ENVIRONMENTAL APPENDIX C

ATTACHMENT C-1
AQUATIC RESOURCES ASSESSMENTS

ENVIRONMENTAL MONITORING OF MOBILE BAY AQUATIC RESOURCES AND POTENTIAL IMPACTS OF THE MOBILE HARBOR GENERAL REEVALUATION REPORT

Jacob F. Berkowitz, Safra Altman, Kevin Reine, Dara Wilbur, Michael E. Kjelland, Terry Gerald, Sung-Chan Kim, Candice D. Piercy, Todd M. Swannack, William T. Slack, Jack Killgore Jr., Kevin D. Philley, Nathan R. Beane, Christina L. Saltus, Matt T. Balazik, Tyler A. Keys, and Corey J. Trahan

U.S. Army Engineer Research and Development Center, Vicksburg, MS 39180, USA.

April, 2019

Prepared for:

U.S. Army Corps of Engineers
Mobile District
109 St. Joseph St.
Mobile, AL 36602

by

Engineer Research and Development Center
3909 Halls Ferry Road
Vicksburg, MS 39056

HX5, LLC
212 Eglin parkway, SE
Fort Walton Beach, FL 32548

Abstract

The following report assesses potential impacts to aquatic resources resulting from proposed navigation channel expansion activities within Mobile Bay, Alabama. The report was conducted for the U.S. Army Corps of Engineers (USACE) – Mobile District, supporting development of a supplemental Environmental Impact Statement. Specifically, changes in water quality and hydrodynamics are evaluated for potential impacts to benthic macroinvertebrates, wetlands, submerged aquatic vegetation, oysters, and fish. The assessment includes extensive characterization of baseline conditions, followed by evaluation of estimated post project conditions related to aquatic resource habitat (e.g., changes in salinity, dissolved oxygen). Additionally, an analysis of potential impacts related to a 0.5 m sea level rise scenario are evaluated. Results suggest that no substantial impacts in aquatic resources within the study area are anticipated due to project implementation, as the area of greatest potential changes to environmental conditions are already adapted to natural shifts in salinity (and other factors) as well as conditions resulting from the existing navigation channel. Although sea level rise has the potential to alter aquatic resource habitats with Mobile Bay, additional impacts related to project implementation remain negligible under the 0.5 m sea level rise scenario. The sections below provide detailed information regarding the study design, execution, and results.

Executive summary

The U.S. Army Corps of Engineers (USACE) – Mobile District is evaluating potential expansion of the Mobile Bay navigation channel, including deepening and widening activities. These structural modifications to the navigation channel can potentially alter circulation and transport processes within Mobile Bay, which may impact aquatic resources. An assessment of aquatic resources was conducted to evaluate potential changes in habitat related to five aquatic resource categories identified by an interagency team including: benthic macroinvertebrates, wetlands, submerged aquatic vegetation (SAV), oysters, and fish. The approach included analysis of baseline conditions, onsite analysis, and evaluations of predicted post-project conditions generated using robust hydrodynamic and water quality models. The following assessment describes baseline characterization and distribution of existing resources, followed by analysis of projected post-project conditions (e.g., salinity, dissolved oxygen) with the potential to impact the presence and productivity of each target aquatic resource. A 0.5 m sea level rise scenario is also evaluated in accordance with current USACE guidance.

The benthic macroinvertebrate assessment results indicate that benthic macrofaunal assemblages transition from polychaete-rich assemblages in the estuary to being dominated by insects in freshwater habitat. Expected post project conditions suggest mean bottom salinity increases 1 -3 ppt. The greatest salinity increases are projected to occur within the transitional and estuarine zones where benthic macrofaunal assemblages are dominated by polychaete worms that are well adapted to experiencing salinity fluctuations that occur during tidal exchanges. Impacts of harbor deepening on benthic macrofauna due to salinity intrusion are predicted to be negligible, with no effects on higher trophic levels, such as fish, because prey availability and distributions are unlikely to be affected.

The wetland assessment identified >40 habitat types occurring across a wide range of salinity regimes. Projected changes in water quality will not exceed wetland plant community mortality or productivity thresholds within the study area, suggesting that impacts to wetlands are not expected. While the 0.5 m sea level rise scenario will result in increased wetland inundation within portions of Mobile Bay, implementation of the project is expected to have limited additional impacts on wetlands.

The SAV assessments identified > 600 acres of sea grasses encompassing 55 community types. Expected post project conditions suggest that > 93% of SAV communities will not experience substantial salinity increases. Where potential salinity thresholds may be exceeded, affected species are dominated by invasive species (Eurasian watermilfoil) or occur during short duration (<7 day) events. Dissolved oxygen levels remain within SAV tolerance limits across all scenarios examined.

Simulated oyster larvae movement through integrated hydrodynamic, water quality, and larval tracking modeling was successfully implemented. Dissolved oxygen levels stay well above the minimum oyster tolerance threshold for simulated scenarios with and without sea level rise. Salinity values in regions of the bay were below or above mortality threshold values, but project implementation did not increase the number of oysters exposed to these exceedingly high or low salinities. Additionally, the oyster model results do not project an increase in larvae flushing out of Mobile Bay or substantial changes in larval distribution due to project implementation.

For the fisheries assessment, a total of 2,097,836 individuals representing 162 species were recorded and used in the analysis, which include five salinity tolerance guilds ranging from freshwater to marine habitat conditions. The freshwater entering estuary salinity guild is likely the most susceptible to changes in salinity due to project construction. However, the salinity range this guild occupies suggests that differences between baseline and project alternative with and without sea level rise would have to be much greater than the model outputs suggest to have a significant impact on this guild's abundance. Given these relationships, impacts to the Mobile Bay fishery are not expected.

Chapter 1: Project purpose and overview

Chapter 1: Introduction

1.1 Purpose.

The purpose of this study is to document the wide array of aquatic resources within Mobile Bay and investigate potential changes in natural resource habitat and productivity associated with proposed deepening and widening of the Mobile navigation channel. Aquatic resources evaluated will include benthic macroinvertebrates, wetlands, submerged aquatic vegetation (SAVs), oysters, and fish.

1.2 Background

The study site occupies Mobile Bay, Alabama, which is formed by the Fort Morgan Peninsula to the east and Dauphin Island, a barrier island on the west. Mobile Bay is 413 square miles (1,070 km2) in area (Figure 1.1). It is 31 miles (50 km) long with a maximum width of 24 miles (39 km). The deepest (75 feet, 23 m) areas of the Bay are located within the federal navigation channel, which serves Alabama's only port for ocean-going vessels. The average depth of the bay is around 10 feet (3 m). The Mobile Bay watershed is the sixth largest river basin in the United States and the fourth largest in terms of streamflow. It drains water from three-fourths of Alabama as well as portions of Georgia, Tennessee, and Mississippi into Mobile Bay. Both the Mobile River and Tensaw River empty into the northern end of the Bay. Several smaller rivers: Dog River, Deer River, and Fowl River, on the western side of the Bay and the Fish River on the eastern side also empty into the Bay, making it an estuary. A feature of all estuaries is a transition zone, where the freshwater from the rivers mixes with the tidally-influenced salt water of the Gulf of Mexico.

Figure 1.1 Overview of Mobile Bay within southern Alabama. The lines indicate the location of the navigation channel and proposed work location examined within the current report. Potential dredged material placement locations (not discussed herein) are shaded in grey.

The principal navigation problem is that vessels are experiencing delays leaving and arriving at the port facilities, and their cargo capacities are limited. This problem is a result of the increasing number and size of vessels entering and departing the port. In the last five years, the Alabama State Port Authority (ASPA) added two new facilities at the lower end of the Mobile River (at the upper portion of Mobile Bay). One is the Choctaw Point container terminal and the other is the Pinto Island Terminal. Both facilities increased the amount of traffic into the port. The existing channel depths and widths limit vessel cargo capability and also restrict many vessels to one-way traffic and light loading. Therefore, evaluating deepening and widening of the Bar and Bay channels up to their fully authorized dimensions is being proposed to alleviate harbor delays and improve cargo capacity. These structural modifications to the navigation channels can potentially alter circulation and transport within Mobile Bay, which may impact aquatic resources. Potential impacts include changes in salinity, sediment transport, and water quality parameters related to aquatic resources in the region.

As part of an investigation of potential environmental effects of widening and deepening of the federal navigation channel, the U.S. Army Corps of Engineers Mobile District requests the assistance of the U. S. Army Engineer Research and Development Center, Environmental Laboratory (ERDC-EL) to assess potential impacts to aquatic resources in locations potentially impacted by saltwater intrusion and other factors. Characterizations of baseline aquatic resources in estuarine, transitional, and freshwater environments are important to establish prior to channel deepening and potential impacts from saltwater intrusion. A key component of the current study is to document changes to aquatic resources along the salinity continuum moving upriver and estimates how far upriver changes may occur after the navigation channel is widened and depended to its new authorized depth. Elevated salinities upriver and in adjacent marshes may result in undesirable impacts to the marshes and their biological resources. Benthic invertebrates, SAV, oysters, fish, and wetlands are critical parts of both estuarine and riverine food webs, providing habitat and forage for economically and ecologically important finfish and shellfish species, which are identified as an important indicator of potential effects, and are routinely monitored as part of environmental assessments. A range of species utilize wetlands as rearing habitats including seasonally flooded bottomland hardwood forests, estuarine environments and tidal marshes. Some examples of commercially or recreationally important fish species that rely on aquatic resources include: Atlantic Croaker, Southern Kingfish or Ground Mullet, Spot, and Hardhead Catfish. Many other fish species located in the Mobile estuary feed primarily on epifauna, crustaceans and mollusks, include crabs, crayfish, snails, clams, etc. Additionally, the Alabama Shad is a freshwater species that feeds almost exclusively on benthic invertebrates.

The ERDC-EL completed numerous aquatic resource assessments, including evaluations of potential impacts associated with navigation projects and alternatives analysis (Figure 1.1; Berkowitz et al. 2016). These studies were successfully executed through a combination of 1) direct measurements of aquatic resources and 2) modeling approaches. Mobile Bay contains a variety of natural resources. An interagency team identified the following resources for evaluation of potential project impacts: wetlands, submerged aquatic vegetation (SAVs), oysters, benthic invertebrates and fish (General Reevaluation Report meeting Mobile, AL 03/31/16). Due to the variety of aquatic resources being evaluated, specific examples of resource assessments are provided in the chapters below. The general approach for all aquatic resource assessments will include 1) assessment of existing resources and 2) analysis of potential impacts based upon water quality and sediment modeling outputs (Bunch, 2016).

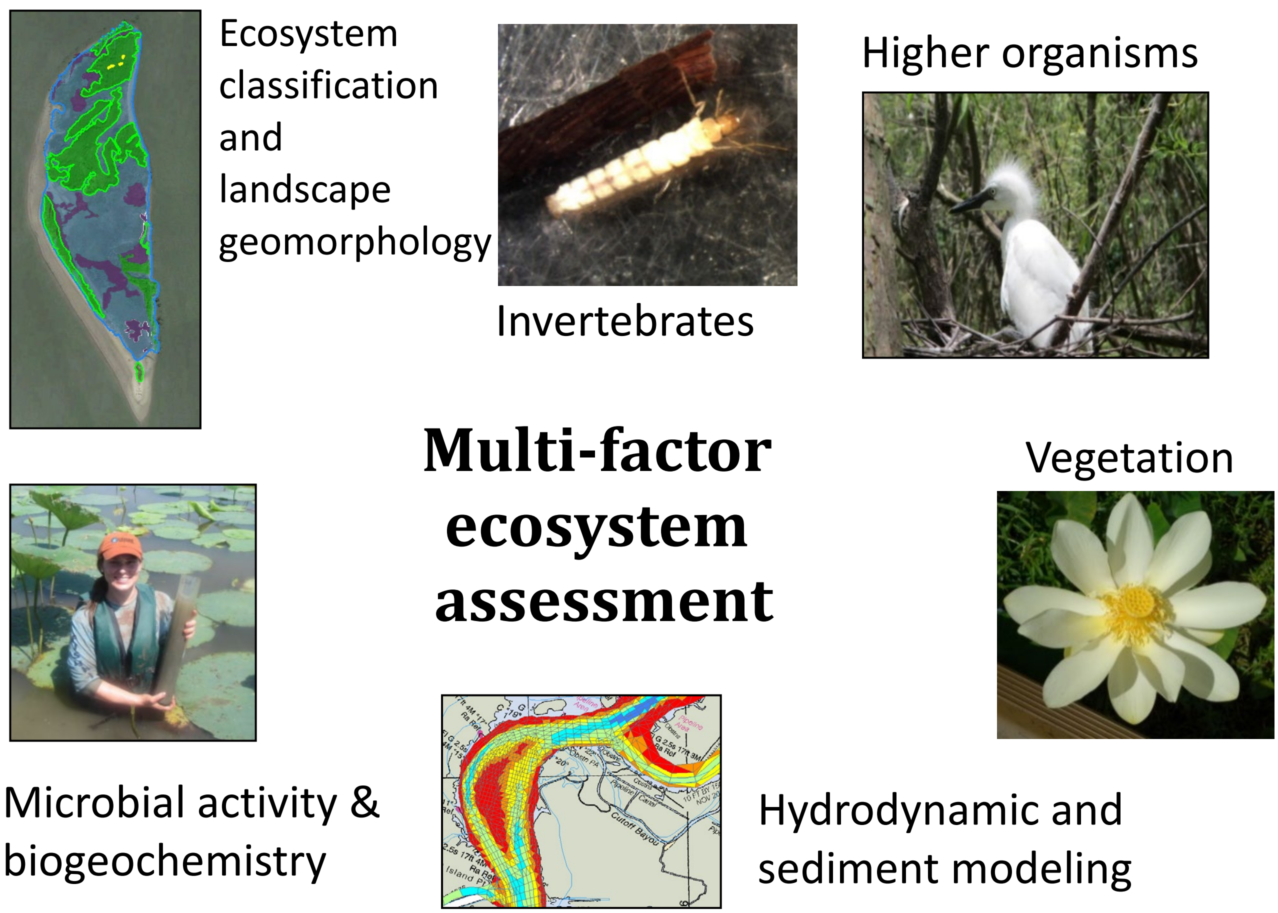


Figure 1.2. Conceptual model of the multi-factor assessment approach evaluating several trophic levels and aquatic resources (Berkowitz et al. 2016). The current assessment evaluates potential impacts to fish, invertebrates, wetlands, oysters, and submerged aquatic vegetation.

1.3 Approach

Mobile Bay contains a variety of natural resources. As a result, an interagency team identified the following resources for evaluation of potential project impacts: wetlands, submerged aquatic vegetation (SAVs), oysters, benthic invertebrates and fish (General Reevaluation Report meeting Mobile, AL 03/31/16). That group also highlighted salinity and water quality as the main parameters of concern and are the focus of the following report. Due to the variety of aquatic resources being evaluated, specific approaches for each resource assessment is provided in the chapters below. The general approach for all aquatic resource assessments will include 1) assessment of existing resources, 2) analysis of potential impacts based upon water quality modeling outputs (Bunch, 2016), and 3) evaluation of potential sea level rise implications. All hydrodynamic and water quality data was generated using a combination of approaches including the Geophysical Scale Multi-Block (GSMB) system, the Curvilinear Hydrodynamic in three-dimension Waterways Experiment Station (CH3D-WES) approach, and the CE-QUAL-ICM water quality component developed and maintained by the US Army Corps of Engineers Engineer Research and Development Center (Cerco and Cole 1995, others). Model outputs allowed for analysis of a variety of water quality parameters including salinity (Figure 1.2).

Detailed model parameterization and implementation information is provided in other documentation associated with the supplemental Environmental Impact Statement and is not reproduced herein. The model documentation includes a discussion of the selected model dataset, verification, validation and other factors. The models utilized in this report make the assumption that available data represents the conditions within Mobile Bay. In particular, the models apply water quality results based upon data from 2010, which was selected because it displayed a range of conditions including high and low water periods characteristic of the study area compared with long term averages (Bunch et al., 2018). The results presented may not reflect potential extreme flood and/or drought year water quality conditions. However, the selected approach captures the range of typical environmental conditions of the Mobile Bay and reflects the natural annual fluctuations in water quality parameters as the system responds to different levels of freshwater discharge. Additionally, the analysis of potential impacts to aquatic resources are dependent upon the accuracy of the water quality model and its projected changes in water quality. While the applied modeling approach proved adequate for evaluating short term effects of project construction and identifying significant shifts in environmental conditions, a second potential limitation of the approach results from difficulty of addressing very minor increases in salinity (e.g., <0.5 ppt) over decadal timescales.

