtle behavior and distribution during different times of the stranding season could be effective.

Compared to the rest of the stranding season, the elevated number of documented strandings during the spring peak has generally been interpreted as indicative of higher sea turtle mortality rates during this time period. Nevertheless, it is possible that sea turtle mortality is constant throughout the spring and summer stranding season, but turtles are more likely to succumb to decomposition and sink before making landfall during summer, leading to fewer stranding observations. Turtles decompose at a slower rate in cooler waters (Higgins et al. 1995, Santos et al. 2018), with results from Santos et al. (2018) suggesting that turtle carcasses have the potential to drift ~2-5 days longer and ~15-30 km further during the cooler spring peak period compared to those turtles that die during the hot summer months (Fig. 4). This difference in drift duration could explain variability in stranding rates during the spring/summer foraging season, though this hypothesis is difficult to quantitatively assess without knowing more about the spatial distribution of true turtle mortality in the Bay. This hypothesis is also consistent with a small fall peak in strandings (Coles 1999, Barco and Swingle 2014), during which time we predict that drift durations should be significantly longer than during the summer. Therefore, although management actions may prioritize mitigation measures during the spring peak period, strong protection of turtle populations is crucial throughout their entire residency in the Bay.

4.4 Conclusions

The results of this study provide the first attempt to predict sea turtle mortality location based on condition code for Virginia waters of the Chesapeake Bay. Despite data limitations, these results provide ample material for developing focused time-area management measures for mitigating sea turtle mortality in the Bay. Although it is difficult to acquire reliable data on lethal fisheries interactions without trained observers in state fisheries, even rough estimates of causes of mortality and distribution of turtle mortality can provide significant information to inform the development of effective management strategies. Given the protected status of sea turtles and importance of the Chesapeake Bay for hundreds of turtles each year, targeted mitigation measures are urgently needed to ensure the persistence of local turtle populations. Furthermore, as temperatures increase due to climate change, the Bay is predicted to become more favorable to sea turtles (Pike 2014), and, therefore, it is important to identify and manage for anthropogenic causes of mortality now before there has been a significant increase in turtle usage of the Bay. Future research and regulatory management efforts should focus on obtaining more detailed spatio-temporal data on anthropogenic activities so that the list of potential mortality drivers can be mitigated based on quantitative comparisons between the distributions of these activities and mortality location predictions, as well as on assessing probability of landfall for different areas of the Bay so as to estimate absolute turtle mortality rates. The experimental and modeling methods developed here provide a sound basis for these future efforts, as well as a template for assessing and understanding stranding data for sea turtles and other marine megafauna around the globe.

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Tables

Table 1. Linear regression parameters including the y-intercept (Y-int.), slope, and significance (signif.), from the decomposition study, relating temperature with minimum (min) and maximum (max) buoyancy times during condition codes 1-3. Note that the y-intercept has been adjusted by 1 to account for the assumption that code 1 turtles are buoyant for only one day. Condition code 1 and minimum time of buoyancy for condition code 2 is not based on experimental data, thus significance values are not reported.

Time period	Condition code	Y-int.	Slope	Signif.
Min	1	0	0	N/A
Max	1	1	0	N/A
Min	2	1	0	N/A
Max	2	14.99206	-0.41947	<0.01
Min	3	16.7177	-0.5021	<0.05
Max	3	29.3221	-0.9079	<0.01

Figure Legends

Figure 1. (A) Domain of the ChesROMS model. (B) Location of top three areas with reported sea turtle strandings in Virginia from 2009-2014, including 1) the bayside of Northampton County, 2) Norfolk, and 3) Virginia Beach. (C) Expanded view of the lower Chesapeake Bay.

Figure 2. Criteria that must be met for each pseudo-particle to be considered "successful" for a particular stranding event.

Figure 3. Duration of positive buoyancy (days) vs average water temperature (^{o}C) based on results from the experimental decomposition study (Santos et al. 2018). Shaded region represents the time period of positive buoyancy for turtles classified as condition code 1 (green), code 2 (yellow) and code 3 (red). As turtles in condition code 1 were not observed in the study, code 1 turtles were assigned a maximum drift duration of 1 day, and drift duration for turtles with condition codes 2 and 3 were increased by 24 hours relative to raw results from the decomposition study. Individual data points are represented for code 2 turtles and code 3 turtles, with shapes representing the minimum (circle) and maximum (triangle) duration a floating carcass spent in each condition code. Solid lines represent linear regressions.

Figure 4. Boxplot of average (A) drift times (days) and (B) drift distances (km) of modeled particles leading to a condition code 3 stranding event. Results are aggregated by week of the year with gray-colored boxes representing strandings occurring during the spring peak time period. (C) Linear regression of drift time (days) vs drift distance (km).

Figure 5. (A) Vessel density (%) based on vessel location data from the Automatic Identification System for non-federal vessels ≥65ft traveling faster than 4km/hr. (B) Relative particle

density (%) for probability of point of origin for turtle mortality leading to a stranding and classified with probable cause of death as vessel strike. (C) Combined joint probability (%) depicting the overlap between vessel activity and the predicted mortality locations of vessel strike strandings.

Figure 6. Relative particle density (%) for probability of point of origin for turtle mortality leading to a stranding and classified with probable cause of death as no apparent injuries during (A) the spring peak and (B) the remainder of the stranding period. From left to right, panels give results for code 1, code 2 and code 3 strandings, respectively. White circles represent stranding locations and black lines represent Virginia Marine Resource Commission waterways. Note that the scales for codes 2 and 3s have been standardized across time periods.

Figure 7. Relative particle density (%) for probability of point of origin for turtle mortality leading to a stranding and classified with probable cause of death as unable to assess during (A) the spring peak and (B) the remainder of the stranding period. From left to right, panels give results for code 1, code 2 and code 3 strandings, respectively. Code 1 strandings were only reported during the spring peak period. White circles represent stranding locations and black lines represent Virginia Marine Resource Commission waterways. Note that the scales for codes 2 and 3s have been standardized across time periods.

Figure 8. Results from Monte Carlo analysis depicting the probability density function that the model predicted overlap is better ($p<0.05$) at predicting overlap with vessel activity than Monte Carlo randomly distribution null models. Colored lines represent p-values for

condition code 1 (blue), 2 (green), and 3 (red). The black solid line represents a significance value of 0.05.

Figure 9. Relative particle density (%) for probability of point of origin for turtle mortality leading to a stranding classified as condition code 2 with no apparent injuries during the spring peak. Results include (A) 0%, (B) 2%, and (C) 4% of direct wind forcing on carcass drift, as well as (D) the mean of the results with the varying wind forcing values combined. Note that the color scales have been standardized.

Figure 10. Harvest (hundreds of thousands of pounds) by (A) drift gillnets, (B) sink/anchor gillnets, (C) haul seines, (D) whelk pots and traps and (E) crab pots and traps gear. Data was obtained from the Virginia Marine Resource Commission and aggregated over 2009-2014.

Figure 11. (A) Menhaden purse seine sets locations (red) aggregated over 2011-2013, obtained from the 2015 Atlantic Menhaden Stock Assessment Report. Length (km) of net per 5 km by 5 km grid cell for (B) staked gill nets and (C) pound nets based on point locations obtained from the Virginia Marine Resource Commission website for 2017, the current license year at the time of the study.