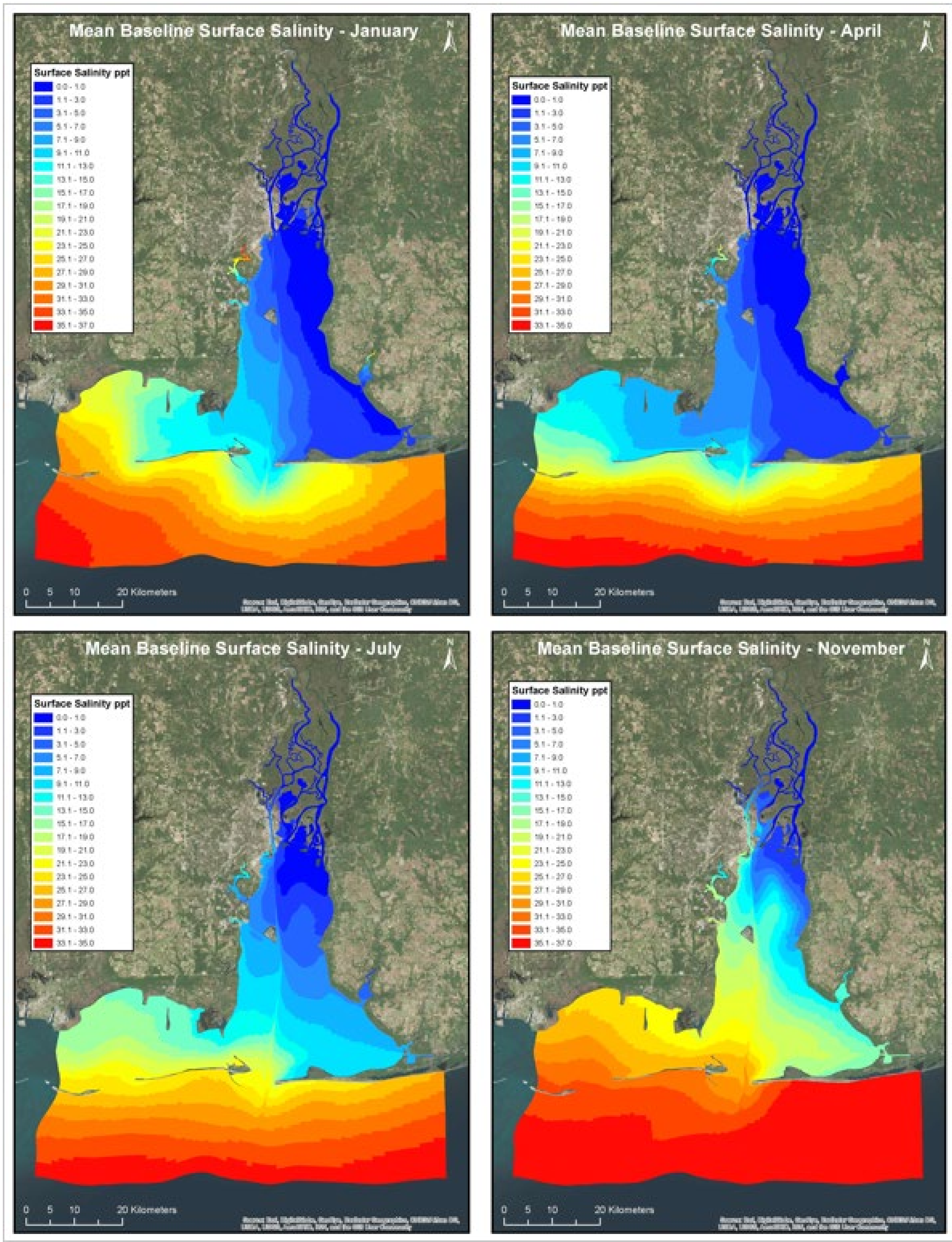


Figure 1.3. Example of surface water quality model outputs for the study area. Baseline (i.e., pre-project) salinity values are presented for winter, spring, summer, and fall (clockwise from top left).

Chapter 2: Benthic invertebrates

Summary

Potential impacts of the harbor deepening project on biological resources in Mobile Bay are a concern to natural resource managers because changes to saltwater – freshwater exchanges in the estuary could affect the distribution of biotic communities, including benthic macroinvertebrates and the fish that feed on them. In this chapter, benthic macroinvertebrates in Mobile Bay and upstream river habitat are examined. Results indicate that benthic macrofaunal assemblages transition from polychaete-rich assemblages in the estuary to being dominated by insects in freshwater habitat. In the fall, a gradual decline in salinity occurred as sampling occurred upstream in the Mobile River declining from 23 to 5 ppt. Benthic community composition remained consistent with estuarine assemblages within this zone, with a numerical dominance of capitellid, pilargiid, and spionid polychaetes. A sharp decline in salinity to freshwater conditions occurred near Bucks, Alabama, which corresponded to a significant change in the composition of benthic macroinvertebrates, i.e., polychaete abundances declined and insect (primarily, Ephemeridae and Chironomidae) abundances increased at this location and stations upstream.

Spring sampling occurred during a freshet, when low salinities were recorded throughout the study area. Benthic macroinvertebrates assemblages in the transitional and freshwater zones were similar to each other, with relatively high insect abundances, whereas estuarine assemblages had higher polychaete abundances. As with the fall sampling, biomass was dominated by bivalve molluscs, especially in the estuarine habitat.

Water quality modeling indicated that mean bottom salinity increases of approximately 1 ppt are expected following harbor deepening and maximum increases of approximately 3 ppt may occur. The greatest salinity increases are projected to occur within the transitional and estuarine zones where benthic macrofaunal assemblages are dominated by polychaete worms that are well adapted to experiencing salinity fluctuations that occur during tidal exchanges. The change to an insect dominated benthic community occurs where freshwater habitat is encountered, which

during fall sampling was well upstream from predicted project impacts. Impacts of harbor deepening on benthic macrofauna due to salinity intrusion are predicted to be negligible, with no effects on higher trophic levels, such as fish, because prey availability and distributions are unlikely to be affected.

2.1 Introduction

General context: The balance between freshwater inflow and saltwater tidal exchanges is an important driver establishing salinity-zone habitats in estuaries (Van Diggelen and Montagna 2016) and salinity strongly influences benthic macroinvertebrate distributions (Telesh and Khlebovich 2010). Changes to this freshwater/saltwater relationship are associated with wetland loss on the northern Gulf of Mexico via altered riverine input of freshwater and sediment (Day et al. 2000) and salt water intrusion via canal dredging (Turner 1997). Channel dredging can affect this relationship, for instance, saltwater intrusion increased in the Pearl River estuary (Yuan and Zhu 2015), Tampa Bay (Zhu et al. 2014), and Lake Pontchartrain (Junot et al. 1983) following dredging. Other factors affect habitat quality and the salinity balance within an estuary, including severe storms, sediment changes, and development; therefore, understanding the influence of a single factor, such as channel dredging, is difficult. Alterations to inputs of freshwater (e.g., droughts, floods, flood control levees) or saltwater (e.g., channel deepening), can affect biotic communities that are adapted to particular salinity zones by changing their taxonomic composition and distributions. Important estuarine biota includes benthic invertebrates, which are relatively stationary, living within bottom sediments. Their abundances and distributions, therefore, can serve as an indicator of environmental conditions in an area. It is expected that saltwater intrusion will facilitate landward migration of estuarine benthic macroinvertebrate assemblages (Little et al. 2017). For instance, upstream migrations of estuarine and marine benthic invertebrates occurred following a drought event that caused salt water incursion (Attrill and Power 2000). Salinity, however, is not the only factor affecting the distributions of benthic invertebrates. They also respond to sediment composition, competition, and predator-prey relationships (Little et al. 2017).

Problem statement: Because benthic invertebrates are important prey items for bottom feeding fishes and crustaceans, changes to invertebrate distributions and abundances could affect these higher trophic organisms. The widening and deepening of the Mobile Bay Federal Navigation Channel is an environmental concern because the possible influx of saltwater into upstream habitats may affect benthic invertebrates and their fish predators. Salinity in Mobile Bay is affected by river inflow, wind, and tides. Periodic breaches to barrier islands such as “Katrina Cut,” which was filled in 2010 (Park et al. 2014), also affect salinity patterns in the Bay. Commercially and recreationally important estuarine fish that feed on benthic invertebrates in these estuarine and freshwater habitats include Atlantic croaker, southern kingfish, spot, and hardhead catfish. The freshwater Alabama shad feeds almost exclusively on benthic invertebrates.

Model purpose: This chapter characterizes baseline benthic infaunal communities in estuarine, transitional, and freshwater habitats in the Mobile Bay watershed. Changes in benthic community composition among these habitat types are documented along the salinity gradient and are used to estimate how far upriver changes may occur following channel deepening.

Model summary: Empirical data were collected to document the distribution and abundance of benthic macroinvertebrates within the potential zone of influence of the harbor deepening project. Multivariate statistical techniques were used to determine the location(s) where the taxonomic composition of these benthic assemblages changed relative to bottom salinity concentrations. Water quality model results were assessed near benthic stations to determine whether projected salinity increases affected macroinvertebrate distributions.

2.2 Methods – Model Development Process

Study Site

Mobile Bay, Alabama is formed by the Fort Morgan Peninsula to the east and Dauphin Island, a barrier island on the west. Mobile Bay is 413 square miles (1,070 km2) in area. It is 31 miles (50 km) long with a maximum width of 24 miles (39 km). The deepest (75 feet, 23 m) areas of the Bay are located within the federal navigation channel, which serves Alabama’s only port for ocean-going vessels, but the average depth of the bay is around 10 feet (3 m). Throughout this shallow estuary, low wind speeds can contribute to stratification and the occurrence of hypoxic events (Turner et al. 1987). Water masses with low dissolved oxygen can be forced onshore, depositing moribund demersal fish and crustaceans in phenomena termed "jubilees" (May 1973). The Mobile Bay watershed is the sixth largest river basin in the United States and the fourth largest in terms of streamflow. It drains water from three-fourths of Alabama as well as portions of Georgia, Tennessee and Mississippi. The Mobile River and Tensaw River empty into the northern end of the Bay. Several smaller rivers: Dog River, Deer River, and Fowl River, on the western side of the Bay and the Fish River on the eastern side also empty into the Bay. River discharge is seasonal with high flows in the late winter and early spring and lowest flows in the summer. Estuarine habitat receives seawater during tidal exchanges, transitional zones have lower salinities and occur upstream in rivers and tributaries, and freshwater zones typically are upstream from the tidal reach of seawater.

Benthic macrofauna in Mobile Bay are dominated by polychaetes and macrofaunal abundances are relatively low in this area compared to other Gulf of Mexico estuaries (HX5, 2016). An examination of the Environmental Monitoring and Assessment Program (EMAP) benthic data set collected by the U.S. Environmental Protection Agency from (1991-1994) to assess the potential foraging value for Gulf sturgeon revealed the macrofaunal densities in Mobile Bay were greatest at water depths of 1.5 to 2.5m, with decreasing densities at greater depths.

Sampling protocol – Process followed

Environmental parameters

Samples were taken by ponar grab with a minimum penetration depth of 10 cm into bottom sediments. Visual observations were made of the degree of penetration of the ponar sampler, in particular as to whether the bucket was completely closed during retrieval and the estimated volume of sediment within the grab sampler. These estimates of the penetration of the ponar sampler were recorded on field data sheets. Of the total number of samples collected, 85% were completely full. These samples were typically comprised of soft muddy sediments. Approximately 9% of the total samples had a penetration depth suitable to between 75% to less than 100% of the volume of the ponar grab. Sediment composition of these samples were typically sandy mud. And 6% of the total samples were 50% to 75% full. Sediment composition of these samples were mostly sand combined with shell hash. Sediment samples were processed using a combination of wet sieving, flotation procedures and coulter counter techniques. Samples were soaked in a 20% sodium hexametaphophate solution to disaggregate the silt and clay fractions, and then agitated in a sonic bath for several minutes. Organic content was measured as weight loss upon ignition. Grain size data analysis was performed using Gradistat 8.0 (Blott and Pye, 2001), which calculates the percentage of sediments in individual grain size categories. Grain size parameters and descriptions were based on the methods of Folk and Ward (1957) and Folk (1968).

Benthic macroinvertebrates

Benthic macroinvertebrates were sampled in October 2016 and May 2017. A total of 240 benthic samples were collected, 120 samples in each season. Samples were collected at 30 stations within each zone (Freshwater, Brackish and Estuarine (upper bay) by ponar grab (Figures 1-4). Successful samples reached a minimum penetration depth of 10 cm into bottom sediments. Samples were sieved in the field using a 0.5 mm mesh to remove excess sediment, placed in individual fabric bags, and preserved in 10% buffered formalin. All samples were collected by ERDC personnel with the assistance of personnel from the USACE: Mobile District (boat and operator). Species were enumerated by LPIL (lowest practical identification level) taxa. Wet-weight biomass was determined after combining LPIL taxa into higher-order taxa (Annelids, Arthropoda, Mollusca, Echinodermata and Miscellaneous). Excess water was removed from the benthic invertebrate sample by placing the sample on a glass microfiber filter. The filter was placed on a manifold apparatus attached to a vacuum pump that removed excess water. Mollusk shells weights were included in the biomass measurements.

Direct measurements of biomass of small invertebrates include wet-weight and ash-free dry weights (AFDW), each method with its on advantages and disadvantages. For example, AFDW requires the invertebrate sample is destroyed in the procedure. When the researcher wishes to preserve the sample for future study or comparisons, wet-weight biomass is the technique of choice.

Statistical Approach – Process followed

Quantification: One-factor Analysis of Variance (ANOVA) tests were used to examine potential differences in water quality parameters among station types. Water quality data met the normality and homogeneity of variance assumptions of this parametric test. A one-factor ANOVA was used to test for habitat type differences in Annelid biomass for which the data also met test assumptions. Non-parametric Kruskal-Wallis tests were best suited to test for potential differences in Arthropod and Molluscan biomass.

Analysis of Similarity (ANOSIM) tests were used to examine potential differences in benthic macrofaunal assemblages among habitat types. ANOSIM results are distinguished on a scale of R = 0 (groups were indistinguishable) to R = 1 (no similarity among groups; Clarke et al., 2014; Clarke and Gorley, 2015). Nonmetric multidimensional scaling (nMDS) ordinations were plotted with each symbol representing a station coded by habitat type. In these plots, stations with similar assemblages are grouped close together and stations with dissimilar assemblage composition are farther apart. In cases where benthic macrafaunal assemblages differed between habitat types, Similarity Percentages (SIMPER) were conducted to identify the taxa contributing at least 5% to the dissimilarities among groups.

Application of Water Quality Modelling Results

Salinity

Although temperature, turbidity, and pH are all physical parameters that reflect and influence ecosystem health, salinity is more highly related to benthic community composition. Model results were used for the bottom three strata to characterize projected salinities following harbor deepening. Projected salinities for cells within a 100m of each benthic station were evaluated for the mean project salinity. To evaluate a worst case scenario, the maximum difference in salinity projected by the model under harbor deepening conditions also was considered for each month for cells within the aforementioned buffer.

Evaluation: Multivariate statistics were conducted using PRIMER 7 (Plymouth Routines In Multivariate Ecological Research), which is ideal for analyzing arrays of species-by-samples data for environmental assessments (Clarke et al. 2014). The non-parametric multivariate model makes few assumptions about the form of the data, using non-metric ordination and permutation tests that are robust and applicable to macroinvertebrate abundance data. PRIMER is a proven, effective statistical tool that has been used to identify macroinvertebrate assemblages associated with salinity zones related to management of freshwater inflows (Palmer et al. 2015).

2.3 Results - Application

Fall 2016

Fall 2016

Environmental Conditions

During the fall (October 2016), water quality parameters were recorded within expected ranges in each zone. Salinities differed significantly among habitat types ($F_{2,85} = 57.4$, $p < 0.001$), declining from averaging 18 ppt in the estuarine zone to 4 ppt in the freshwater zone (Figure 5), with several stations less than 1 ppt. Dissolved oxygen concentrations were above hypoxic concentrations, which are defined as DO concentrations below 2-3 mg/L (Dauer et al. 1992; Diaz and Rosenberg, 1995). Dissolved oxygen concentrations did not differ significantly among habitat types ($F_{2,85} = 1.4$, $p > 0.2$), with highest concentrations in the freshwater zone (Figure 5). Sampling depths were significantly greater in the freshwater habitat ($F_{2,85} = 5.9$, $p = 0.004$), averaging 3.7 m compared to 2.2 m in the transitional and estuarine zones. Bottom water temperatures averaged 25ºC in all locations. Sediments in estuarine habitat were comprised of more fine grain sizes, e.g., silts and clays, compared to the sandier transitional and freshwater habitats (Figure 6). Total organic content was significantly lower in the freshwater ($F_{2,85} = 5.75$, $p = 0.005$) than the estuarine and transitional habitats (Figure 6).

Benthic Macrofauna

A total of 1,789 individual benthic macrofauna from 54 taxa were collected during baseline (October 2016) sampling, with the highest number of taxa and individuals collected in freshwater habitat (Table 1). The distribution and abundance of many species changed along the salinity gradient sampled. For example, the dwarf surf clam Mulinia lateralis, amphipod Grandidierella bonnieroides, and polychaetes Glycinde solitaria, Laeonereis cuveri, and Paraprionospio pinnata, were abundant in the estuary, but not common in the transitional and freshwater zones. In contrast, seven insect taxa were collected in freshwater benthic habitat, one insect taxon in the transitional zone, and none within the estuary (Table 1). Likewise, tubificid oligochaetes were more abundant in the freshwater zone. Several polychaetes were more widely distributed, occurring in all habitat types throughout the study area, including, Mediomastus (LPIL), Parandalia americana, and Streblospio benedicti.

Fall benthic biomass was dominated by bivalve molluscs in the estuarine habitat (Kruskal-Wallis test statistic = 19.6, $p < 0.001$, df = 2; Figure 7). Bivalves were present in only four of 30 samples in the transitional zone, and were uncommon in the freshwater zone. Arthropod (insects) biomass was highest in the freshwater zone (Kruskal-Wallis test statistic = 26.6, $p < 0.001$; df = 2; Figure 7), whereas Annelid (primarily polychaetes) biomass was relatively even across the salinity zones ($F = 2.8$, $p > 0.05$; df = 2; Figure 7).

Benthic Assemblages

The taxonomic composition of benthic macroinvertebrate assemblages overlapped considerably between the estuarine and transitional zones, with more distinct assemblages in freshwater habitat (Figure 8). Within the freshwater zone, samples collected in the Mobile River were similar to estuarine and transitional assemblages and distinct from assemblages collected in the Tombigbee and Alabama Rivers. A diverse array of polychaetes was collected in the Mobile River (Table 2), which accounts for this location's similarity to estuarine and transitional assemblages. When comparing benthic assemblages between only the transitional and freshwater zones (Figure 9), it is more apparent that stations in the lower Mobile River (stations C1-C9) overlapped in composition with assemblages collected downstream in the transitional zone. Therefore, a distinct break in benthic communities is apparent between stations C9 and C10 (Figure 10) in the fall, which is an approximate 4 km stretch of river with several changes in sinuosity between the stations (Figure 3). Stations upstream C9 included tubificid oligochaetes and insects that were not collected downstream, whereas polychaetes were in higher abundances at stations C1-C9 and were uncommon at the upstream stations (Table 3).

Within the transitional zone, Tensaw River assemblages differed from all other locations because the benthic macrofauna were comprised entirely of nemerteans, tubificid oligochaetes, and polychaetes (Table 2). Benthic macrofauna in the Alabama River were the most diverse of any other location and included 14 taxa, with more bivalves and insects than collected in other locations.

Spring 2017

Environmental Conditions

During the spring (May 2017), sampling occurred during a period of high freshwater runoff (a freshet), therefore salinities were very low in all areas, averaging less than 4 ppt in the estuarine and less than 1 ppt in the transitional and freshwater zones (Figure 11). Salinities in the estuarine zone were significantly higher than all other zones ($F_{2,86} = 52.5$, $p < 0.001$). Dissolved oxygen concentrations were high, well above levels associated with hypoxic conditions (Figure 11). Similar to fall sampling, freshwater stations were significantly deeper ($F_{2,86} = 20.8$, $p < 0.001$) than those in the transitional and estuarine zones. Fine-grained sediments (silts and clay) were prevalent in the estuarine and transitional zones, with a greater composition of coarser grain sizes (sands and some gravel) in the freshwater environment (Figure 12). Total organic content was higher in the estuarine than the transitional and freshwater zones (Figure 12) although this difference is not significant ($F_{2,86} = 2.3$; $p > 0.1$). Spring temperatures averaged approximately 23°C at all stations.

Benthic Macrofauna –

A total of 2,165 individual benthic macrofauna from 44 taxa were collected during spring (May 2017) sampling, with the highest number of individuals collected in estuarine habitat (Table 4). A major difference between the fall and spring benthic assemblages is the presence of insects in the estuarine zone and much higher insect abundances in the transitional and freshwater zones. Taxa richness was relatively even among habitat types.

Spring benthic biomass was strongly dominated by molluscs in the estuary (Kruskal-Wallis test statistic = 39.6, $p < 0.001$; df = 2; Figure 13). Annelid biomass differed significantly among habitat types ($F_{2,87} = 4.1$, $p = 0.02$), with lowest biomass in freshwater habitat. Arthropod (primarily crustaceans) biomass was significantly higher in transitional habitat (Kruskal-Wallis test statistic = 12.9, $p = 0.002$; df = 2; Figure 13).

Benthic Assemblages –

In the spring, there was less overlap in the taxonomic composition of macrofaunal assemblages among the different habitat types. For instance each pairwise comparison (ANOSIM) between areas differed significantly (Figure 14). The biggest difference occurred between the estuarine and freshwater assemblages ($R = 0.72$, $p = 0.001$), with smaller differences between estuarine and transitional ($R = 0.28$, $p = 0.001$) and transitional and freshwater ($R = 0.30$, $p = 0.001$) zones. Locations of where benthic assemblages changed between the transitional and freshwater zones were less obvious than fall assemblages (Figure 15), with freshwater invertebrates occurring downriver in the transitional zone (Figure 10).

Application of Water Quality Modelling Results

In the fall, maximum projected differences in salinity ranged from 1.9 to 3.6 ppt and the greatest changes in salinity were projected for the estuarine habitat where benthic macrofauna are well adapted to salinity fluctuations of this magnitude. In the winter, maximum changes to salinity ranged from 2.5 to 3.2 ppt. In the spring, maximum salinity changes were projected to be 2.2 to 3.2 ppt, whereas summer maximum changes ranged from 1.6 to 2.9 ppt.

2.4 Discussion

Potential impacts of the harbor deepening project on biological resources in Mobile Bay are a concern to natural resource managers because the navigation channel has big influence on water circulation, estuarine mixing, and sedimentation patterns in the Bay (Osterman and Smith 2012). The completion of the navigation channel in the 1950s restricted tidal flushing and increased the input of terrestrial organic matter (Osterman and Smith 2012). In addition, hypoxic events are associated with low flow conditions, rather than nutrient loading (Cowan et al. 1995; Park et al. 2007), therefore if channel deepening alters flow conditions, biota in the estuary and watershed could be affected. This examination of benthic macroinvertebrates has established how benthic communities transition from estuarine to freshwater habitat, which largely reflected a change from relatively high abundances of polychaetes to insects, respectively. A similar transition in benthic community composition was reported for Lavaca Bay and Matagorda Bay, Texas, in which polychaetes and crustaceans were indicator taxa for brackish and marine habitats and insect larvae occurred in freshwater areas (Pollack et al. 2009). Likewise, in the fall, when salinities were relatively high, the extent of influence of salt water on benthic macroinvertebrates was evident as far upstream as station C9, which is located south of Bucks, Alabama. At this location, immediately upstream from C9, the Mobile River takes two sharp 90 degree bends, first east, then north, which may contribute to the abrupt salinity decline between stations C9 (5 ppt) and C10 (<1 ppt) if tidal forces were weaker than the opposing conditions created by flow and river sinuosity. These results indicate that under the environmental conditions present in the fall of 2016, a clear break in the upstream influence of estuarine waters occurred near Bucks, Alabama. Downstream from this location, fall benthic macroinvertebrate assemblages were similar through the transitional habitat and into the estuary.

In the spring, salinities were less than one ppt throughout all transitional and freshwater stations, therefore, a clear break in benthic macroinvertebrate composition related to salinity change was not evident.

Application of Water Quality Modelling Results

Salinity

These most extreme projected changes in salinity occurred within the transitional and estuarine zones where benthic macrofaunal assemblages are dominated by polychaete worms that experience greater salinity fluctuations during tidal exchanges. Differences in benthic macrofaunal assemblages occur where freshwater habitat begins, which in the fall, was further upstream than the water quality grid extended. There is no indication that the location of the freshwater transition point will be affected by the harbor deepening project. Impacts to higher trophic levels, such as fish, will be negligible because prey availability and distributions are unlikely to be affected.

Sea Level Rise

Maximum potential salinity changes projected by the water quality model under a scenario of sea level rise did not predict conditions that were more extreme than previously reported. For instance, fall maximum salinity changes could be as small as 1.2 ppt instead of 1.9 ppt, whereas spring maximum salinity predictions were as low as 0 ppt. Based on these model predictions, there is no indication that sea level rise will substantially affect benthic macrofaunal assemblage distributions.

Dissolved Oxygen

Estuarine organisms respond to decreasing dissolved oxygen in variable ways depending on their life stage and mobility. In general, however, a consistent pattern of response occurs at very low dissolved oxygen concentrations, i.e., below 2 mg/L. Mobile fish and crustaceans avoid benthic a habitat with oxygen concentrations below 2 mg/L. Less mobile benthic invertebrates, such as burrowing species, exhibit stress behaviors (e.g., emerging from sediments) at oxygen concentrations from 1.5-1 mg/L, with mortality occurring if durations of low dissolved oxygen concentrations are extensive (Rabalais et al., 2001).

A worst case scenario of harbor deepening project impacts on dissolved oxygen concentrations was evaluated by determining the minimum concentrations predicted under project conditions in the summer. High temperatures combine with low dissolved oxygen concentrations to create the most deleterious biological impacts. Minimum summer (June – September) dissolved oxygen concentrations ranged from 6.7 -7.1 mg/L, which is a concentration well above hypoxic levels that would induce stress responses or mortality in benthic macroinvertebrates.

Model limitations:

Predictions of potential impacts to benthic macroinvertebrates are dependent upon the accuracy of the water quality model and its projected changes to salinity.

Benthic macroinvertebrates were sampled only during two seasons (fall and spring), therefore summer distributions and abundances are inferred, but not documented.

Spring macroinvertebrate sampling occurred during a period of extremely high freshwater inflows, therefore spring invertebrate distributions during less extreme environmental conditions were not documented.

2.5 References

Attrill, M.J. and M. Power. 2000. Effects on invertebrate populations of drought induced changes in estuarine water quality. Marine Ecology Progress Series 203: 133-143.

Barry Bunch, Earl Hayter, Sung-Chan Kim, Elizabeth Godsey and Ray Chapman. 2018. Three Dimensional Hydrodynamic, Water Quality, and Sediment Transport Modeling of Mobile Bay. ERDC-LR-X.

Blott, S. J. and K. Pye. 2001. GRADISTAT: a grain size distribution and statistics package for the analysis of unconsolidated sediments. Earth Surface Processes and landforms 26, 1237-1248.

Clarke, K. R. and R. N. Gorley. 2015. PRIMER v7: user manual/tutorial. PRIMER-E Ltd., Plymouth, United Kingdom.

Clarke, K. R., Gorley, R. N., Somerfield, P. J., and R. M. Warwick. 2014. Change in marine communities: an approach to statistical analysis and interpretation. 2nd edition. PRIMER-E, Plymouth, United Kingdom.

Cowan, J.L.W., J.R. Pennock, and W.R. Boynton. 1996. Seasonal and interannual patterns of sediment-water nutrient and oxygen fluxes in Mobile Bay, Alabama (USA): regulating factors and ecological significance. Marine Ecology Progress Series 141: 229-245.

Dauer, D. M., A. J. Rodi Jr., and J. A. Ranasinghe. 1992. Effects of low dissolved oxygen events on the macrobenthos of the lower Chesapeake Bay. Estuaries 15: 384-391.

Day, J.W. Jr., G.P. Shaffer, L.D. Britsch, D.J. Reed, S.R. Hawes, and D. Cahoon. 2000. Pattern and process of land loss in the Mississippi Delta: A spatial and temporal analysis of wetland habitat change. Estuaries 23: 425-438.

Diaz, R. J. and R. Rosenberg. 1995. Marine benthic hypoxia: a review of its ecological effects and the behavioural responses of benthic macrofauna. Oceanography and Marine Biology Annual Review 33: 245-303.

Folk, R. L. 1968. Petrology of Sedimentary Rocks. Hemphills, University of Texas., Austin, TX 170 p.

Folk, R. L. and W. C. Ward. 1957. Brazos River bar: a study in the significance of grain size parameters. Journal of Sedimentary Petrology 27: 3-26.

HX5. 2016. Evaluating Gulf sturgeon foraging habitat value: A meta-analysis approach. Prepared for U.S. Army Corps of Engineers, Mobile District, 109 St. Joseph St., Mobile, AL 36602. 58 pp.

Junot, J. A., M. A. Poirrier and T. M. Soniat. 1983. Effects of saltwater intrusion from the Inner Harbor Navigation Canal on the benthos of Lake Pontchartrain, Louisiana. Gulf Research Reports 7: 247-254.

Little, S., P.J. Wood, and M. Eliott. 2017. Quantifying salinity-induced changes on estuarine benthic fauna: The potential implications of climate change. Estuarine, Coastal and Shelf Science 198: 610-625.

May, E.B. 1973. Extensive oxygen depletion in Mobile Bay, Alabama. Limnology and Oceanography 18: 353-366.

Osterman, L.E. and C.G. Smith. 2012. Over 100 years of environmental change recorded by foraminifers and sediments in Mobile Bay, Alabama, Gulf of Mexico, USA. Esturarine, Coastal and Shelf Science. 115: 345-358.

Palmer, T.A., P.A. Montagna, R.H. Chamberlain, P.H. Doering, Y. Wan, K.M. Haunert and D.J. Crean. 2015. Determining the effects of freshwater inflow on benthic macrofauna in the Caloosahatchee Estuary, Florida. Integrated Environmental Assessment and Management 12:529-539.

Park, K., Kim, C., and W.W. Schroeder. 2007. Temporal variability in summertime bottom hypoxia in shallow areas of Mobile Bay, Alabama. Estuaries and Coasts 30: 54-65.

Park, K., S.P. Powers, G.S. Borsarge, and H. Jung. 2014. Plugging the leak: Barrier island restoration following Hurricane Katrina enhances larval retention and improves salinity regime for oysters in Mobile Bay, Alabama. Marine Environmental Research 94: 48-55.

Pollack, J. B., J. W. Kinsey, and P. A. Montagna. 2009. Freshwater Inflow Biotic Index (FIBI) for the Lavaca-Colorado Estuary, Texas. Environmental Bioindicators 4:2 153-169.

Rabalais N. N. D. E. Harper Jr, and R. E. Turner. 2001. Responses of nekton and demersal and benthic fauna to decreasing oxygen concentrations. In: Rabalais NN, Turner RE (eds) Coastal hypoxia: consequences for living resources. Coastal and Estuarine Studies 58. American Geophysical Union, Washington, DC, p 115–128.

Telesh, I.V. and V.V. Khlebovich. 2010. Principal processes within the estuarine salinity gradient: a review. Marine Pollution Bulletin 61:149-155.

Turner, R.E., W.W. Schroeder, and W.J. Wiseman. 1987. The role of stratification in the deoxygenation of Mobile Bay and adjacent shelf bottom waters. Estuaries 10: 13-19.

Turner, R.E. 1997. Wetland loss in the northern Gulf of Mexico: Multiple working hypothesis. Estuaries 20:1-13.

Van Diggelen, A. D. and P. A. Montagna. 2016. Is salinity variability a benthic disturbance in estuaries? Estuaries and Coasts 39: 967-980.

Yuan, R. and J. Zhu. 2015. The effects of dredging on tidal range and saltwater intrusion in the Pearl River estuary. Journal of Coastal Research 31: 1357-1362.

Zhu, J., R.H. Weisberg, and L.Y. Zheng. 2014. Influences of channel deepening and widening on the tidal and nontidal circulations of Tampa Bay. Estuaries and Coasts 38: 132-150.

Table 2.1. Total abundances of benthic macroinvertebrates collected in each area during Fall (October 2016) sampling (30 stations per area).						
Class	Family	LPIL	Estuarine	Transitional	Freshwater	Total
Arachnida	Araneae	Arachnida (LPIL)	0	0	3	3
		Hydracarina (LPIL)	0	0	1	1
Bivalvia	Bivalvia	Bivalve (LPIL)	0	0	2	2
	Mactridae	*Mulinia lateralis*	71	2	1	74
		Rangia cuneata	0	1	0	1
	Mytilidae	*Ischadium recurvum*	0	0	2	2
	Sphaeriidae	Sphaeriidae (LPIL)	0	0	4	4
	Tellinidae	*Macoma mitchelli*	0	1	0	1
	Unionidae	Unionidae (LPIL)	0	0	1	1
Crustacea	Ampeliscidae	Ampelisca (LPIL)	0	1	0	1
	Aoridae	*Grandidierella bonnieroides*	10	2	1	13
	Corophiidae	Corophiidae (LPIL)	0	0	2	2
		Monocorophium insidiosum	0	0	2	2
	Decapoda	Crab Megalops (LPILL)	0	0	1	1
	Harpacticoida	Harpacticoida (LPIL)	0	0	2	2
	Idoteidae	*Edotia triloba*	4	6	2	12
	Mysidacea	Mysidacea (LPIL)	0	2	0	2
	Mysidae	*Americamysis bahia*	0	9	0	9
		Bowmaniella (LPIL)	1	0	0	1
	Oedicerotidae	Ameroculodes (LPIL)	0	1	0	1
	Ogyrididiae	*Ogyrides alphaerostris*	2	2	0	4
	Palaemonidae	*Palaemon pugio*	0	0	1	1

	Portunidae	*Callinectes sapidus*	0	0	1	1
	Ceratopoginidae	Ceratopoginidae (LPIL)	0	0	7	7
	Chaoberidae	Chaoborus (LPIL)	0	0	2	2
		Chironomidae Pupa (LPIL)	0	0	5	5
		Chironomini (LPIL)	0	0	42	42
	Chironomidae	Tanypodinae (LPIL)	0	44	47	91
	Ephemeridae	Hexagenia (LPIL)	0	0	86	86
Insecta	Trichoptera	Trichoptera (LPIL)	0	0	1	1
Nematoda	Nematoda	Nematoda (LPIL)	1	1	13	15
		Nemertea 1 (LPIL)	62	18	5	85
Nemertea	Nemertea	Nemertea 2 (LPIL)	5	0	0	5
Oligochaeta	Tubificidae	Tubificidae (LPIL)	6	3	194	203
	Ampharetidae	*Hobsonia florida*	6	3	4	13
	Archiannelida	Archiannelida (LPIL)	0	0	1	1
		Capitella (LPIL)	40	27	0	67
	Capitellidae	Mediomastus (LPIL)	106	125	54	285
Polychaeta	Chaetopteridae	*Spiochaetopterus oculatus*	1	0	0	1
	Gonianidae	*Glycinde solitaria*	48	3	0	51
		Alitta succinea	2	3	6	11
	Nereidae	*Laeonereis cuveri*	16	0	0	16
	Nereididae	Nereidae (LPIL)	11	5	0	16
	Onuphidae	*Diopatra cuprea*	1	0	0	1
Polychaeta	Pectinariidae	*Pectinaria gouldii*	1	0	0	1

		Parandalia americana	125	72	79	276
		Sigambra (LPIL)	0	1	0	1
	Pilargiidae	*Sigambra tentaculata*	4	0	0	4
	Sabellidae	Sabellidae (LPIL)	0	0	1	1
		Marenzellaria viridis	0	0	6	6
		Paraprionospio pinnata	34	1	7	42
		Polydora (LPIL)	0	1	0	1
	Spionidae	*Streblospio benedicti*	31	211	70	312
Total Taxa Richness			23	25	35	54
Total Abundance			588	545	656	1,789

Table 2.2. Average abundances of benthic macroinvertebrates in each location within the Estuarine, Transitional, and Freshwater zones in October 2016.