Figures

Figure 1

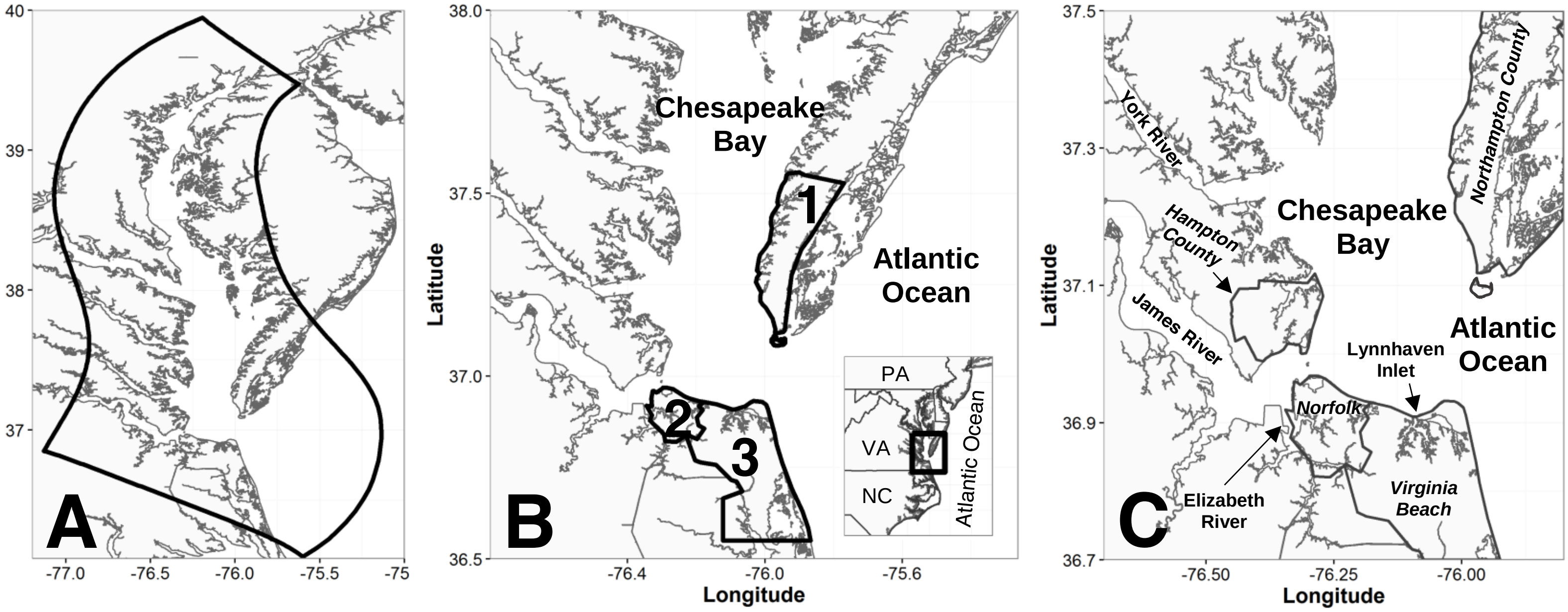
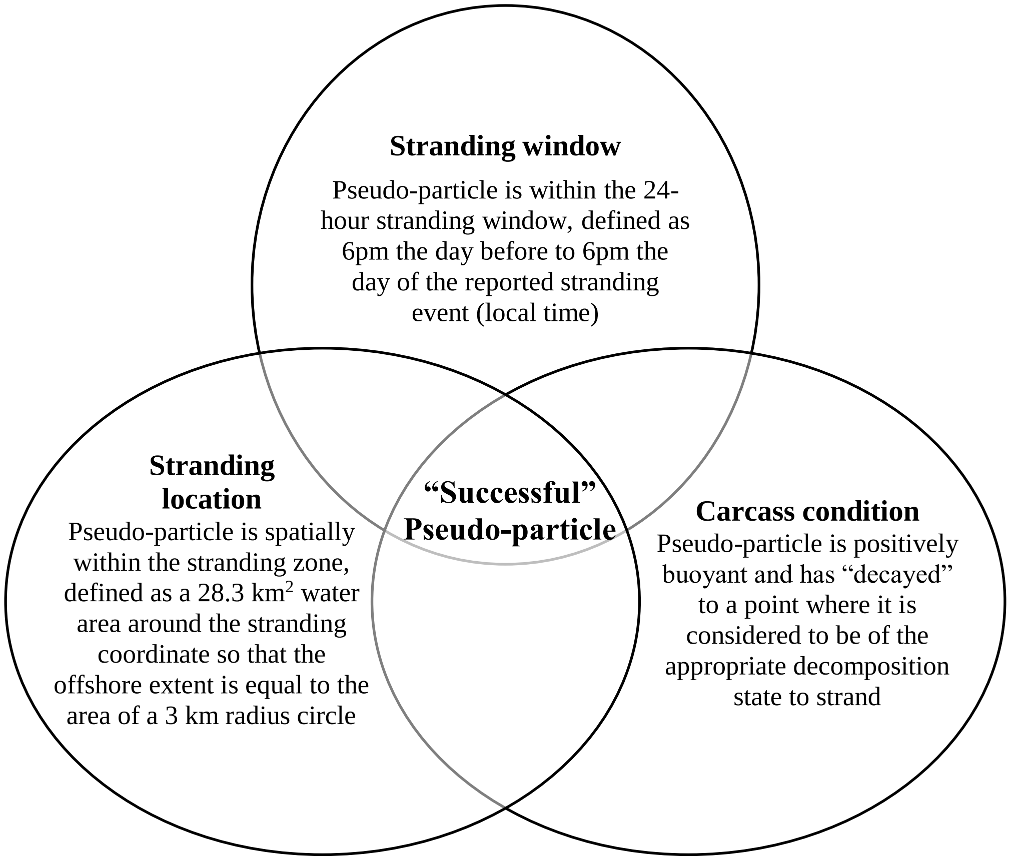
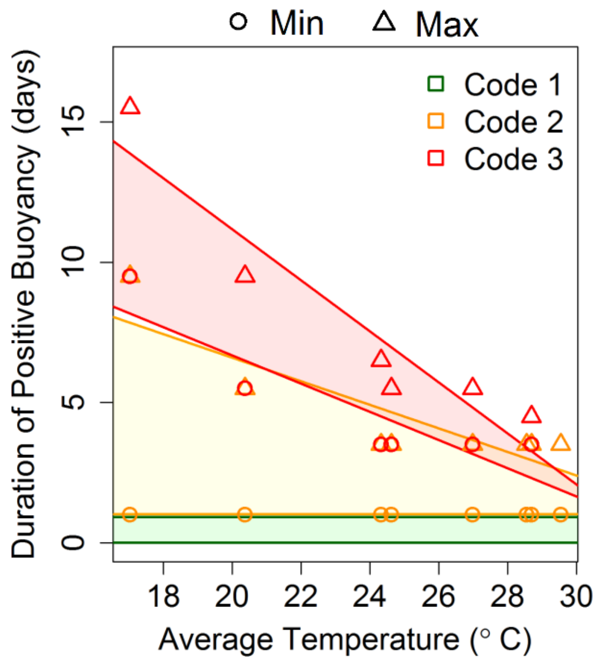


Figure 2


Stranding window
Pseudo-particle is within the 24-hour stranding window, defined as 6pm the day before to 6pm the day of the reported stranding event (local time)
Stranding location
Pseudo-particle is spatially within the stranding zone, defined as a 28.3 km^2 water area around the stranding coordinate so that the offshore extent is equal to the area of a 3 km radius circle
“Successful” Pseudo-particle
Carcass condition
Pseudo-particle is positively buoyant and has “decayed” to a point where it is considered to be of the appropriate decomposition state to strand

1011 Figure 3



1012 Figure 4

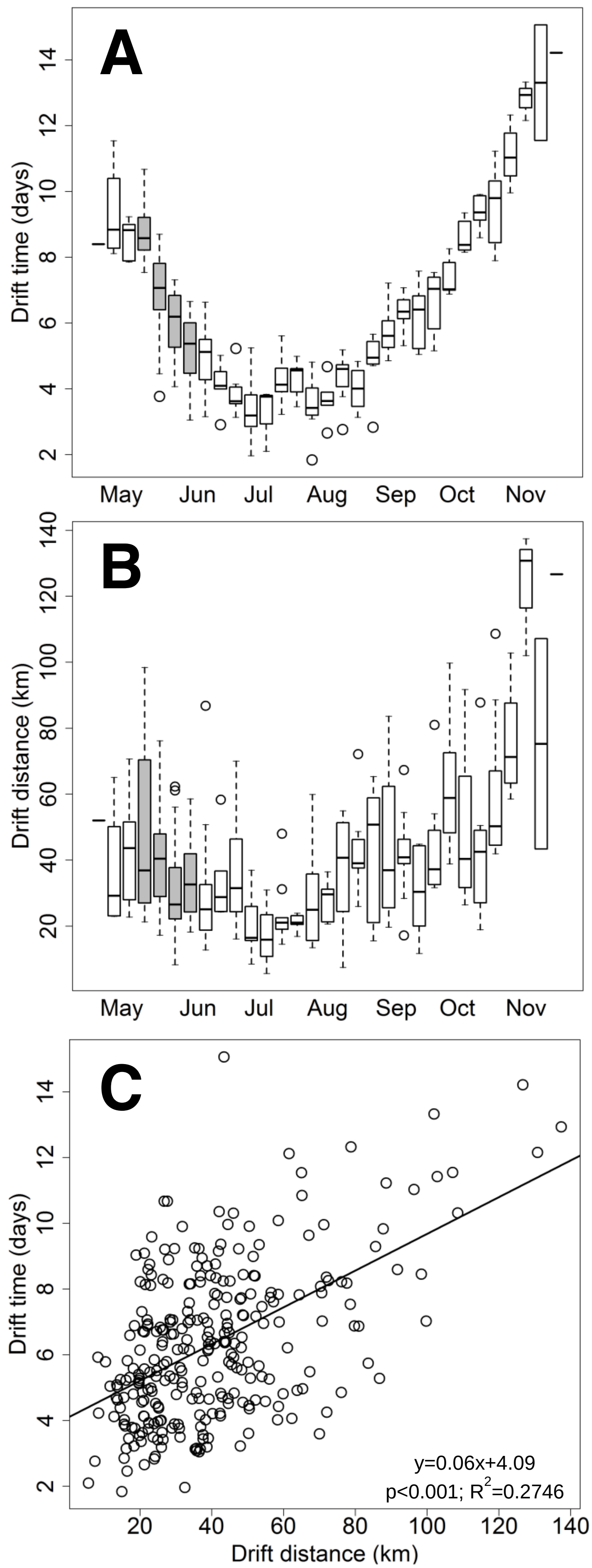


Figure 5

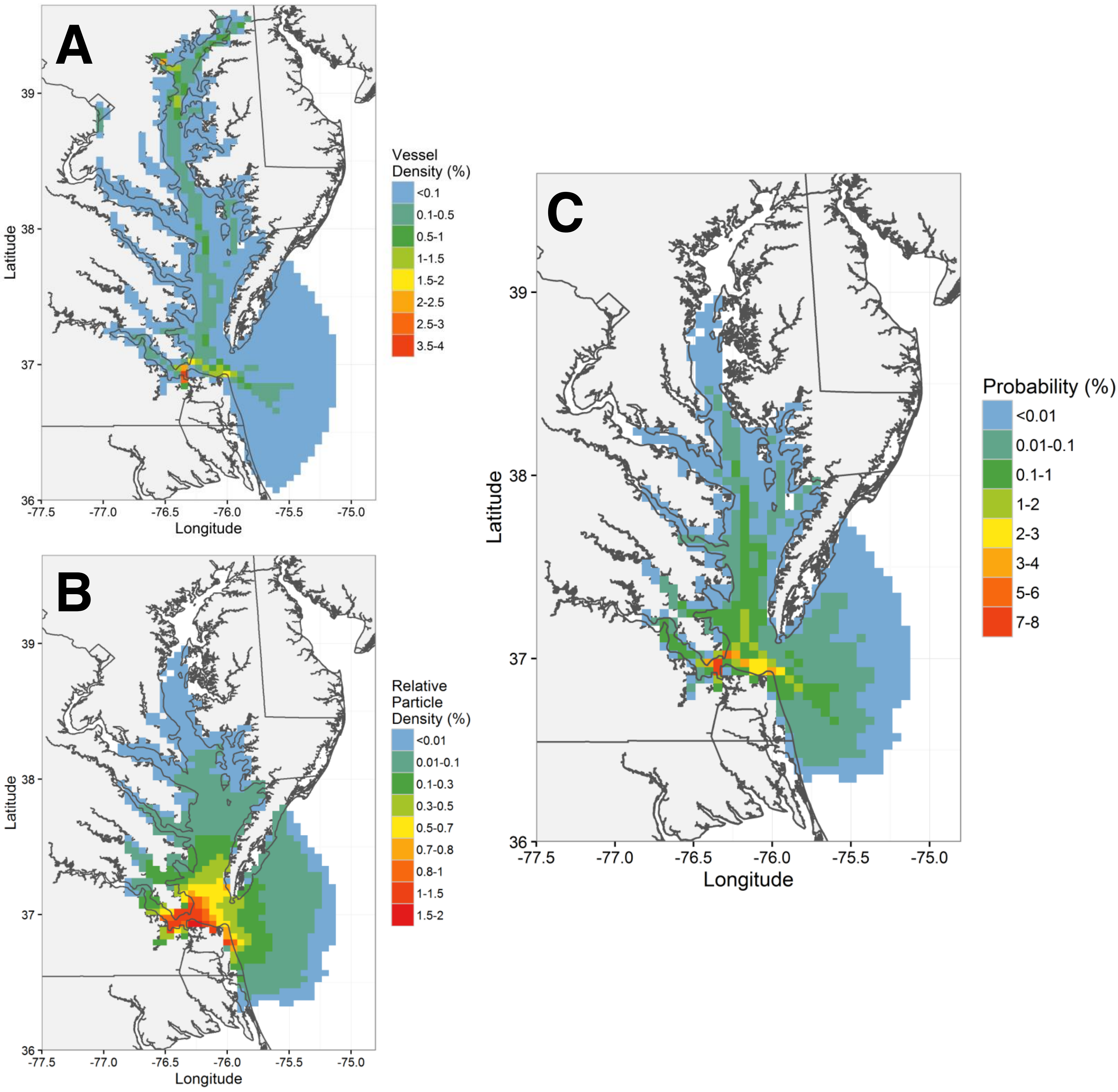
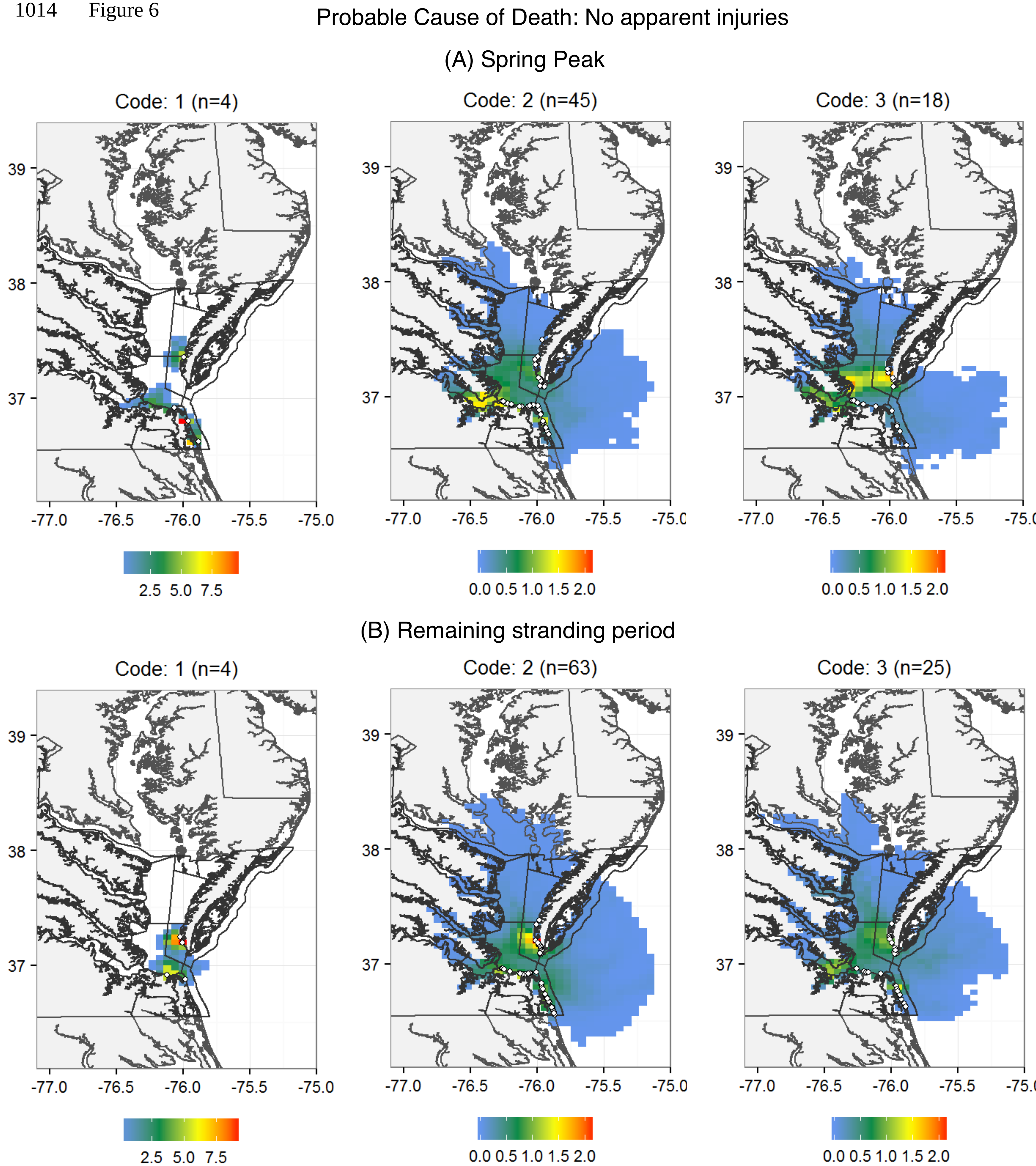
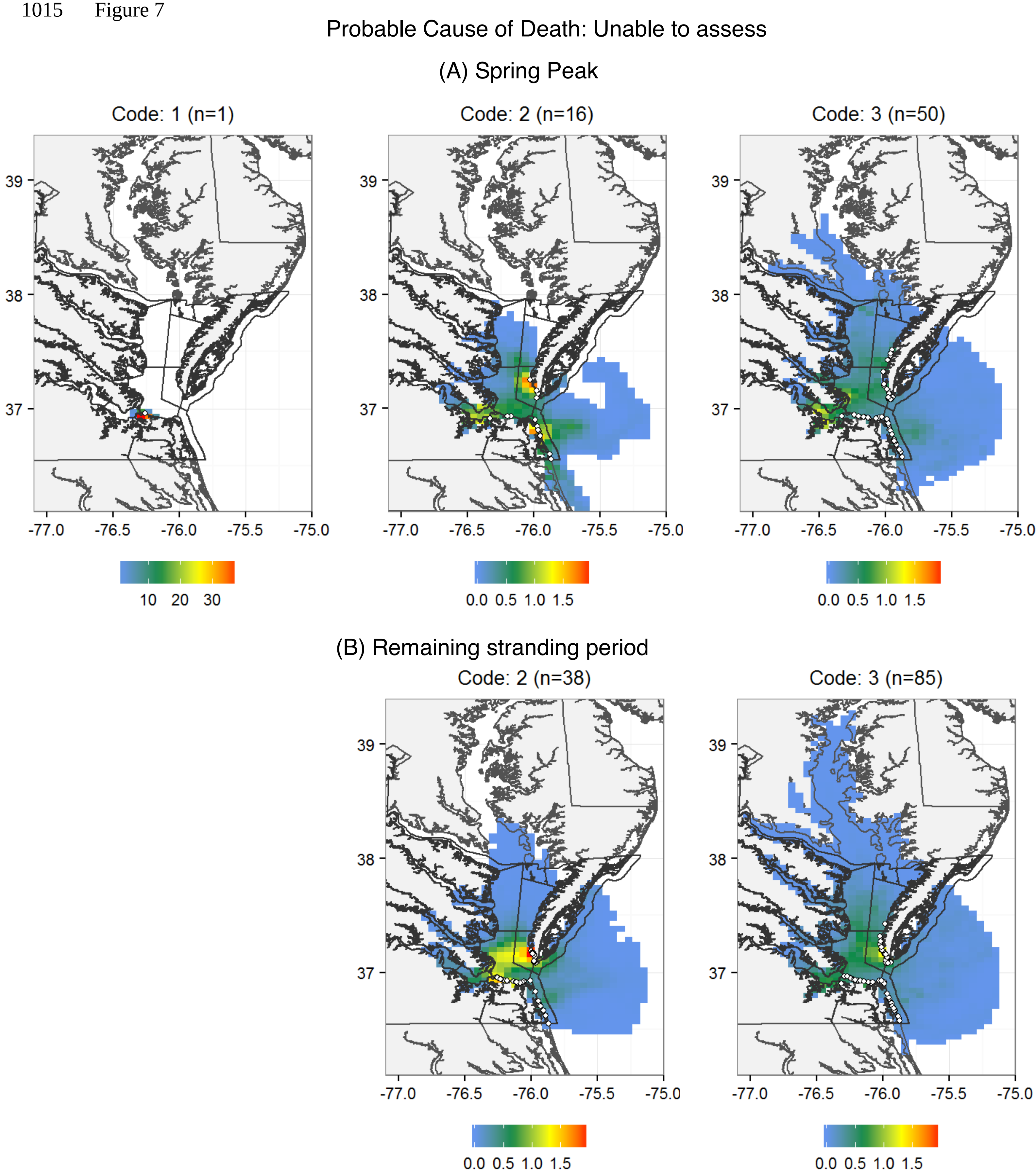