		Estuarine	Transitional					Freshwater		
Class	Family	Estuarine	Raft River	Tensaw River	Chac. Bay	Apalachee River	Grand Bay	Mobile River	Tom. River	Alabama River
Arachnida	Araneae	0	0	0	0	0	0	0	0	0.33
Bivalvia	Mactridae	2.45	0	0	0	1	0	0	0	0
	Mysidae	0	0.21	0	0	0	0	0	0	0
	Sphaeriidae	0	0	0	0	0	0	0	0	0.44
	Unionidae	0	0	0	0	0	0	0	0	0.11
Crustacea	Corophiidae	0	0	0	0	0	0	0	0	0.22
	Harpacticoida	0	0	0	0	0	0	0	0.25	0
	Idoteidae	0.14	0.29	0	0	1	0	0.15	0	0
	Ogyridiae	0.07	0	0	0	0	0	0	0	0
Insecta	Ceratopoginidae	0	0	0	0	0	0	0	0.63	0.22
	Chaoberidae	0	0	0	0	0	0	0	0	0.22
	Chironomidae	0	0.29	0	4.67	0	6.5	0.38	4.25	5
	Ephemeridae	0	0	0	0	0	0	1.69	5	2.7
	Trichoptera	0	0	0	0	0	0	0	0	0.11
Nematoda	Nematoda	0	0	0	0	0	0	0	0	1.22
Nemertea	Nemertea	2.31	0.64	0.29	0.67	1	0	0.38	0	0
Oligochaeta	Tubificidae	0.21	0.21	0	0	0	0	1.23	6.63	1F
Polychaeta	Ampharetidae	0.21	0	0.29	0	0	0	0.31	0	0
	Archiannelida	0	0	0	0	0	0	0	0	0.11
	Capitellidae	5.03	3.86	10.14	1.33	3	4.25	3.92	0	0.22
	Gonianidae	1.66	0.21	0	0	0	0	0	0	0
	Nereidae	0.62	0.14	0	0	0	0	0.46	0	0
	Nereididae	0.38	0	0.29	0	0	0	0	0	0
	Pilargiidae	4.45	4.07	2	0	0	0	6.08	0	0
	Spionidae	2.24	3.29	22.71	0	0	1.25	6.08	0	0

Table 2.3. Benthic macroinvertebrate mean abundances of taxa that contributed at least 5% to dissimilarities between freshwater stations downstream from C10 and upstream from C9 (SIMPER).			
	Taxa	Downstream	Upstream
Oligochaeta	Tubificidae	0	9.2
Polychaeta	Pilargiidae	8.8	0
	Spionidae	8.6	0.3
	Capitellidae	5.6	0.2
Insecta	Ephemeridae	0	4.1
	Tanypodinae	0	2.2
	Chironomidae	0.1	2.2

Table 2.4. Total abundances of benthic macroinvertebrates collected in each area during (May 2017) sampling (30 stations per area).						
Class	Family	LPIL	Estuarine	Transitional	Freshwater	Total
Arachnida	Araneae	Hydracarina (LPIL)	0	0	2	2
Bivalvia	Mactridae	*Mulinia lateralis*	114	11	13	138
	Mytilidae	*Ischadium recurvum*	0	0	1	1
	Sphaeriidae	Pisidium (LPIL)	0	0	2	2
		Sphaeriidae (LPIL)	0	0	2	2
	Tellinidae	*Macoma mitchelli*	45	10	0	55
Crustacea	Alpheidae	Alpheidae (LPIL)	1	0	0	1
	Aoridae	*Grandidierella bonnieroides*	0	4	5	9
	Corophiidae	Corophiidae (LPIL)	0	0	1	1
		Monocorophium insidiosum	0	12	91	103
	Cumacea	Cumacea (LPIL)	3	0	0	3
	Gammaridae	*Gammarus mucronatus*	1	2	3	6
	Harpacticoida	Harpacticoida (LPIL)	0	3	0	3
	Haustoriidae	Lepidactylus (LPIL)	0	4	0	4
	Idoteidae	*Edotia triloba*	7	1	0	8
	Melitidae	*Melita nitida*	0	1	0	1
	Mysidacea	Mysidacea (LPIL)	0	5	0	5
	Oedicerotidae	Ameroculodes (LPIL)	12	0	0	12
	Xanthidae	Xanthidae (LPIL)	1	0	0	1
Gastropoda	Cyclichnidae	*Acetocina canaliculata*	1	0	0	1
	Gastropoda	Gastropoda (LPIL)	0	0	1	1
Insecta	Chaoberidae	Chaoborus (LPIL)	0	5	1	6

		Chironomidae Pupa (LPIL)	0	0	10	10
		Chironomini (LPIL)	13	116	192	321
	Chironomidae	Tanypodinae (LPIL)	6	70	16	92
	Coleoptera	Coleoptera larva	0	0	17	17
	Ephemeridae	Hexagenia (LPIL)	0	24	44	68
Nematoda	Nematoda	Nematoda (LPIL)	1	2	54	57
		Nemertea 1 (LPIL)	9	22	35	66
Nemertea	Nemertea	Nemertea 2 (LPIL)	18	0	0	18
		Tubificidae (LPIL)	7	5	109	121
Oligochaeta	Tubificidae	Tubificoides (LPIL)	19	39	0	58
	Ampharetidae	*Hobsonia florida*	77	18	10	105
		Capitella (LPIL)	39	1	1	41
		Heteromastus filiformis	2	0	0	2
Polychaeta	Capitellidae	Mediomastus (LPIL)	341	155	3	499
	Gonianidae	*Glycinde solitaria*	1	0	0	1
	Nereidae	Nereidae (LPIL)	4	3	0	7
	Orbiniidae	Leitoscoloplos (LPIL)	1	0	0	1
		Parandalia americana	88	113	17	218
	Pilargiidae	*Sigambra tentaculata*	5	0	0	5
		Marenzellaria viridis	0	3	43	46
		Polydora (LPIL)	0	0	2	2
Polychaeta	Spionidae	*Streblospio benedicti*	35	10	0	46
Total Taxa Richness			26	25	25	44

Total Abundance	851	639	675	2,165

Table 2.5. Average abundances of benthic macroinvertebrates in each location within the Estuarine, Transitional, and Freshwater zones in May 2017.										
		Estuarine	Transitional					Freshwater		
Class	Family	Estuarine	Raft River	Tensaw River	Chac. Bay	Apalachee River	Grand Bay	Mobile River	Tom. River	Alabama River
Arachnida	Araneae	0	0	0	0	0	0	0	0.17	0.13
Bivalvia	Mactridae	3.80	0.57	0.29	0	0.50	0	0.92	0.17	0
	Mytilidae	0	0	0	0	0	0	0.08	0	0
	Sphaeriidae	0	0	0	0	0	0	0.23	0	0.13
	Tellinidae	1.5	0.29	0	0.33	0.50	1.00	0	0	0
Crustacea	Alpheidae	0.03	0	0	0	0	0	0	0	0
	Aoridae	0	0.14	0	0	1.00	0	0.38	0	0
	Corophiidae	0	0	0.29	0	5.00	0	6.92	0.17	0.13
	Cumacea	0.1	0	0	0	0	0	0	0	0
	Gammaridae	0.03	0.07	0	0	0.50	0	0.23	0	0
	Harpacticoida	0	0.14	0.14	0	0	0	0	0	0
	Haustoriidae	0	0.07	0.43	0	0	0	0	0	0
	Idoteidae	0.23	0	0	0	0.50	0	0	0	0
	Melitidae	0	0.07	0	0	0	0	0	0	0
	Mysidacea	0	0	0	1.00	0.50	0.25	0	0	0
	Oedicerotidae	0.4	0	0	0	0	0	0	0	0
	Xanthidae	0.03	0	0	0	0	0	0	0	0
Gastropoda	Cyclichnidae	0.03	0	0	0	0	0	0	0	0
	Gastropoda	0	0	0	0	0	0	0.08	0	0
Insecta	Chaoberidae	0	0.36	0	0	0	0	0	0	0.13
	Chironomidae	0.63	5.29	1.71	21.33	4.50	6.75	4.00	4.00	17.75
	Coleoptera	0	0	0	0	0	0	0.08	0	2.00
	Ephemeridae	0	0	0.14	0	11.50	0	0.62	2.67	2.50
Nematoda	Nematoda	0.03	0.07	0	0	0	0.25	0.38	0.17	6.00
Nemertea	Nemertea	0.9	0.57	0.86	1.67	0	0.75	0.23	0	4.00

Oligochaeta	Tubificidae	0.87	2.14	1.29	0.67	0.50	0.50	2.00	9.00	3.63
Polychaeta	Ampharetidae	2.57	0.64	0	0.33	1.50	1.25	0.77	0	0
	Capitellidae	12.73	5.29	1.14	4.00	3.50	13.75	0.23	0.17	0
	Gonianidae	0.03	0	0	0	0	0	0	0	0
	Nereidae	0.13	0	0	0.67	0.50	0	0	0	0
	Orbiniidae	0.03	0	0	0	0	0	0	0	0
	Pilargiidae	3.10	6.36	1.00	0	3.00	2.75	1.31	0	0
	Spionidae	1.17	0.29	0.29	0.33	0	1.50	1.23	4.83	0

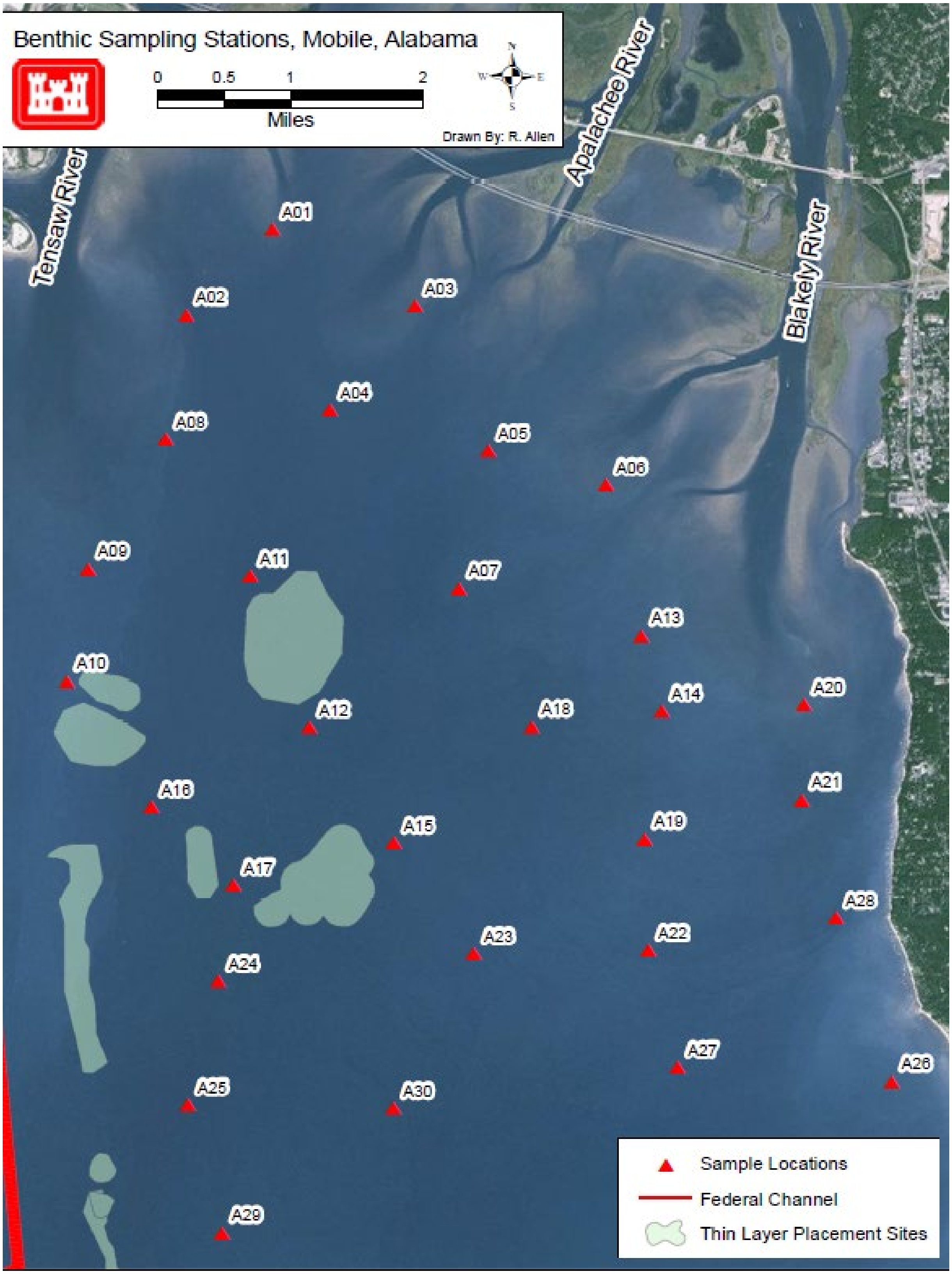


Figure 2.1. Benthic station locations for estuarine habitat in upper Mobile Bay.

Figure 2.2. Benthic stations locations in the transition zone.

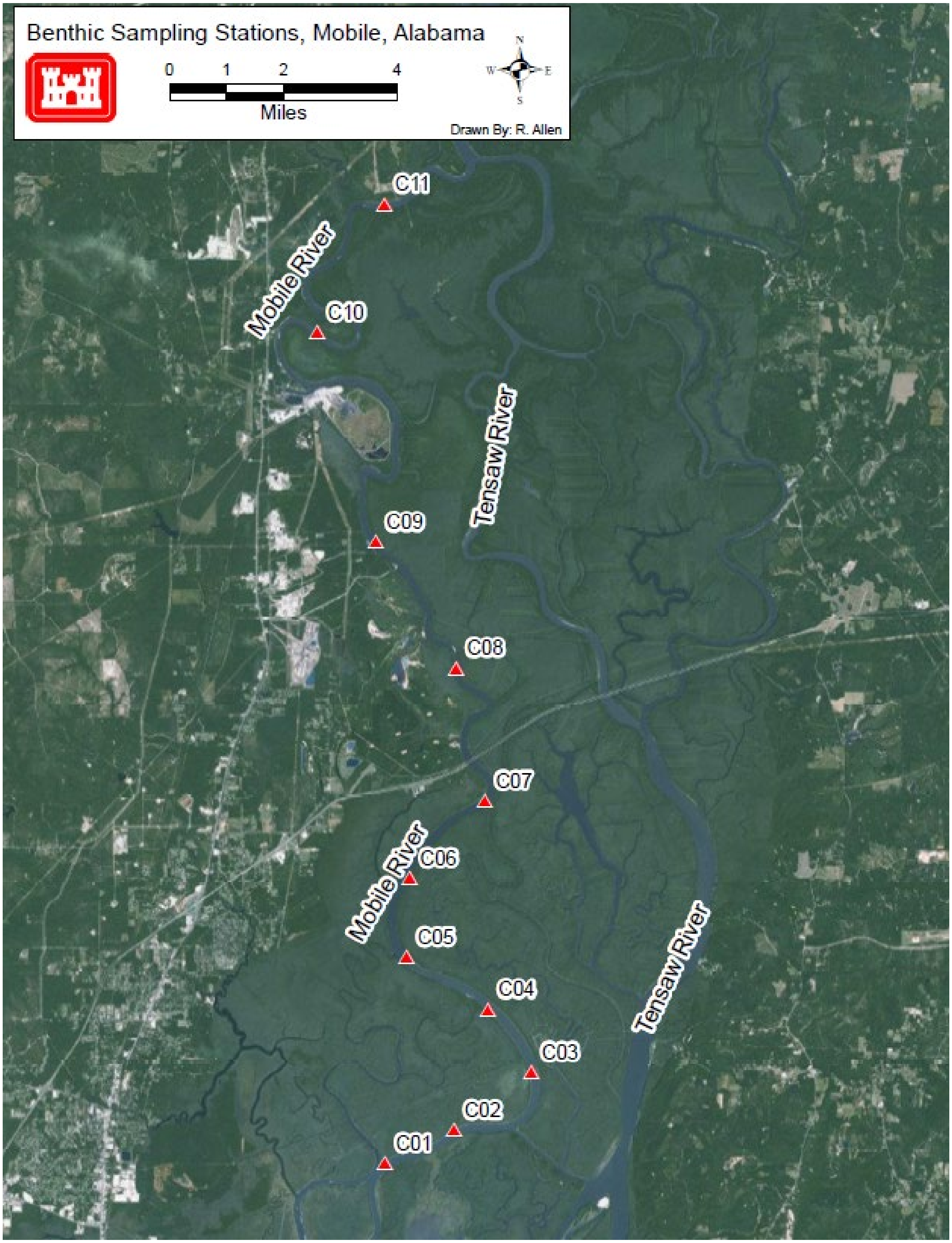


Figure 2.3. Benthic stations locations in the western portion of freshwater zone.

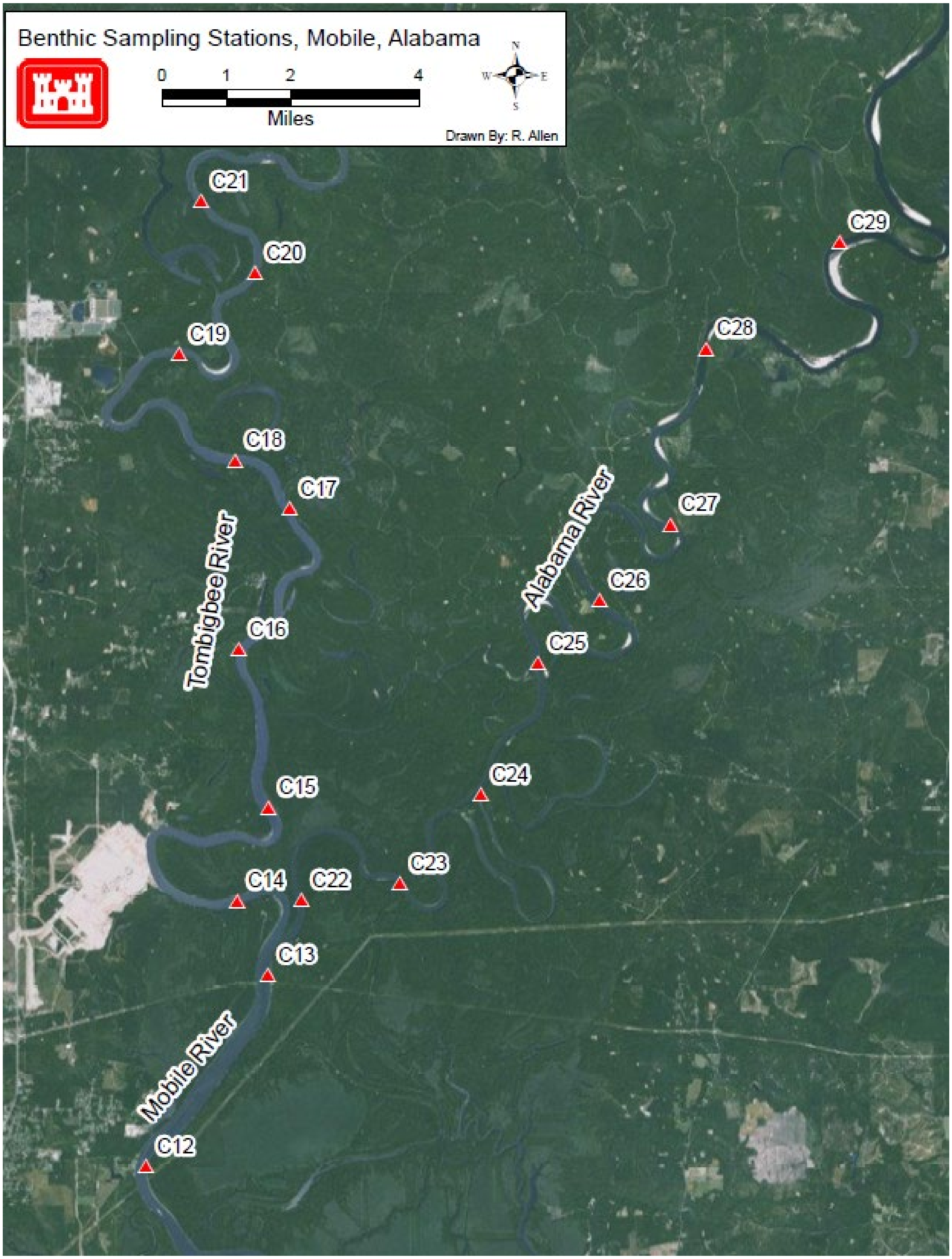


Figure 2.4. Benthic stations locations in the eastern portion of freshwater zone.

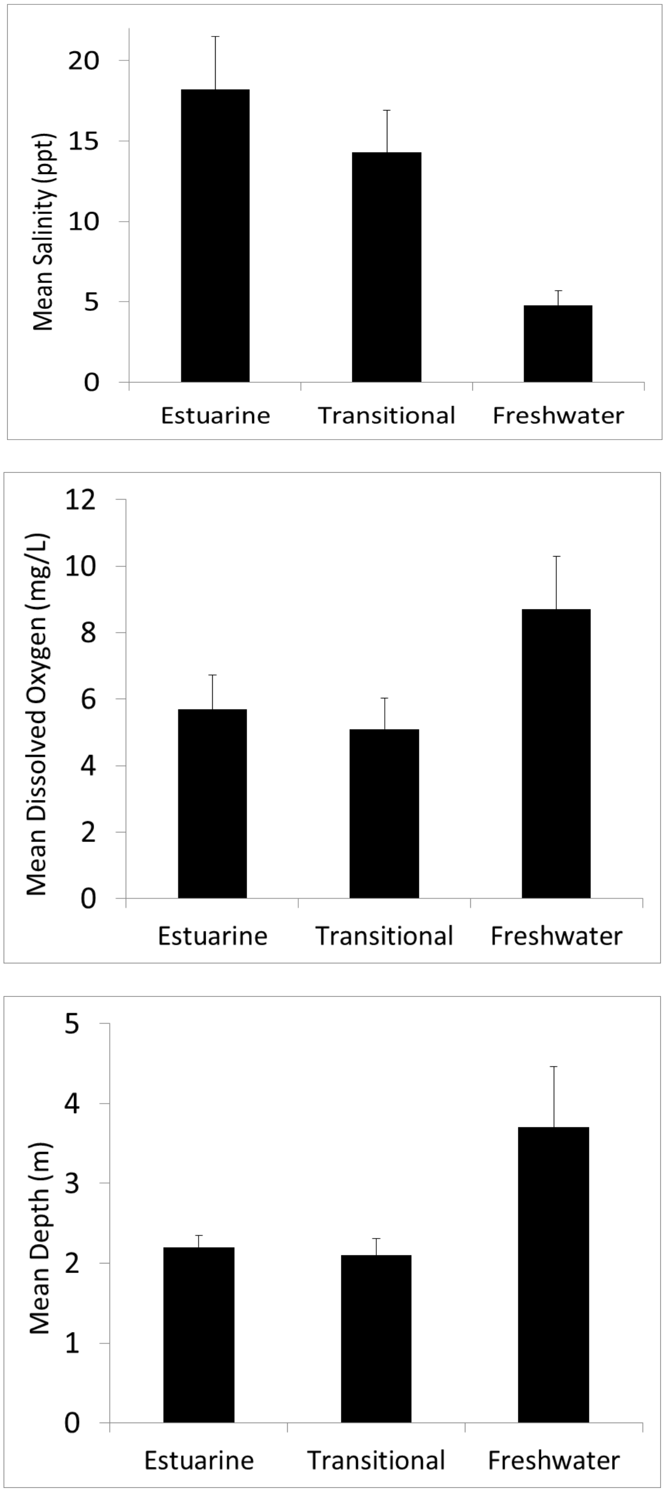
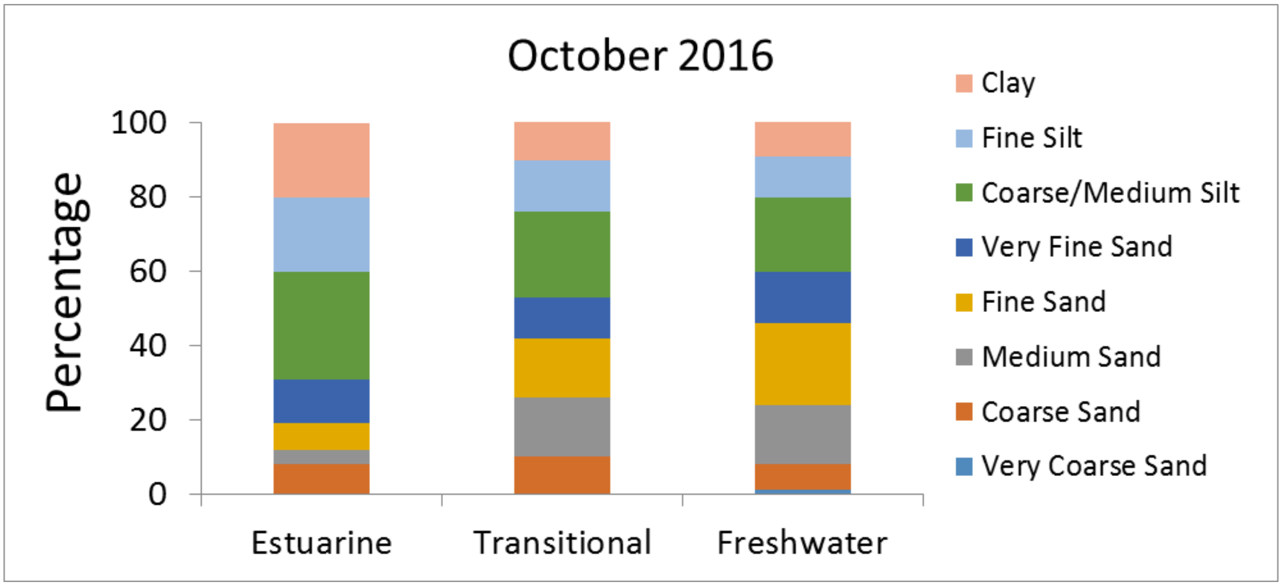


Figure 2.5. Mean (+ standard error) salinity, dissolved oxygen, and depth at stations in the estuarine, transitional, and freshwater zones during fall (October 2016) sampling.



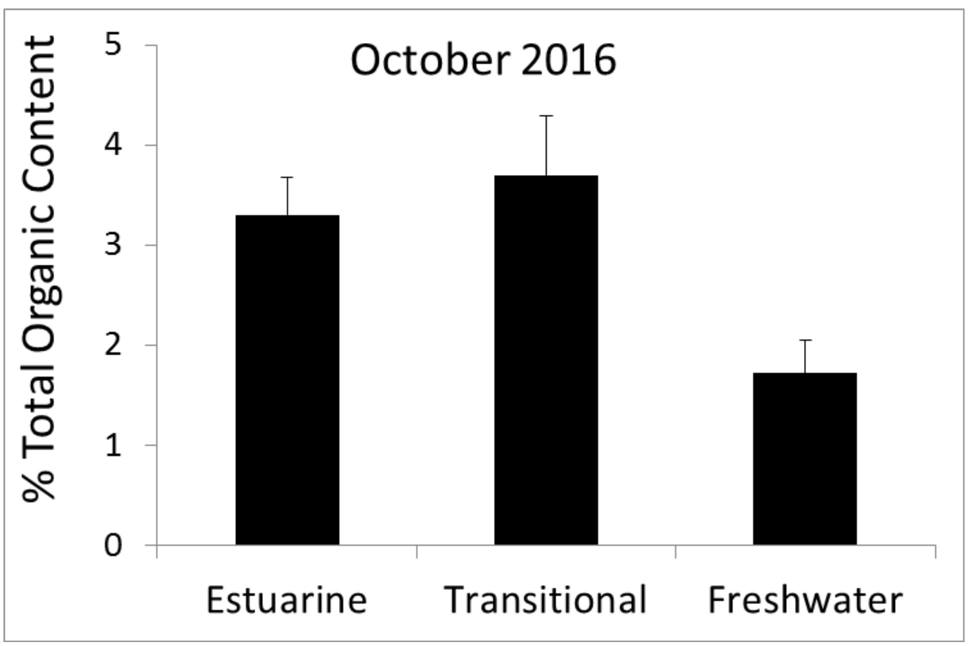


Figure 2.6. Sediment grain size distributions and % TOC in the estuarine, transitional, and freshwater zones during the fall 2016 sampling period.

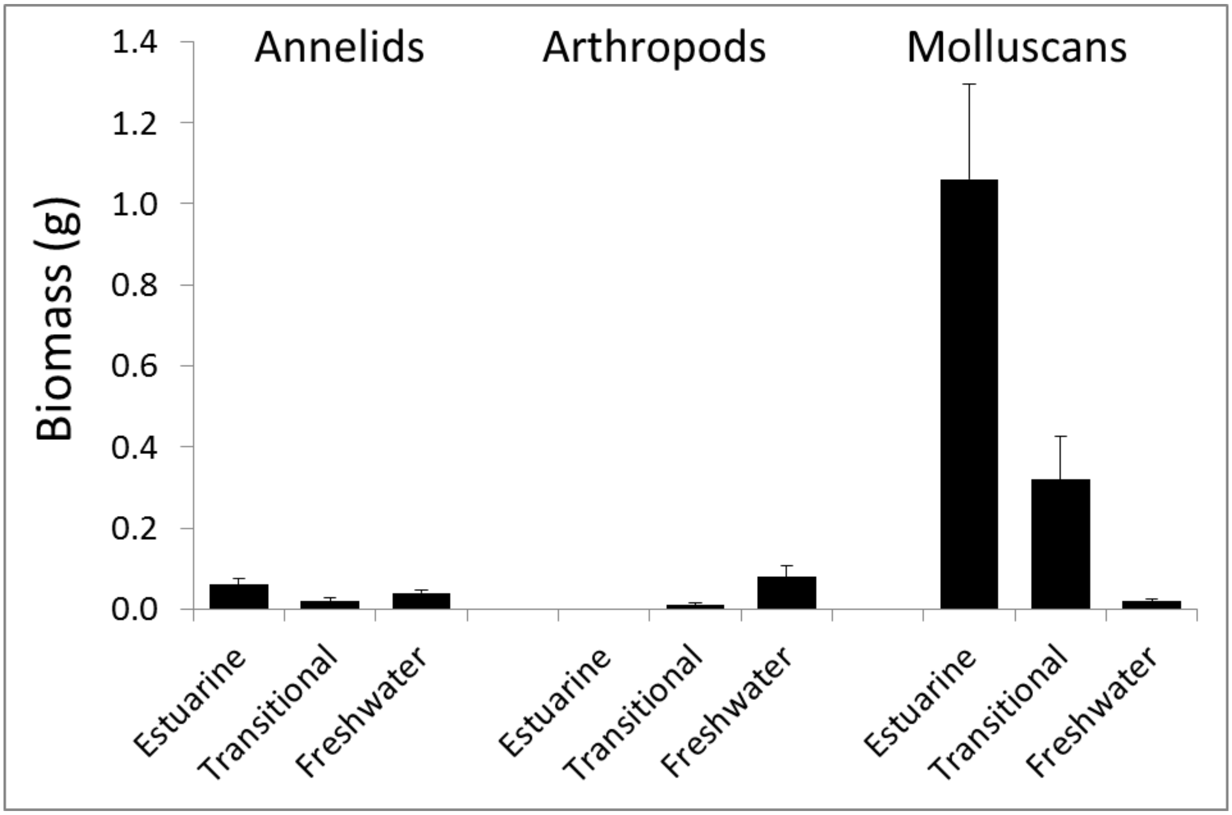
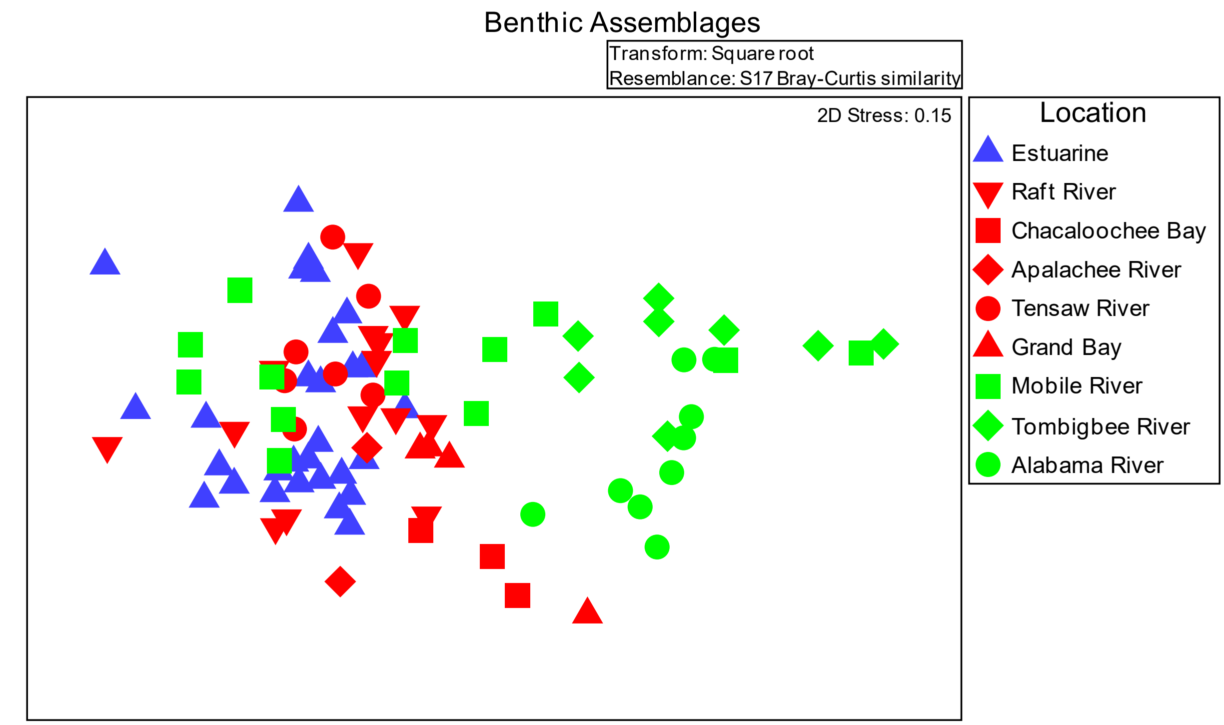


Figure 2.7. Mean fall biomass (+ standard error) of Annelids, Arthropods and Molluscs in each sampling area.



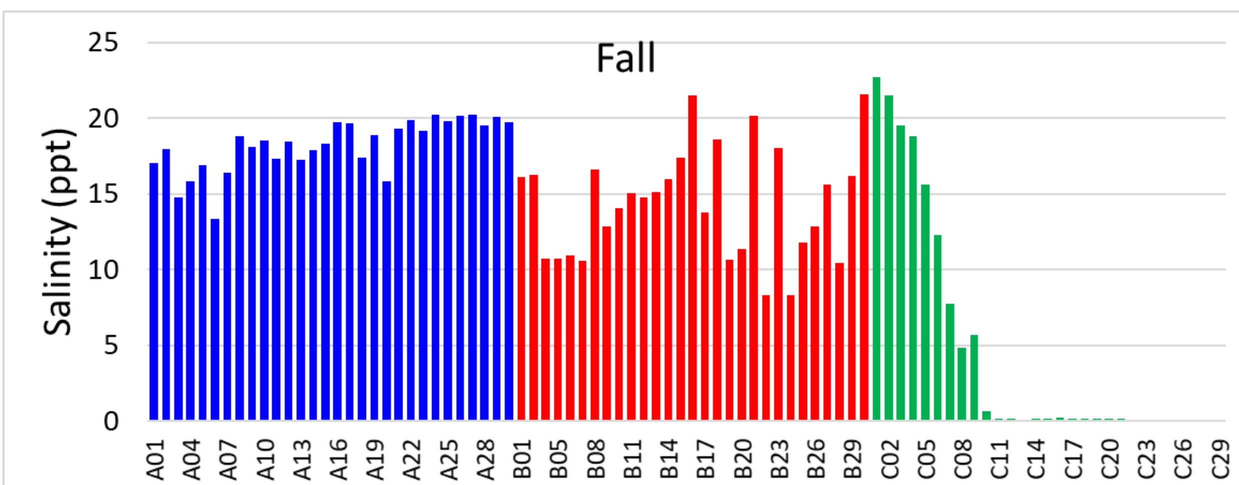


Figure 2.8. (above) Non-metric multidimensional scaling plot of samples collected during fall sampling (October 2016) in the estuarine (blue symbols), transitional (red symbols), and freshwater (green symbols) zones. (below) Salinities at each station at the time of fall sampling.