A
Vessel Density (%)
<0.1
0.1-0.5
0.5-1
1-1.5
1.5-2
2-2.5
2.5-3
3.5-4
Latitude
Longitude
B
Relative Particle Density (%)
<0.01
0.01-0.1
0.1-0.3
0.3-0.5
0.5-0.7
0.7-0.8
0.8-1
1-1.5
1.5-2
C
Probability (%)
<0.01
0.01-0.1
0.1-1
1-2
2-3
3-4
5-6
7-8
36
37
38
39
-77.5
-77.0
-76.5
-76.0
-75.5
-75.0

Figure 6

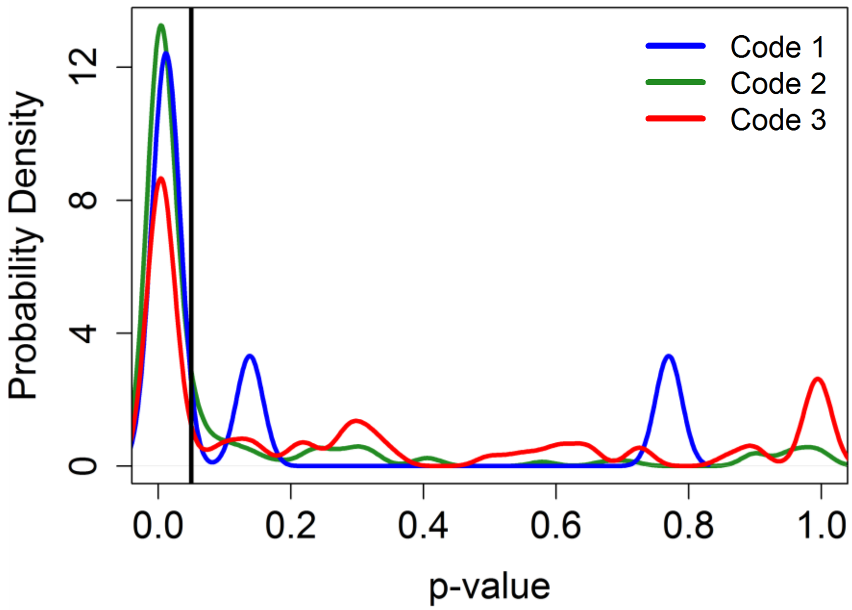


Probable Cause of Death: No apparent injuries
(A) Spring Peak
Code: 1 (n=4)
Code: 2 (n=45)
Code: 3 (n=18)
39
38
37
-77.0
-76.5
-76.0
-75.5
-75.0
2.5 5.0 7.5
0.0 0.5 1.0 1.5 2.0
0.0 0.5 1.0 1.5 2.0
(B) Remaining stranding period
Code: 1 (n=4)
Code: 2 (n=63)
Code: 3 (n=25)
2.5 5.0 7.5
0.0 0.5 1.0 1.5 2.0
0.0 0.5 1.0 1.5 2.0

1015 Figure 7



1016 Figure 8



1017 Figure 9

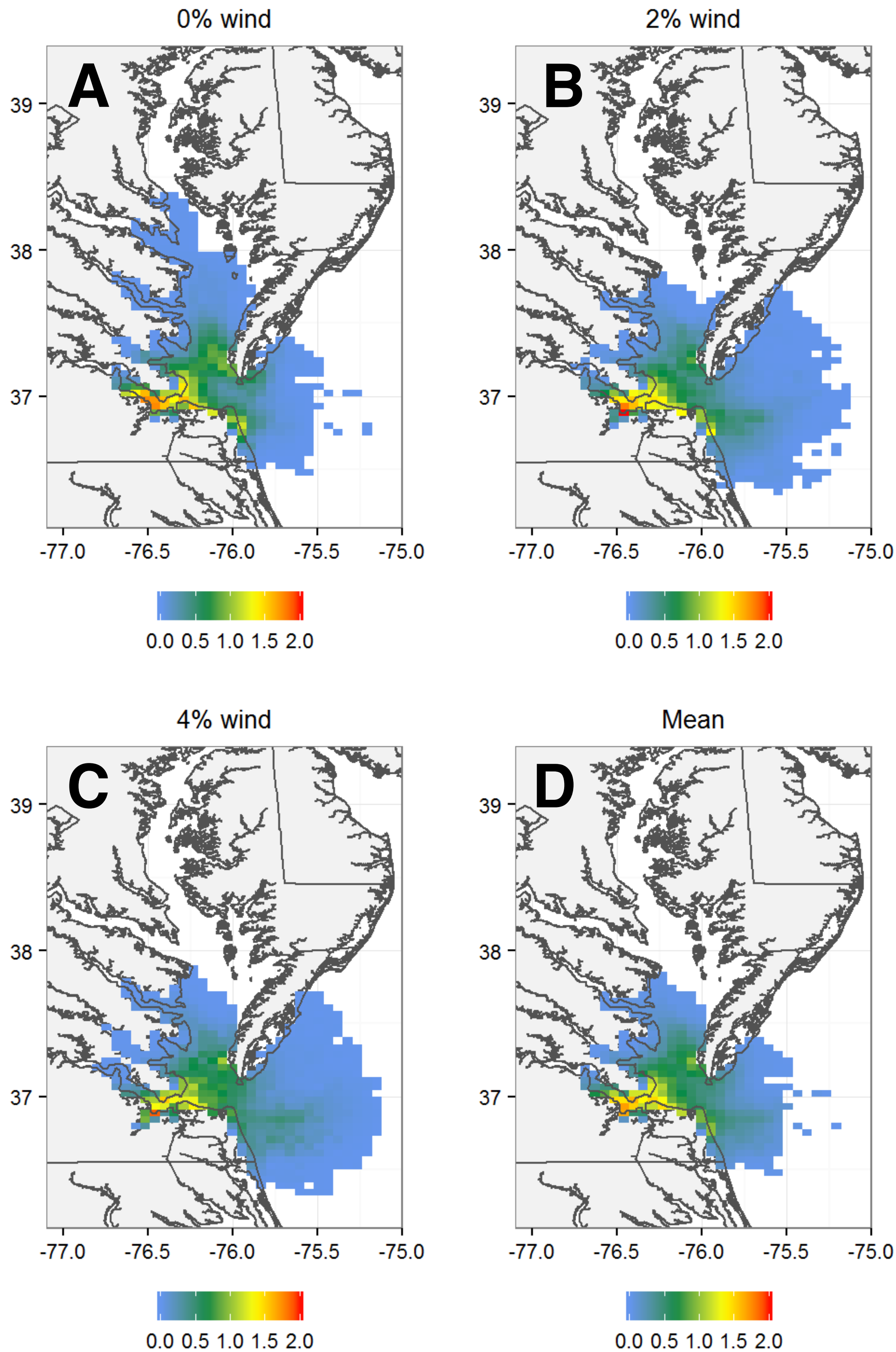
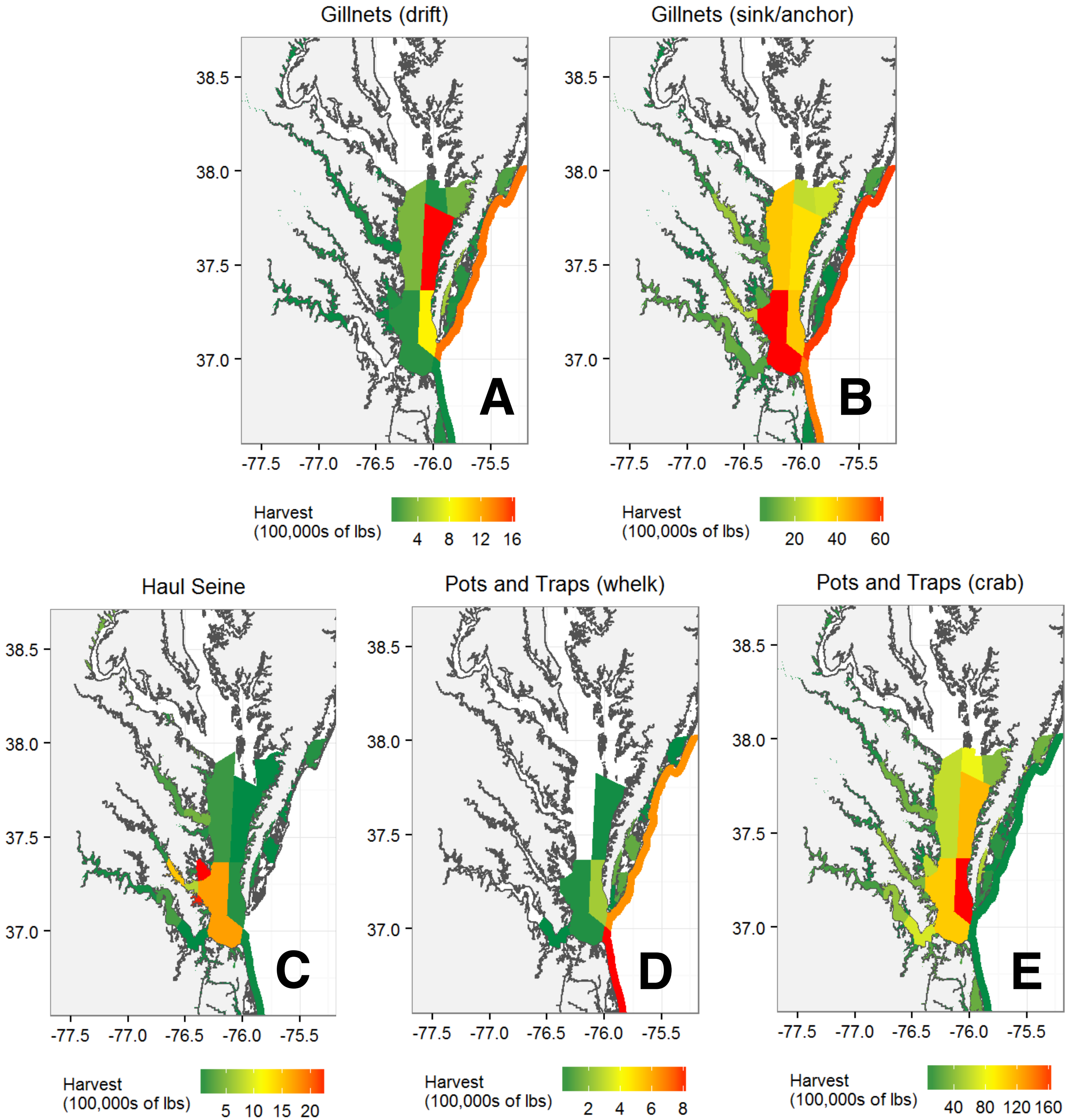
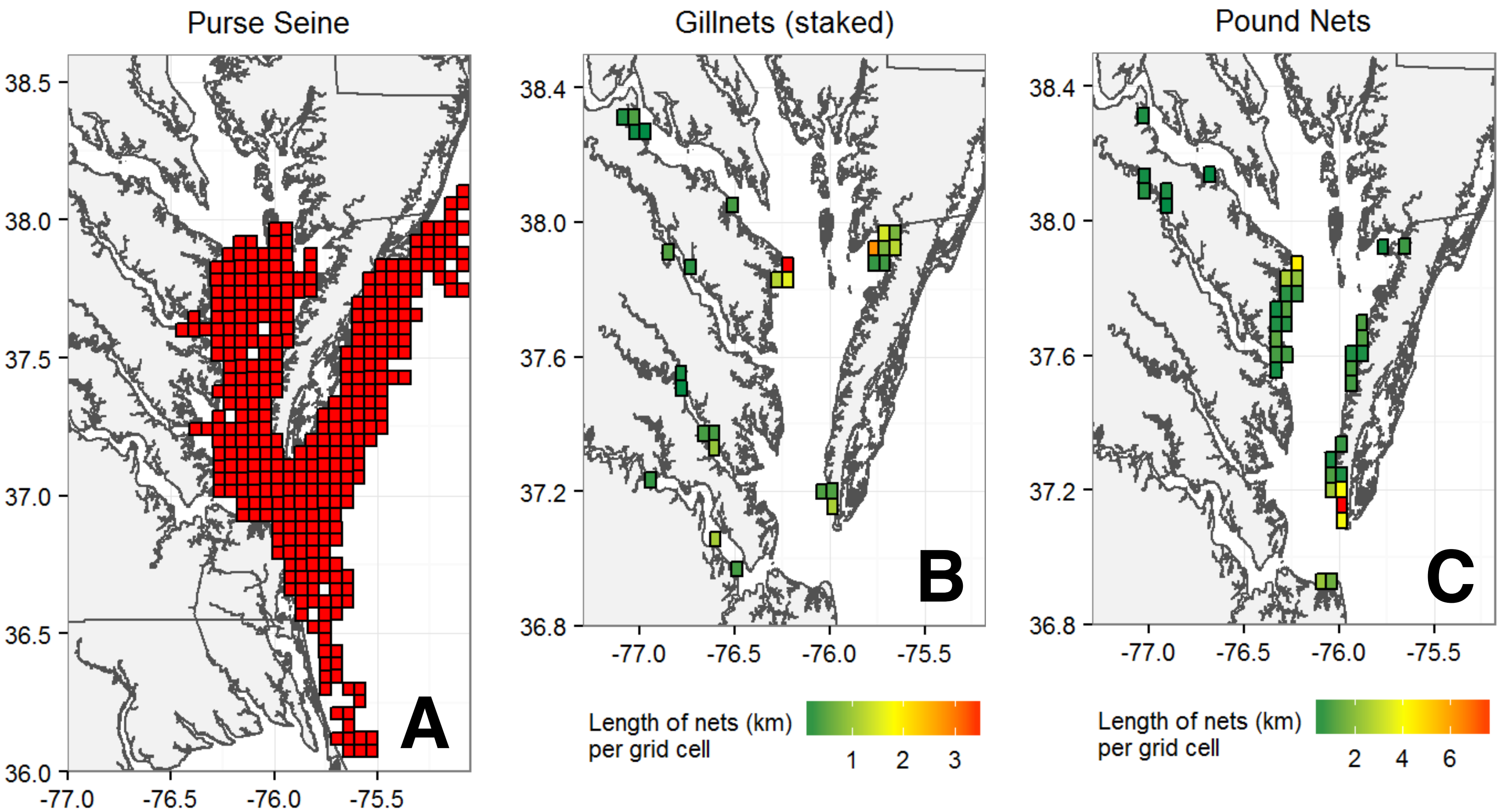