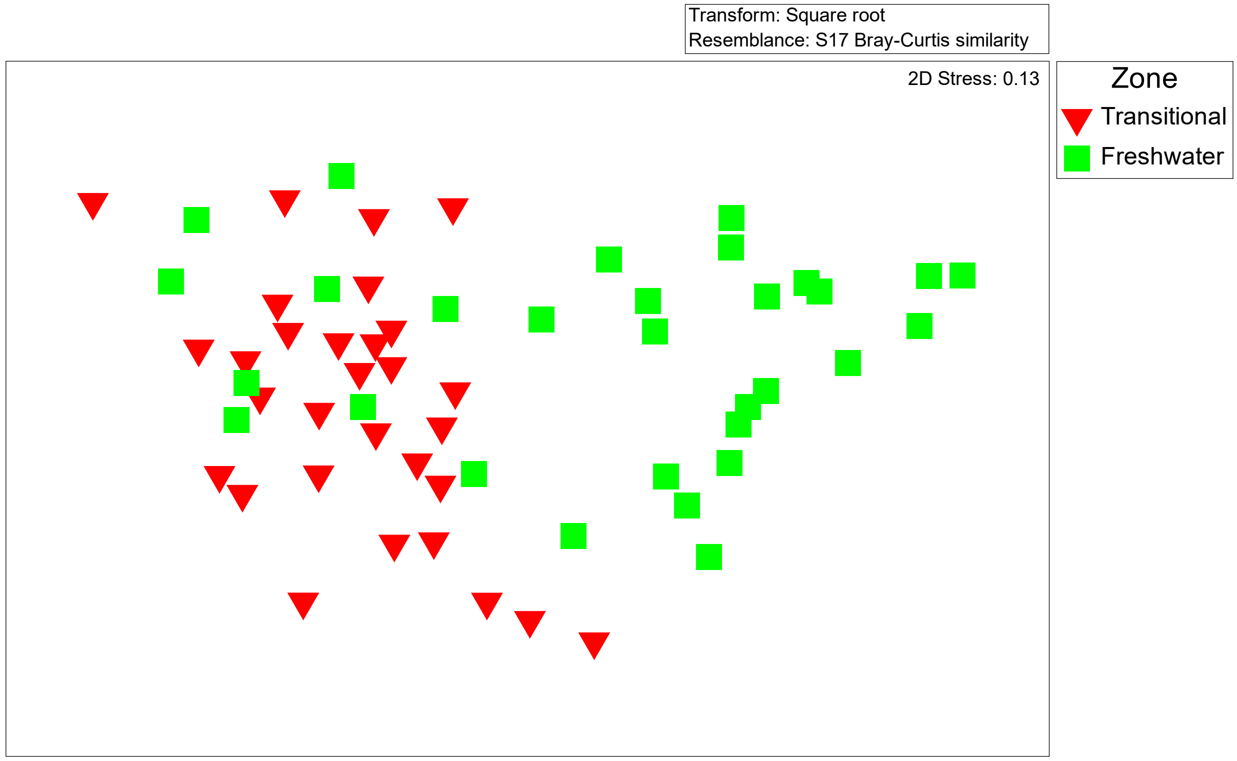


Figure 2.9. Non-metric multidimensional scaling plot of samples collected during fall sampling (October 2016) in the transitional (red symbols) and freshwater (green symbols) zones.

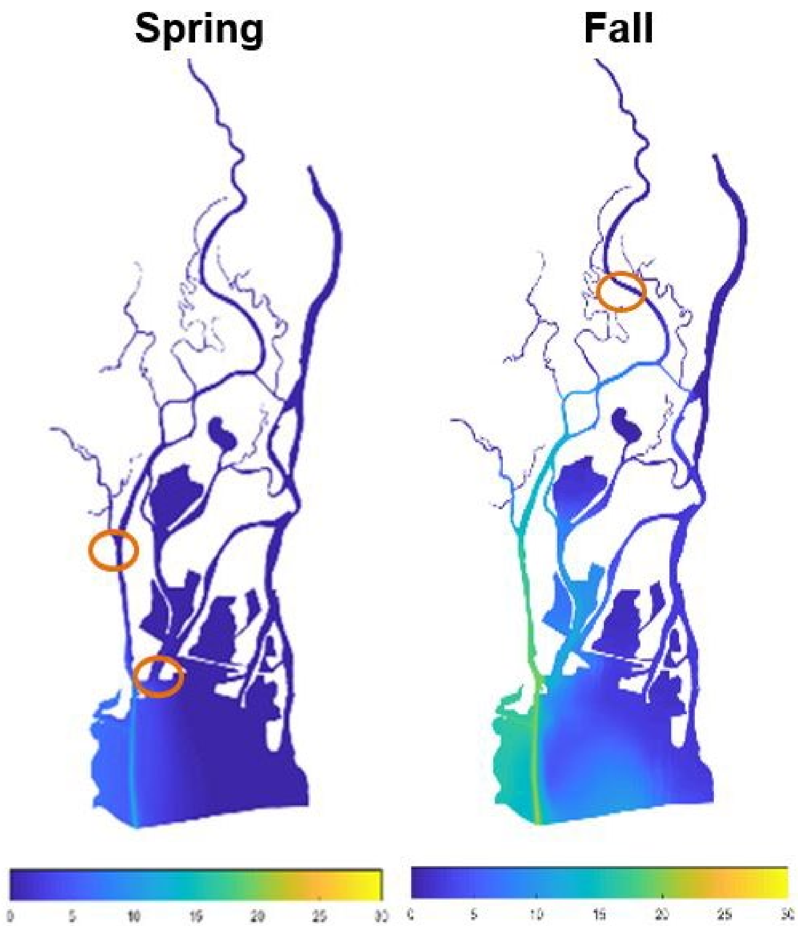


Figure 2.10. Location (orange ovals) of transitions between estuarine and freshwater benthic invertebrate communities.

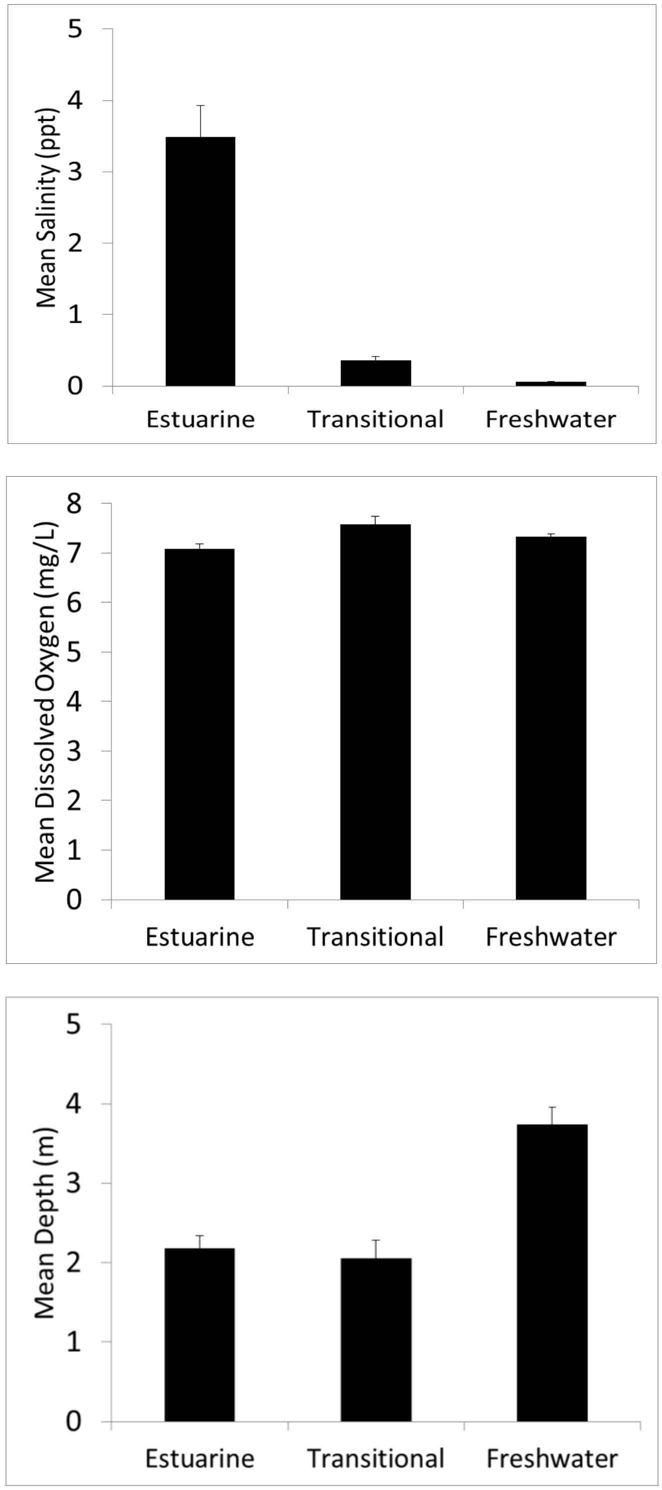
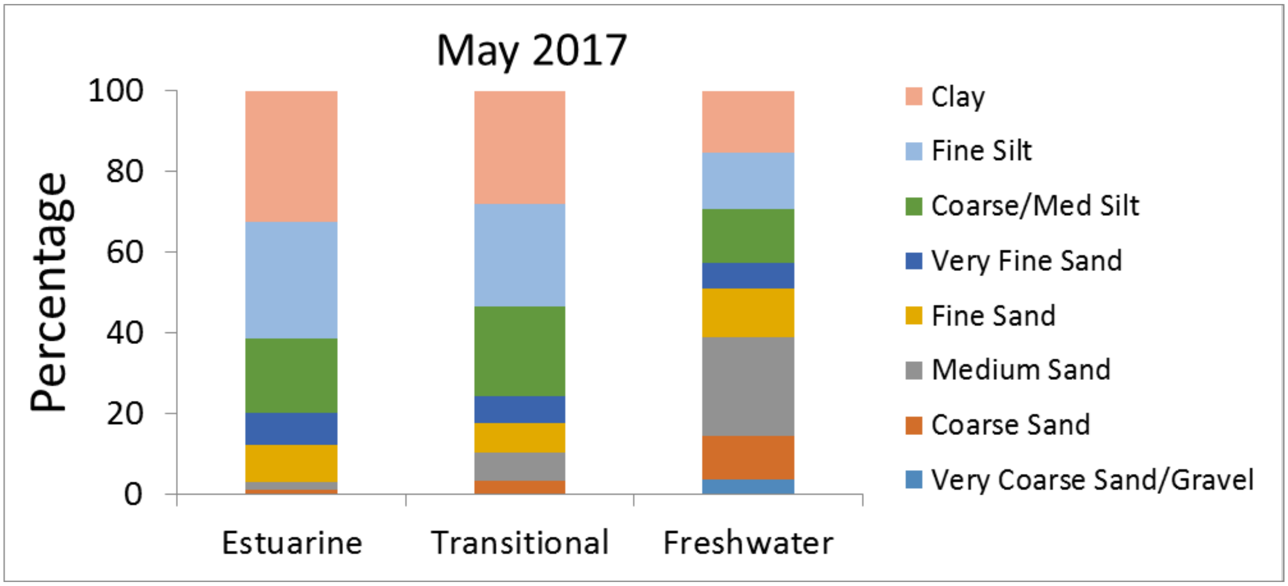


Figure 2.11. Mean (+ standard error) salinity, dissolved oxygen, and depth at stations in the estuarine, transitional, and freshwater zones during spring (May 2017) sampling.



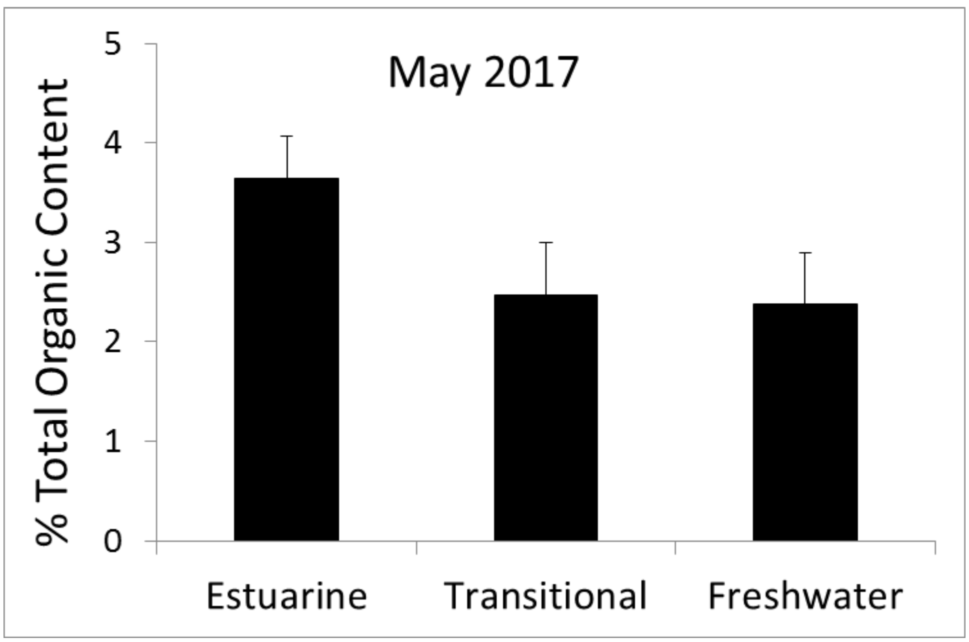


Figure 2.12. Sediment grain size distributions and % TOC in the estuarine, transitional, and freshwater zones during the spring 2017 sampling period.

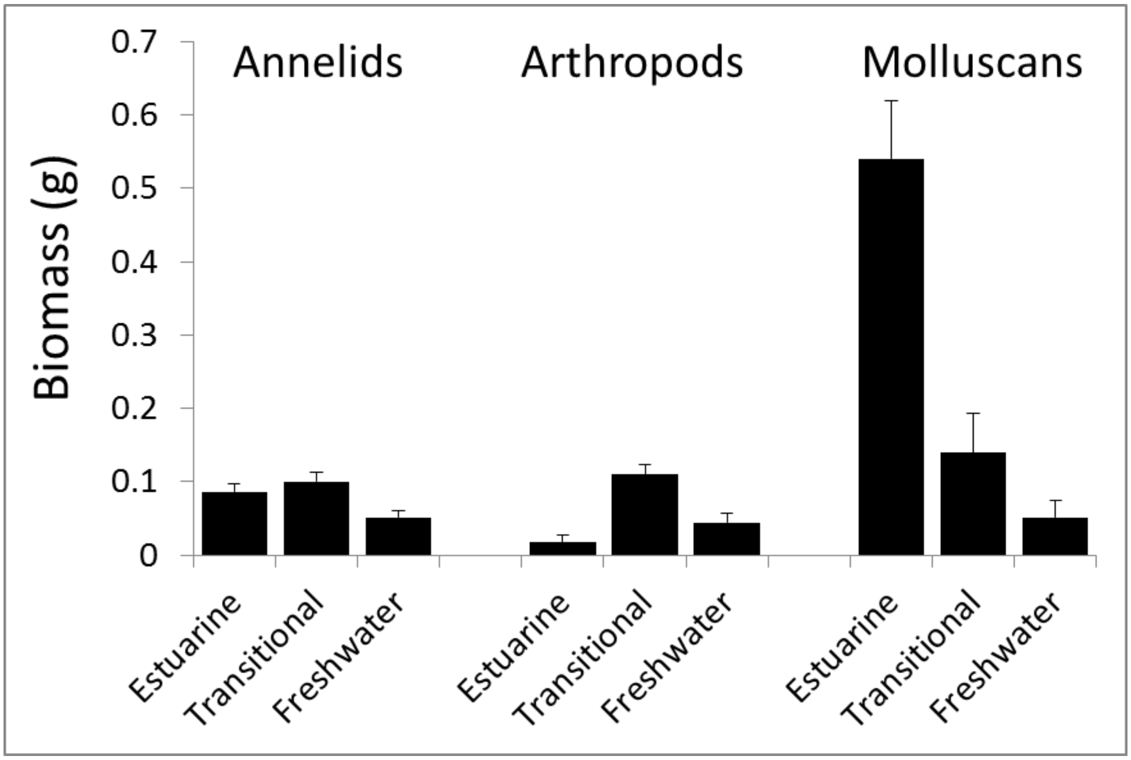
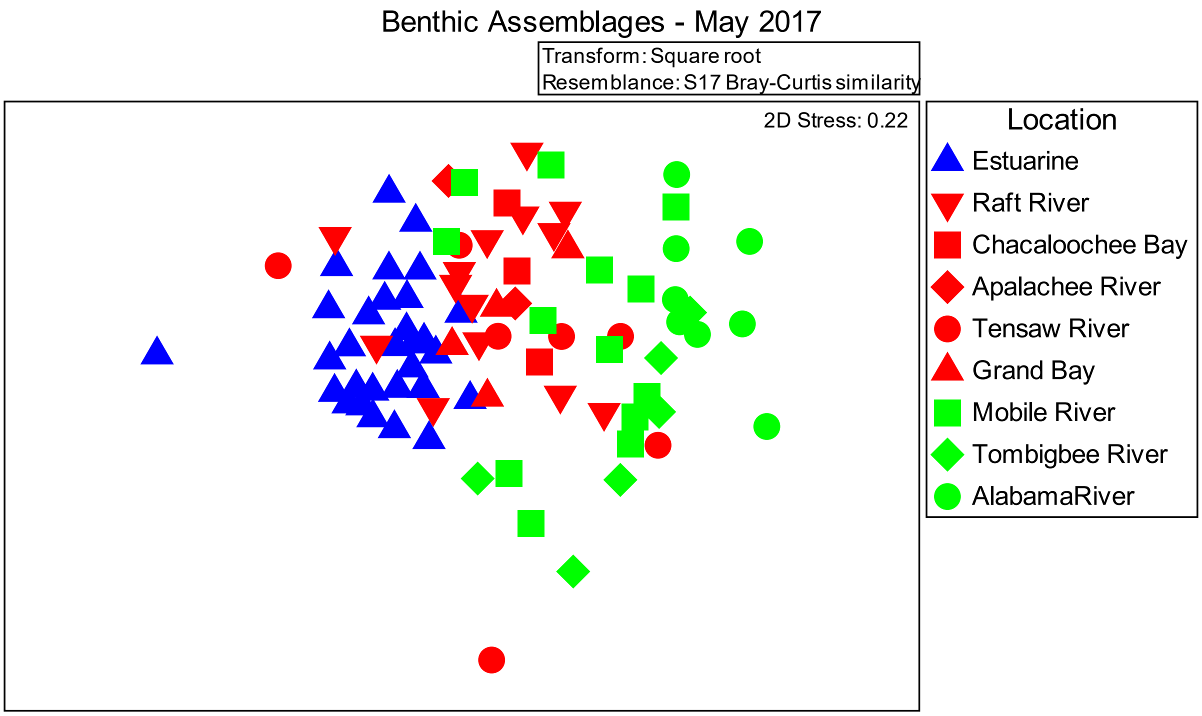


Figure 2.13. Mean spring biomass (+ standard error) of Annelids, Arthropods and Molluscs in each sampling area.