Figure 10


Gillnets (drift)
38.5
38.0
37.5
37.0
-77.5
-77.0
-76.5
-76.0
-75.5
A
Harvest
(100,000s of lbs)
4 8 12 16
Gillnets (sink/anchor)
B
Harvest
(100,000s of lbs)
20 40 60
Haul Seine
C
Harvest
(100,000s of lbs)
5 10 15 20
Pots and Traps (whelk)
D
Harvest
(100,000s of lbs)
2 4 6 8
Pots and Traps (crab)
E
Harvest
(100,000s of lbs)
40 80 120 160

1019 Figure 11



Appendix

Figure A.1. Frequency of all reported stranding events per week of the year for 2009-2014. Shaded areas represent the spring peak (red; 3-5 weeks) and the remainder of the stranding period (green; 19-23 weeks).

Figure A.2. Virginia Marine Resource Commission waterways (black outline) and system (color) identification.

Figure A.3. Harvest (hundreds of thousands of pounds) by (A) staked gillnet and (B) pound net gear. Data was obtained from the Virginia Marine Resource Commission and aggregated over 2009-2014.

Figure A.1

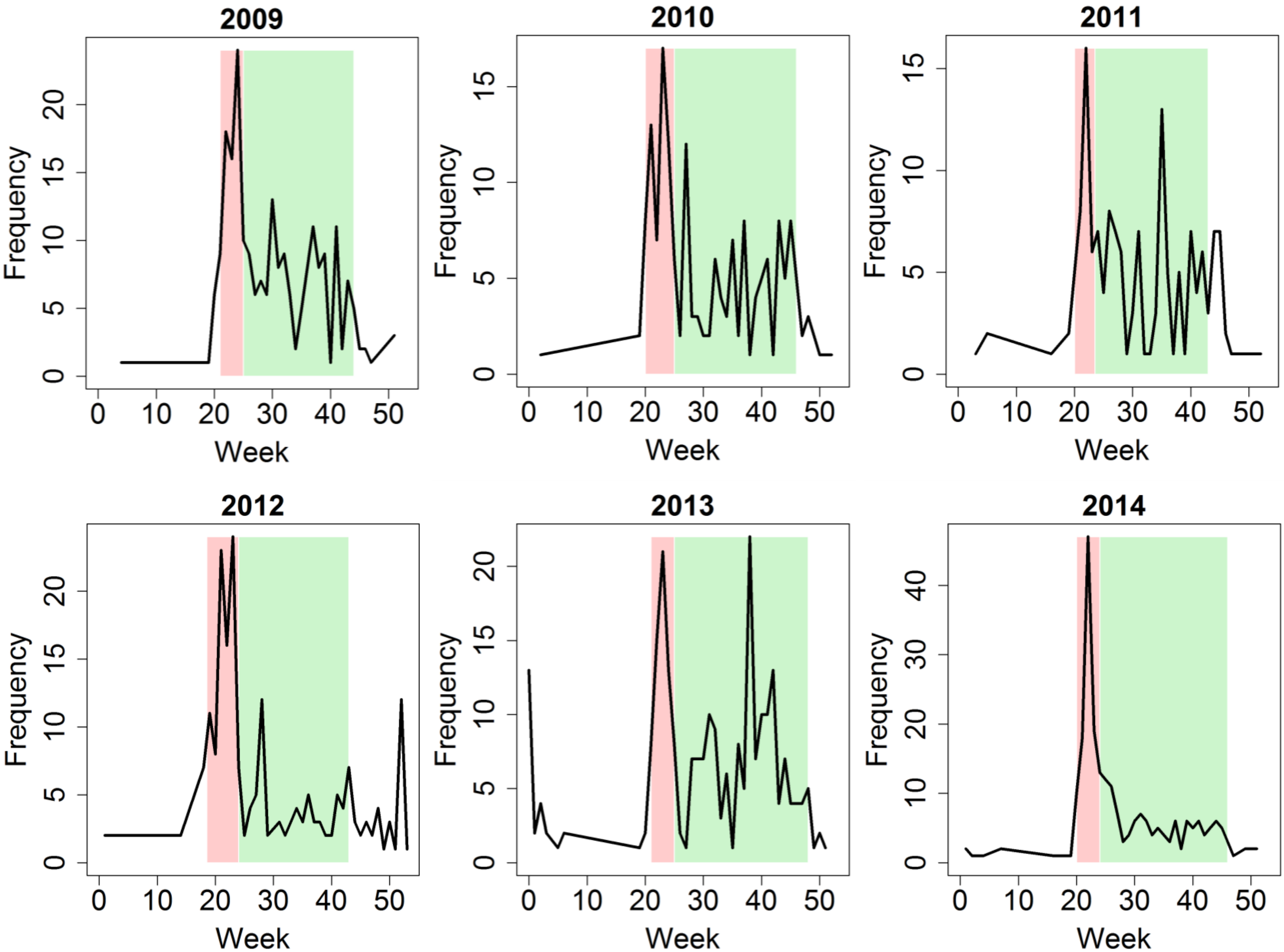


Figure A.2

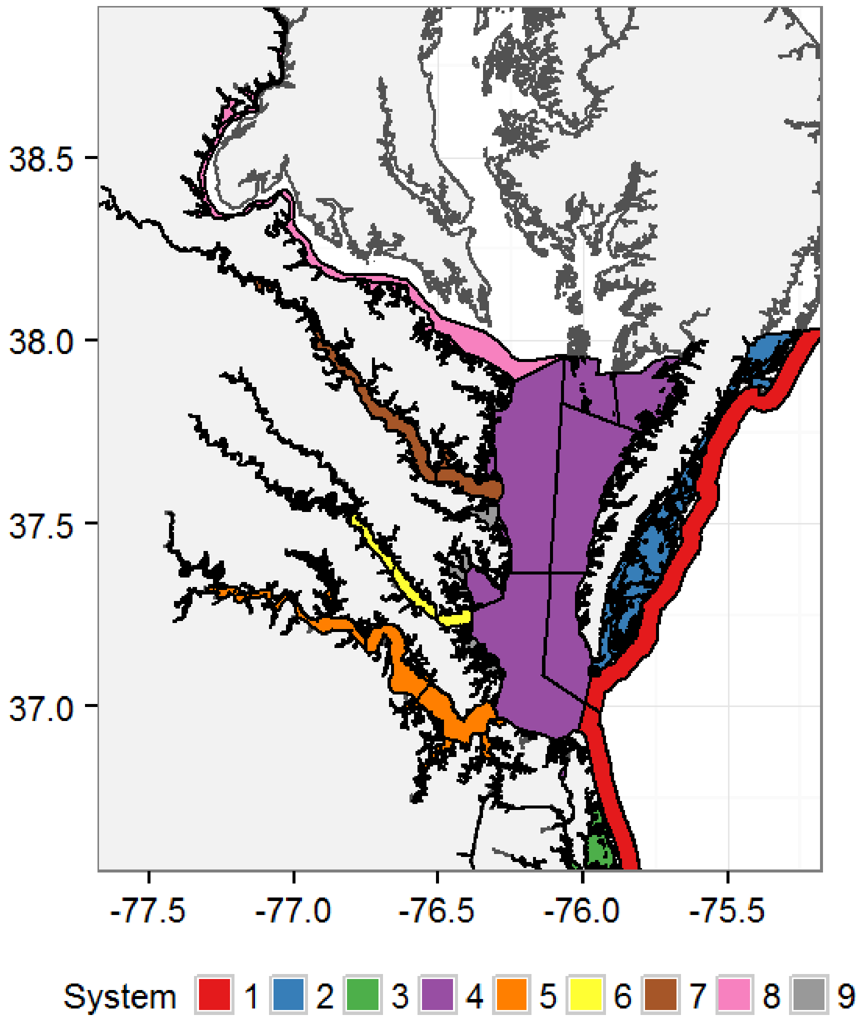
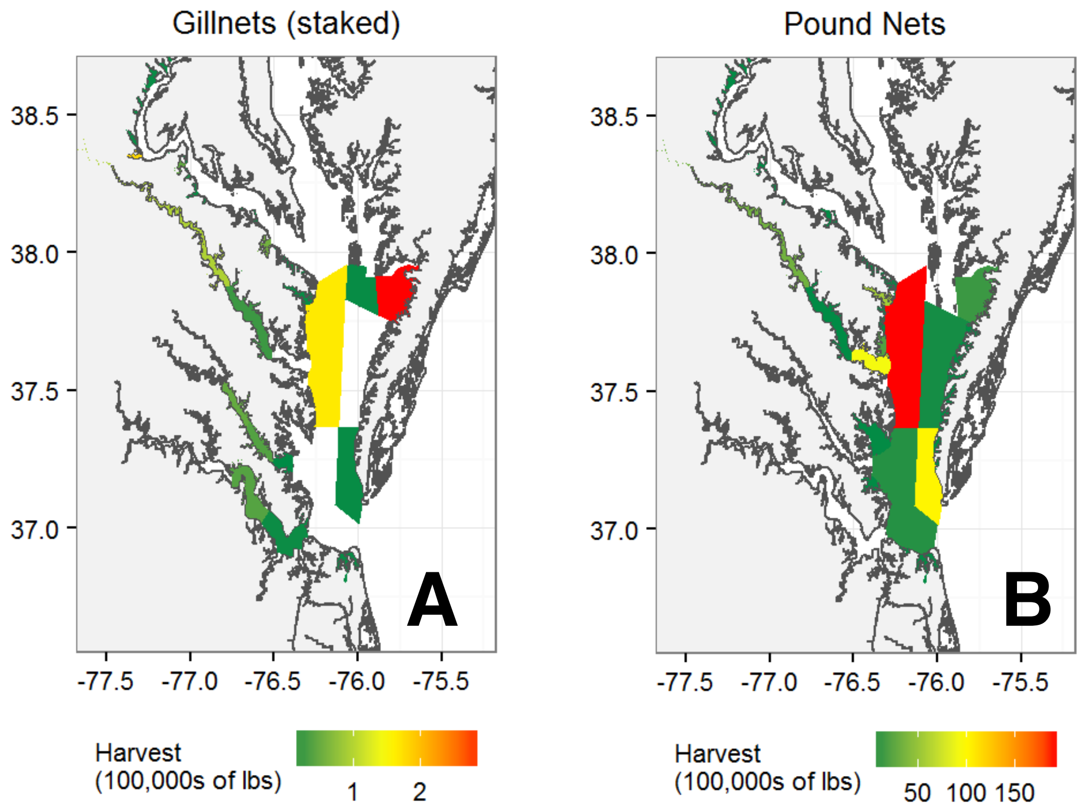


Figure A.3

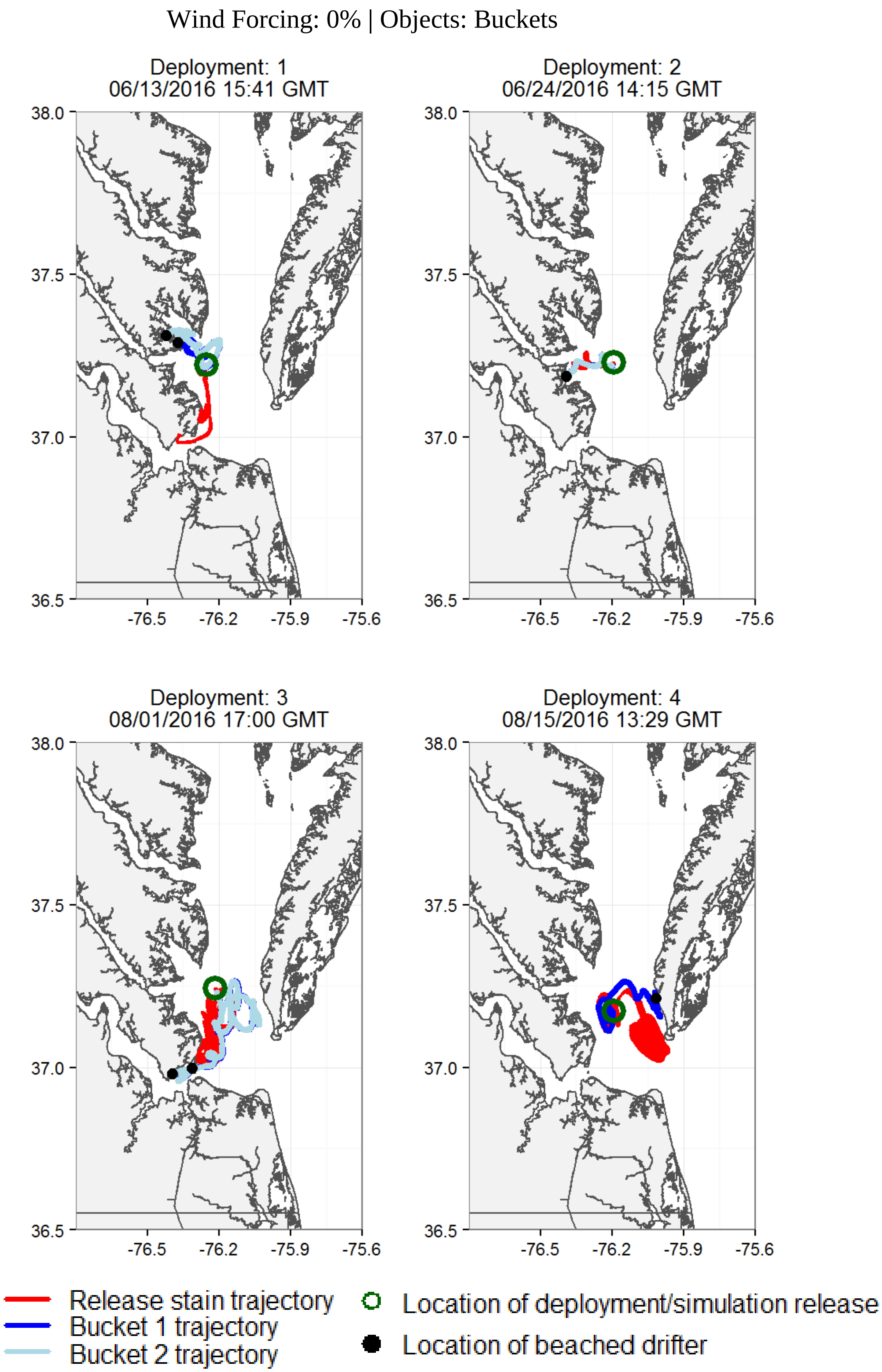

Gillnets (staked)
38.5
38.0
37.5
37.0
-77.5
-77.0
-76.5
-76.0
-75.5
A
Harvest
(100,000s of lbs)
1
2
Pound Nets
38.5
38.0
37.5
37.0
-77.5
-77.0
-76.5
-76.0
-75.5
B
Harvest
(100,000s of lbs)
50
100
150