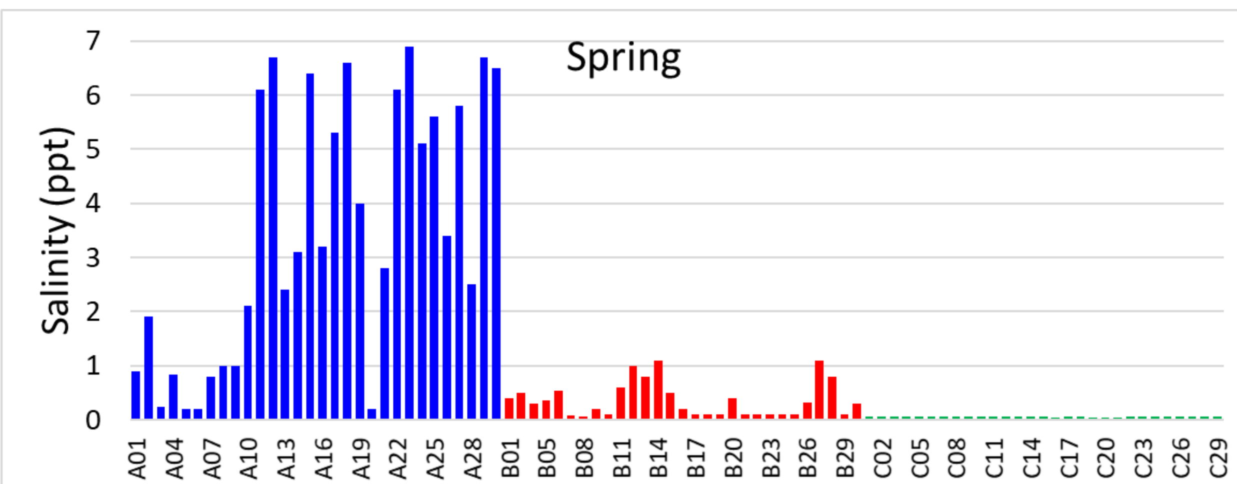


Figure 2.14. (above) Non-metric multidimensional scaling plot of samples collected during spring sampling (May 2017) in the estuarine (blue symbols), transitional (red symbols), and freshwater (green symbols) zones. (below) Salinities at each station at the time of spring sampling.

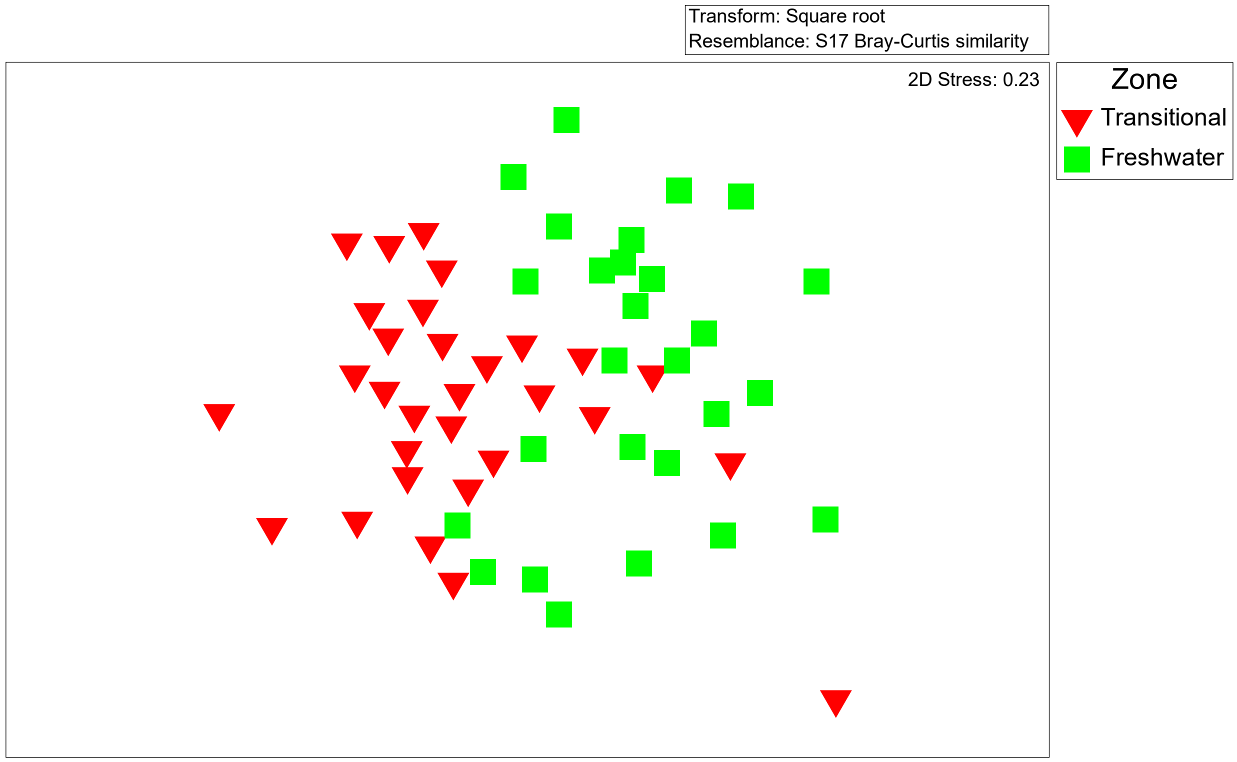


Figure 2.15. Non-metric multidimensional scaling plot of samples collected during spring sampling (May 2017) in the transitional (red symbols) and freshwater (green symbols) zones.

Chapter 3: Wetlands

Summary

Mobile Bay contains a wide variety of wetland types including freshwater, transitional and estuarine communities. As a result, extensive on-site sampling and remote sensing approaches identified and mapped a total of 3525 individual wetland features based upon vegetation assemblages. The resulting map contained 41 wetland communities occurring over an area of 72505 acres, providing the most comprehensive wetland map available for the greater Mobile Bay ecosystem. The combination of elevation, salinity, and other factors dictate the distribution of wetland community types within the study area. As a result, the analysis of potential impacts associated with the proposed navigation channel deepening and widening focused on 1) anticipated increases in water salinity following project implementation and 2) impact of sea level rise on increased wetland inundation (e.g., drowning) under a projected 0.5 m (1.64 ft) sea level rise scenario. When examining potential salinity increases, projected salinity increases remained below established thresholds for both wetland community mortality and levels associated with decreased productivity. The highest projected salinity increases in area containing wetlands (< 2.0 ppt) occurred within the lower portion of the study area and adjacent to the navigation channel, where plant communities are already adapted to higher salinity conditions. Projected 0.5 m sea level rise scenarios will increase wetland inundation within the study area, potentially shifting wetland community types and/or increasing the amount of open water features. However, given the degree of natural sea level rise impacts, additional negative effects associated with the navigation project remain negligible. As a result, project implementation is not expected to negatively impact wetlands within the study area.

3.1 Introduction

General context: Wetlands occur in areas with sufficient surface inundation or ground water saturation at a frequency and duration to support, and that under normal circumstances do support, a prevalence of vegetation adapted for life in saturated soil conditions (Environmental Laboratory 1987). As a result of these characteristics, wetlands represent one of the most productive ecological components on the landscape (Reddy and DeLaune 2008) and wetland features can be readily delineated using a combination of on-site investigation and off-site mapping approaches (Tiner 2016). Wetlands provide a number of valuable ecological functions (e.g., flood water retention, storm surge reduction, wildlife habitat) which benefit society (e.g., recreation, flood risk reduction; Novitski 1996). The distribution of wetlands and various wetland community types on the landscape is dictated by elevation, substrate, hydroperiod, hydropattern, and water composition (Cowardin et al., 1979). In particular the salinity of water supporting wetlands maintains a controlling factor in wetland zonation in many areas (Huckle et al., 2000), with salinity displaying the capacity to alter patterns of wetland community distribution and productivity in coastal and estuarine environments (Crain et al., 2004). For example, alteration of natural salinity regimes and saltwater intrusion have contributed to wetland impacts in southern Louisiana and elsewhere (Day et al. 2000; Turner 1997). Potential forcing factors leading to increased salinity include increasing storm surge frequency and intensity, channel dredging, decreased freshwater inflows, and intensive groundwater withdrawal (Hauser et al. 2015; Yuan and Zhu 2015). In areas where increased salinity occurs, wetland plant communities may display decreased productivity, shift to more salt tolerant species, or undergo conversion to open water features (Boesch et al. 1994; Brock et al. 2005). Notably, wetland floral communities and fauna living in wetland sediments are adapted to life under anaerobic (i.e., low oxygen) conditions (NRC 1995). As a result, the assessment of potential water quality changes resulting from proposed dredging activities focuses on salinity and does not evaluate the dissolved oxygen levels examined in other aquatic resource categories discussed herein (e.g., oysters, fisheries, etc).

Problem statement: Mobile Bay supports one of the largest in tact wetland ecosystems in the United States, including over 250,000 acres within the Mobile-Tensaw River Delta (AWF 2018). Wetlands within the Bay provide essential habitat for a wide variety of recreational and commercially valuable species, including rearing and cover areas for fishes and waterfowl (Chabreck 1989). Additionally, Mobile Bay contains diverse plant communities including many rare, listed, and endemic species (Stout et al., 1998). The widening and deepening of the Mobile Bay Federal Navigation Channel poses potential environmental concerns because the possible influx of saltwater into upstream areas may alter wetland habitat assemblages, distribution, or productivity. Salinity in Mobile Bay is affected by river inflow, wind, and tides as well as periodic storm surges resulting from hurricanes and other weather events (Park et al. 2014). These natural patterns of spatial and temporal salinity fluctuations resulted in the development of diverse and resilient wetland community types within Mobile Bay. However, potential changes in water quality resulting from the implementation of the proposed Navigation Channel expansion must be evaluated to determine if post-project water quality conditions will impact wetland resources.

Model purpose: This chapter characterizes baseline wetland community assemblages and distribution in estuarine, transitional, and freshwater habitats throughout Mobile Bay and the associated Delta region. Potential changes in wetland community type, distribution, and productivity are documented to determine whether and to what extent impacts may occur following channel deepening.

Model summary: Quantitative species composition data were collected at over 800 on-site locations to document the distribution and community assemblages of wetlands within the potential zone of influence of the harbor deepening project. Off-site approaches linked those ground measurements with aerial imagery and other resources to map the location and extent of each wetland community observed in the study area. Salinity tolerance classes were established for each wetland community using existing literature sources; including thresholds for decreased productivity and mortality. Hydrodynamic and water quality model results were evaluated to determine if post project conditions would increase salinity values beyond the established salinity thresholds to a degree that would alter wetland community productivity or distribution within Mobile Bay.

3.2 Methods – Model Development Process

Study Site

Mobile Bay, Alabama is located between the Fort Morgan Peninsula to the east and Dauphin Island, a barrier island on the west. Mobile Bay is 413 square miles in area, 31 miles long with a maximum width of 24 miles. The deepest (75 feet) areas of the Bay are located within the federal navigation channel, which serves Alabama's only port for ocean-going vessels, but the average depth of the bay is around 10 feet. The Mobile Bay watershed is the sixth largest river basin in the United States and the fourth largest in terms of hydrologic discharge. It drains water from portions of Alabama Georgia, Tennessee and Mississippi. Five river systems feed into the Bay including the Mobile, Tensaw, Dog, Deer, and Fowl Rivers, establishing a complex assemblage of habitats ranging from freshwater (northern portions of the Mobile-Tensaw River Delta) to increasing saline conditions as the Bay grades towards the northern Gulf of Mexico. Freshwater river discharges, and thus salinity, vary seasonally with high flows typically occurring in the late winter and early spring and low flows dominating during the summer. The lower and mid-portions of the Bay (e.g., estuarine habitats) receive seawater during normal tidal exchanges. Mobile Bay is located within Major Land Resource Area 152A – the Eastern Gulf Coast Flatwoods of Land Resource Region T - Atlantic and Gulf Coast Lowland Forest and Crop Region (NRCS 2006).

The study area utilized to evaluate wetlands focused on the central and southern portions of the Mobile Bay and the Five River Delta region, the area identified as having the highest likelihood of potential impacts associated with the proposed Navigation project (Figure 3.1). The study area included the portions of the Delta south of the Interstate 65 bridge, above which freshwater communities are dominant. The southern extent of the sampling included wetlands dominated by communities adapted to saline conditions. As a result, the study area encompasses the entire salinity gradient occurring with the Mobile Bay region, ranging from salt-intolerant bottomland hardwood forest species assemblages in the north to the halophytic plant communities common throughout coastal wetlands of the northern Gulf of Mexico.

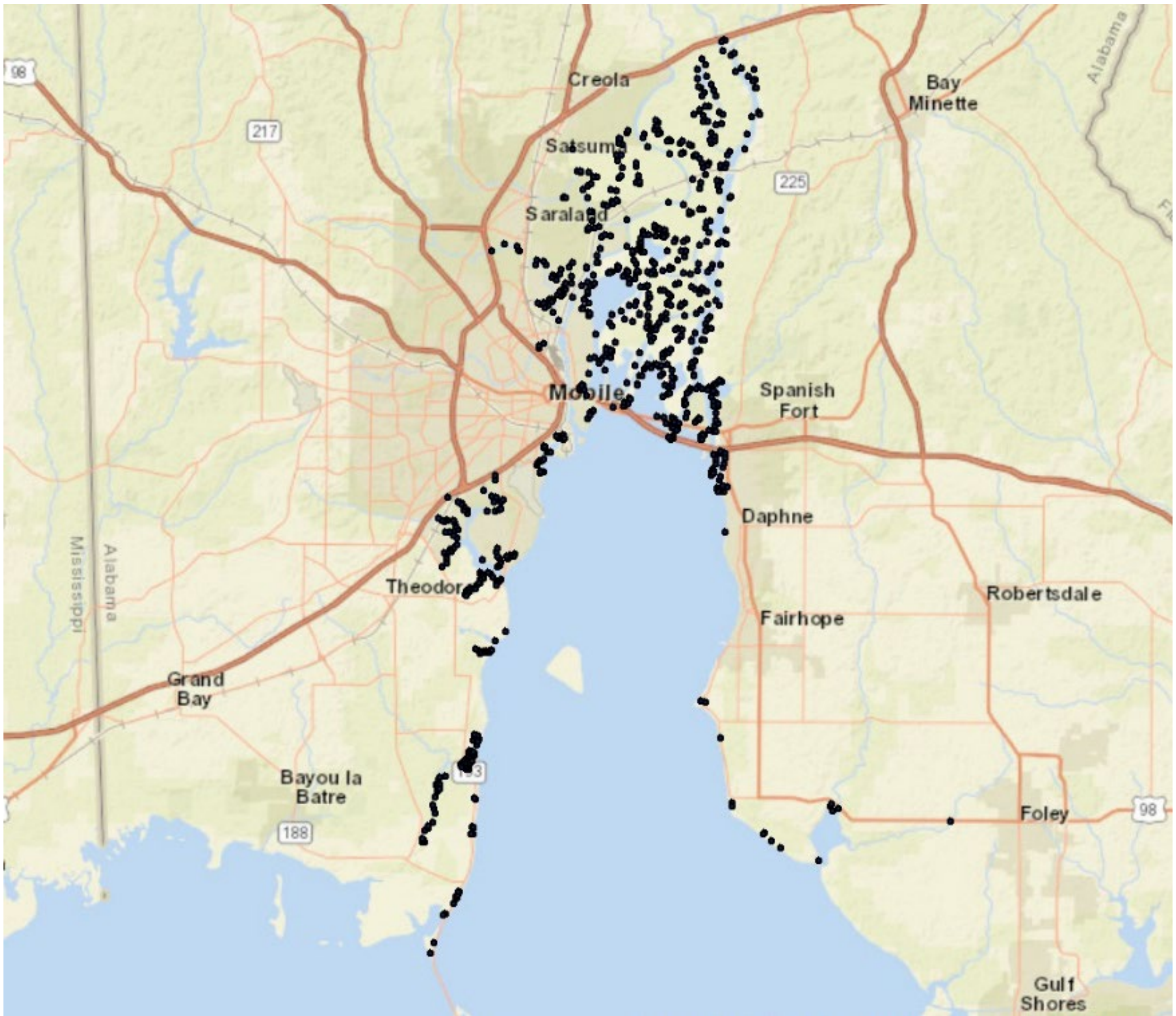


Figure 3.1. The study area focused on portions of the Mobile Bay and Five River Delta region south of the Interstate 65 bridge, encompassing the Dog river area and extending southward to Heron Bay in the west and Weeks Bay to the east. The points indicate on-site sample locations.