Supplement materials

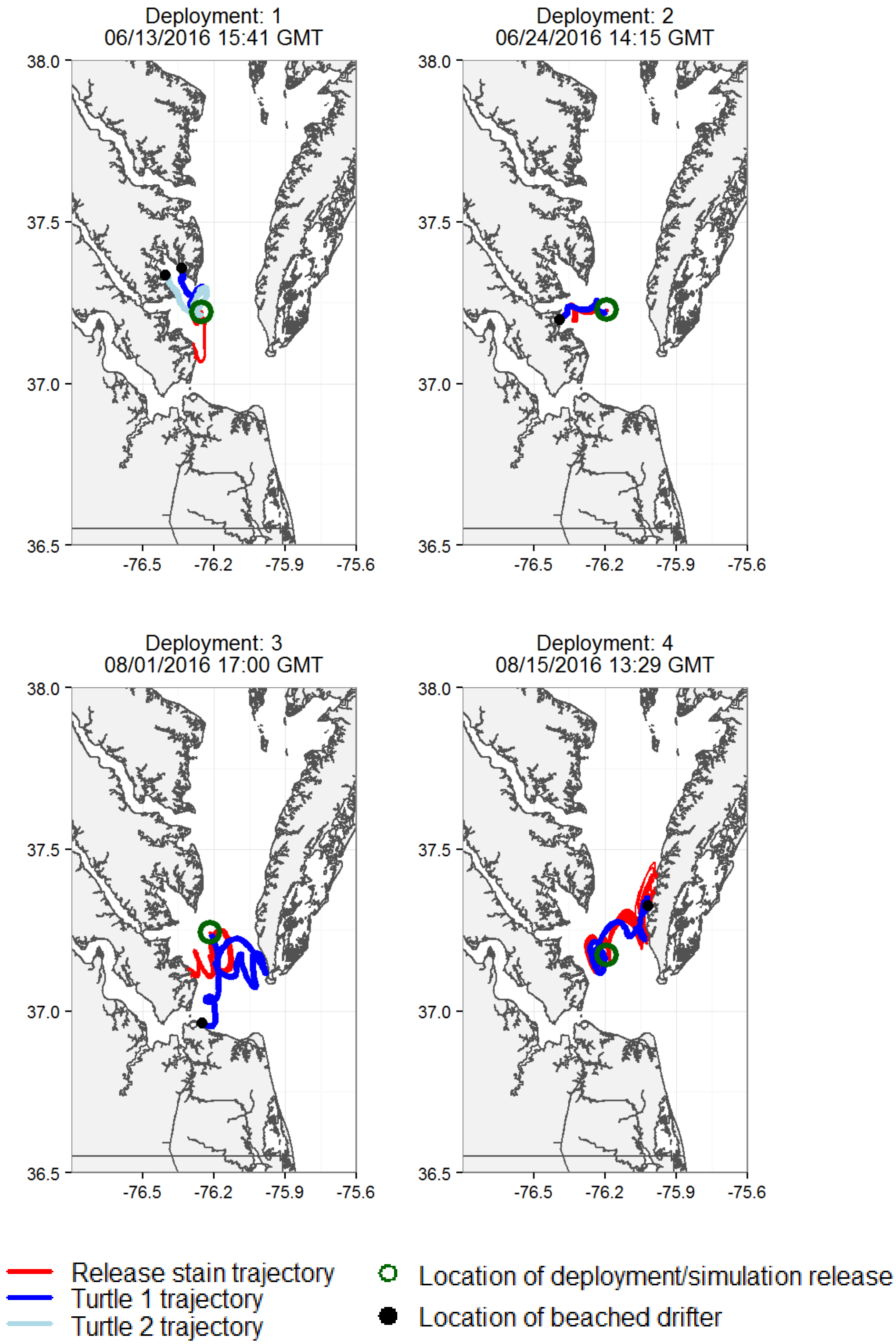
Drift simulations to compare drift trajectories from modelled pseudo-particles to experimentally-observed data

Model simulations were performed using a "release stain" strategy in Ichthyop (Lett et al. 2008). Ten-thousand pseudo-particles were released in the ChesROMS model at the times of the releases of the pseudo-carcasses and buckets shown in Figure 6 of Santos et al. (2018). Particles werre released within a 3 km buffer of the release positions. Although ChesROMS model skill has not been previously demonstrated for surface velocities using drifters, it has undergone extensive skill assessment (Feng et al. 2015, Irby et al. 2016, Luettich et al. 2017, Moriarty 2017, Irby et al. 2018, Da 2018, Da et al. 2018). Furthermore, we note that the physical circulation is very well represented within the Bay, as is evidenced by the high skill of the model in reproducing observed salinity.

The movement of the pseudo-particles were tracked as they moved forward until the moment in time that the actual drifters beached. Drift trajectories of modeled pseudo-particles were compared to the drift pathways and stranding locations of the actual drifters. When comparing with the movements of the bucket drifters, no wind forcing was added to the ChesROMS currents, while in the case of the pseudo-carcasses, winds were added at 2% (the value closest to observed values in Santos et al. (2018) and the wind forcing that is (primarily) used in this study). Simulations were repeated for each of the four drifter deployments.


Wind Forcing: 0% | Objects: Buckets
Deployment: 1
06/13/2016 15:41 GMT
Deployment: 2
06/24/2016 14:15 GMT
Deployment: 3
08/01/2016 17:00 GMT
Deployment: 4
08/15/2016 13:29 GMT
38.0
37.5
37.0
36.5
-76.5
-76.2
-75.9
-75.6
Release stain trajectory
Bucket 1 trajectory
Bucket 2 trajectory
Location of deployment/simulation release
Location of beached drifter

Wind Forcing: 2% | Objects: Pseudo-turtles



Overall, results from the stain simulations compared well with actual drift trajectories. The particle tracks have the same overall form as the drift pathways of the objects, following the same tidal oscillations and overall direction of transport, and generally arriving close to the stranding area in three of the four deployments. However the model does miss some complexity in transport, particularly for Deployment 1. The release time of deployment 1 occurred at approximately 15:41 GMT on June 13, 2016, right around the time that the tides in the area were turning after experiencing a high tide at 15:44 GMT, as reported by a nearby buoy (National Oceanographic and Atmospheric Administration's Tidal Current Predictions (http://tidesandcurrents.noaa.gov/) for station ACT5406 York River Entrance Channel, NW end). It is possible that the close proximity of the drifter release to the changing of the tides could have caused some of the observed discrepancies between model results and drifter experiments for this release event. For real drifters released at essentially the same time and place, we observed relatively rapid separations between paired identical drifters during slack tides. The proximity of this release to changing tides combined with relatively small spatial or temporal misalignments between the model and real currents could therefore explain the observed discrepancies.

To reduce the importance of poor alignment between the model and true current variability for any individual stranding, the approach followed in this paper have been to aggregate over many stranding events. We never present results for a single stranding or specific transport times, but instead look at averages over many events. It is also worthwhile to note that the coarse domain of the ChesROMS model may cause inaccurate simulation of particles in the coastal area. Using a model with higher horizontal resolution and/or an unstructured grid that better represents the complex coastline, such as the Semi-implicit Cross-scale Hydroscience

Integrated System Model (Ye et al. 2016, 2018), is an important avenue for future improvement to our model

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