Wetlands within Mobile Bay developed on prograding alluvial deposits as the river sediments are discharged into the drowned Pleistocene river valley (Gastaldo 1989). As a result of the observed salinity gradient increasing from north to south, wetlands in the northern portion of the Bay are characterized by bottomland hardwood forests containing *Taxodium distichum*, *Nyssa aquatica*, *N. biflora*, *Acer sp., Carya sp., Fraxinus sp., Quercus sp., and Ulmus sp.* Herbaceous species within this zone include *Typha domingensis*, *T. latifolia*, *Sagittaria lancifolia*, *Schoenoplectus americanus*, and *Alternanthera philoxeroides*. Additionally a number of aquatic bed species (e.g., *Nuphar sp., Nelumbo lutea*) can be found adjacent to open water reaches in many wetland areas. Wetlands within the southern portion of the Delta form a transition zone of estuarine adapted, moderate salinity tolerant species dominated by a mixture of shrubs including *Baccharis glomeruliflora, B. halimifolia, Ilex sp., Morella cerifera, Persesa palustris,* and *Sabal minor*. The lower portions of the Bay include an array of moderate to high salt tolerant herbaceous species including *Spartina cynosuroides, Panicum virgatum, Cladium jamaicense, and Juncus roemerianus.* Dense nearly monotypic stands of *Phragmites karka* also occur within the study area, occupying both disturbed (i.e., near the highway 98 causeway) and natural portions of the Bay. A detailed description of species composition and distribution within Mobile Bay is provided in the results section below.

Sampling protocol – Process followed

1) On-site wetland sampling: Ground based wetland sampling occurred during November 2016, utilizing water-craft and the regional road network to access wetlands throughout Mobile Bay. Due to the warm climate and year round growing season of southern Alabama, November represents an appropriate time to conduct wetland surveys in the study area, as most vegetation maintain leaves and fruiting bodies during the fall and the full cohort of species has undergone the annual growth cycle (USDA-NRCS 2006). During that period, data from 802 distinct locations within the Bay were evaluated to enable development of a comprehensive map of wetland features within the study area (Figure 3.1). At each sample location, the species composition of each vegetation community was documented using established measurement techniques including determinations of percent groundcover, establishment of species dominance, and other factors according to the guidance provided for the Gulf and Coastal Plain regions as outlined in USACE (2010).

At a subset of study locations (65), 0.1 acres circular plots were established to further document species richness, abundance, and wetland community structure (Oliver and Larson 1996). Sample locations were selected at representative locations within specific wetland communities to characterize wetland community classes and support the large scale mapping objectives using a targeted sampling approach (Environmental Laboratory 1987). In narrow or elongate communities, plot dimensions were modified to prevent overlap with adjacent vegetation types (USACE 2010). Across all sample locations, trees were defined as woody vegetation, excluding vines, ≥ 4 in in diameter at breast height (DBH) and >20 ft in total height. Saplings/shrubs included all woody vegetation, excluding woody vines, greater than 3.2 ft in height, but less than 4 in DBH. Herbaceous plants were defined as any non-woody species, and woody species <3.2 ft in height regardless of size. Woody vines included all climbing woody vegetation greater than 3.2 ft in height, regardless of diameter. This approach allowed for determination of species richness, abundance density, and other common approaches to characterize wetland vegetation community dynamics (Tiner 2016).

2) Digitization and wetland mapping: Wetland features within the study area were digitized based on direct observations, aerial imagery interpretation, topographic maps, National Wetland Inventory data, high-resolution ortho-imagery, light detection and ranging (LiDAR) analysis, data layers available in the geospatial data gateway (https://datagateway.nrcs.usda.gov/) and other resources (USFWS 2016). The digital mapping effort utilized approaches outlined in USDA-NRCS (1996) and Berkowitz et al., (2016; 2017) to assess reflectance patterns, texture, color signatures, and other characteristics; linking study locations with known species assemblages to areas displaying similar diagnostic features (Figure 3.2 and 3.3). Digitization efforts resulted in the high resolution mapping of over 77000 acres of wetlands within the study area. Each mapped wetland feature was uploaded to an ARC-GIS database in which each feature was given a unique identifier and wetland classification code within the database attribute table.

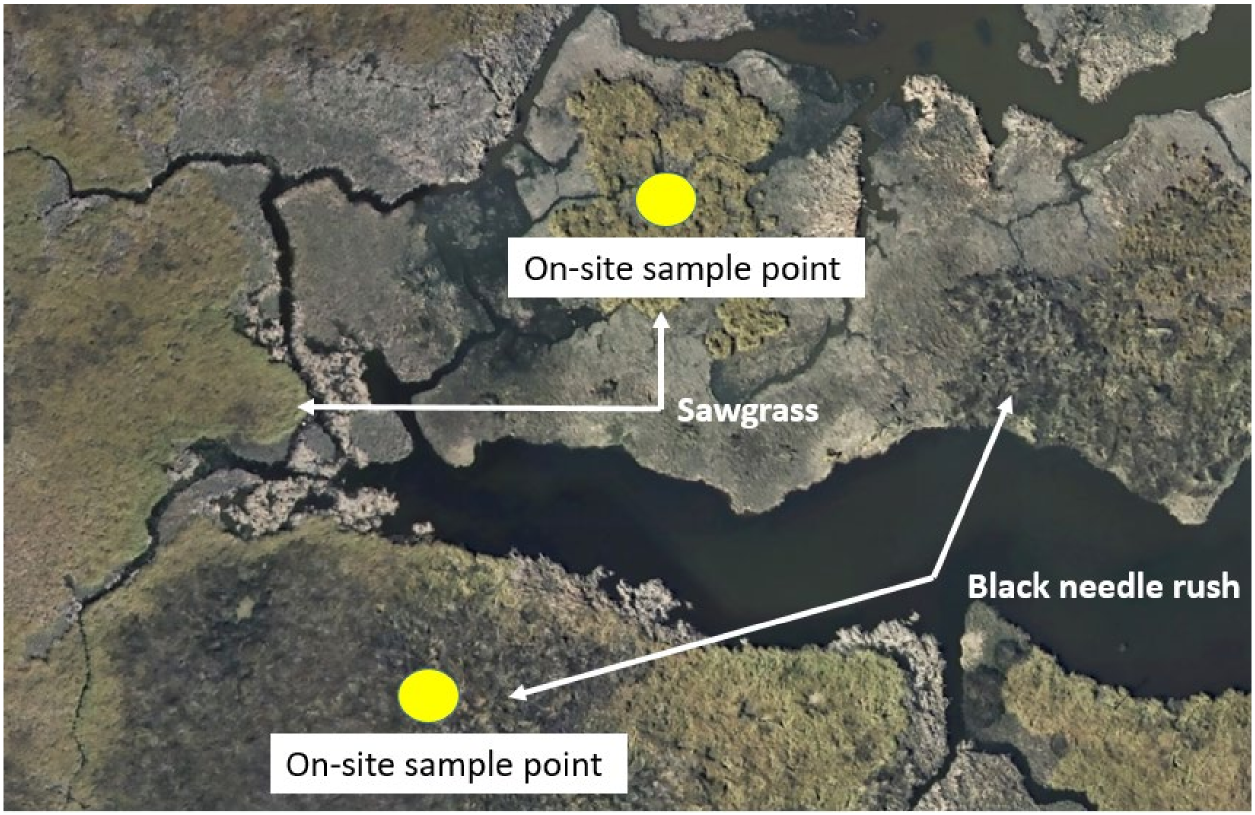


Figure 3.2: Example of wetland vegetation community mapping approach in which known on-site sample locations are used to extrapolate to un-sampled communities using distinct diagnostic features. Note that with salt-tolerant communities *Cladium jamaicense* (sawgrass) maintains a blonde color while *Juncus roemerianus* (black needle rush) displays a distinct dark color and rough texture.

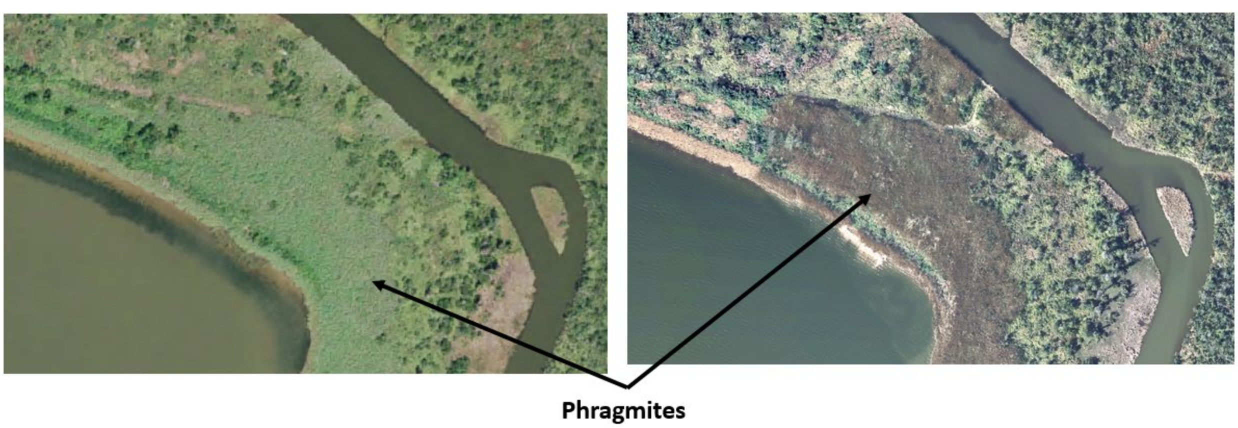


Figure 3.3. Example of wetland vegetation community mapping approach in which *Phragmites karka* occurs as largely monotypic, globular or linear shaped features located parallel to open water areas. Light green colors provide a distinctive signature for mapping using growing

season imagery, while late season and winter images display characteristic dark color. Coarse textures remain prevalent in images collected throughout the year.

3) Establishing salinity thresholds: Salinity tolerance thresholds for each wetland community type were obtained from peer reviewed journal publications and salinity classes documented within the United States Department of Agriculture (USDA) PLANTS database (https://plants.usda.gov). Two sets of species salinity thresholds were established for evaluation. First, plant species were evaluated to determine if changes in salinity would exceed available mortality thresholds. Second, plant species were evaluated to determine if changes in salinity would impact productivity and growth pattern as defined as a reduction in plant productivity. The ideal growth salinity ranges available from USDA (2000) are not associated with mortality, but represent salinity levels required to induce an estimated 10% reduction in plant productivity. For example, Crain et al. (2004) documented that *Spartina patens* (a halophyte) displayed significant mortality at very high salinity values (>60 ppt). However, the species tolerates salinities of 2.6 - 6.4 ppt (USDA PLANTS database; Table 3.2) and up to 35 ppt (Hester et al., 2005) without decreasing productivity. Similarly, *Typha domingensis* exhibited mortality at 15 ppt, while a decrease in growth was documented at salinities of 3.5 ppt (Glenn et al. 1995). In many cases, salinity based mortality thresholds were not available within the established literature as most studies of salinity focus on agricultural food crops not found in wetlands and other natural ecosystems (Downton and Läuchli 1984; Grieve 2012). In cases where no mortality thresholds were available, productivity thresholds were applied. Further, many of the plant communities examined contained a mixture of species. When mixed species communities were evaluated, the dominant species with the lowest established salinity threshold was applied. For example, wetland complexes containing a mixture of *Spartina cynosuroides* (a high salinity tolerance species adapted to values >6.4 ppt) and *Panicum virgatum* (a moderate salinity tolerant species with a preferred salinity range of 2.6 - 6.4 ppt) were evaluated using the moderate salinity productivity threshold of 2.6 – 6.4 ppt. This approach ensured that the assessment of potential wetland impacts provided a conservative estimate throughout the analysis. Once established the salinity thresholds were appended to the attribute table database for each mapped wetland feature outlined above.

4) Evaluation of potential changes in water quality: Extensive water quality and hydrodynamic data was generated to evaluate both present day (i.e., existing/baseline) conditions within Mobile Bay as well as estimated post-project conditions. Available water quality parameters included salinity, dissolved oxygen, and other factors (e.g., nutrients). For the assessment of wetland resources, potential changes in salinity were evaluated due to the fact that wetlands are adapted to saturated and anaerobic soil conditions (Vepraskas and Craft 2016). Additionally, the river systems flowing into Mobile Bay are rich in both nutrients and sediment resulting in fertile substrate within the Bay (AWF 2018), suggesting that change to the navigation channel would have little effect on other water quality parameters.

All hydrodynamic and water quality data was generated using a combination of approaches including the Geophysical Scale Multi-Block (GSMB) system, the Curvilinear Hydrodynamic in three-dimension Waterways Experiment Station (CH3D-WES) approach, and the CE-QUAL-ICM water quality component developed and maintained by the US Army Corps of Engineers Engineer Research and Development Center (Cerco and Cole 1995). Detailed model parameterization and implementation information is provided in other documentation associated with the proposed navigation project and is not reproduced herein. As a result, the section below outlines how the hydrodynamic and water quality outputs were interpreted and applied to the assessment of wetland resources within the study area.

The water quality data included baseline condition and estimated post product conditions for > 48000 individual cells organized into 30 blocks (or groups of cells) encompassing the entire area of Mobile Bay (Figure 3.4). Within each individual cell, surface water quality data was generated for three scenarios 1) baseline conditions, 2) post project implementation condition, and 3) post project condition with an estimated 0.5 m sea level projection. Scenario 3 was included in the analysis in accordance with current US Army Corps of Engineer guidance which requires incorporation of estimated sea level rise implications. A 0.5 m sea level rise projection was selected for analysis because it represents the intermediate projection for the study area.

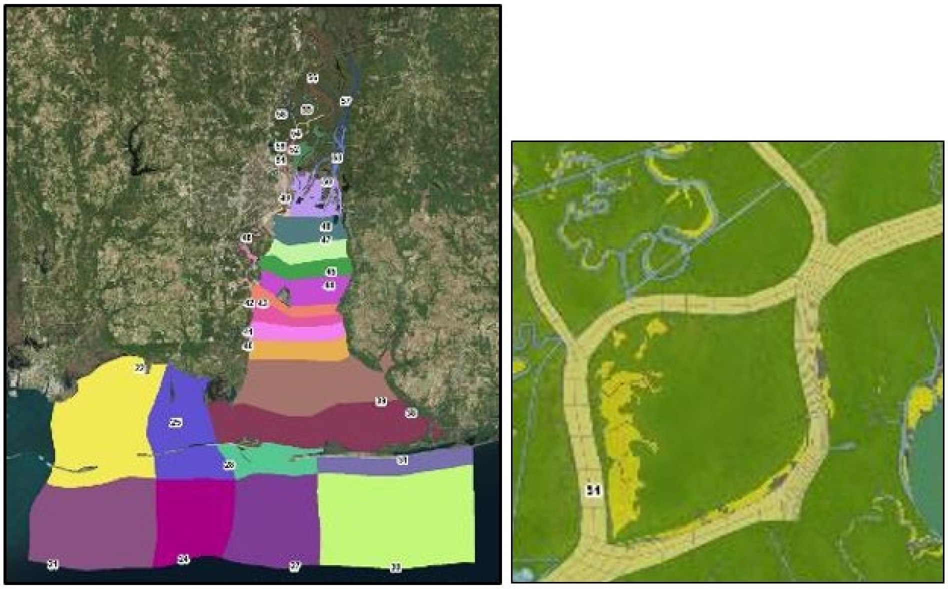


Figure 3.4. Overview of the area evaluated for potential changes in water quality, which consisted of 30 blocks (left). Each individual block was comprised of hundreds of smaller individual cells (right) each of which contained unique water quality data under the three scenarios: baseline, post project, and sea level rise. The data generated from each individual cell was linked with the nearest environmentally relevant wetland feature to evaluate potential changes in water quality resulting from the proposed navigation project.

In order to conduct the wetland assessment, the difference in monthly mean salinity values was determined between the three scenarios examined. For example, within each individual water quality cell, the difference between baseline conditions and estimated post project conditions were determined (scenario $2_{SALINITY}$ – scenario $1_{SALINITY}$). Similarly, the difference between the baseline condition and estimated sea level rise values was determined (scenario $3_{SALINITY}$ – scenario $1_{SALINITY}$). Following the determination of anticipated salinity differences between model scenarios, all cells with estimated changes in mean salinity ≥0.5 ppt for any month during the year were extracted from the grid and identified for further analysis.

A methodology was implemented to link each wetland feature within the closest cell within the study area. Specifically, any wetland feature within 1000 ft of a water quality cell within the study area was selected using a nearest neighbor feature in ARC-GIS. Salinity difference from the identified cells were then appended to attribute table of the wetland features for analysis. The link between wetland features and individual cells were evaluated to ensure that the selected cell provides a hydrologic connection to the adjacent wetland feature. This evaluation was required in areas where with high sinuosity, natural levees or other barriers, or other features that prevent the closest water quality cell from representing the source of water to the wetland feature. Once each wetland feature was linked with the appropriate cell, estimated changes in monthly salinity data were evaluated under the baseline condition, as well as under the post project implementation condition, and the post project condition plus 0.5 m sea level projection scenarios outlined above. The scenario results associated with each wetland feature were compared to the established salinity thresholds in order to identify potential impacts.

Statistical Approach – Process followed

Quantification: Extensive ground and remote sensing studies were implemented to quantify the distribution of wetland communities within the study area. For each wetland community assemblage identified, salinity tolerance thresholds were established. Water quality parameters were generated under the three scenarios described above and linked with tolerance limits for each wetland feature.

Evaluation: Descriptive statistics including monthly and seasonal mean values as well as standard deviations if the mean are reported for each wetland community. Additionally, the estimated increase in salinity was evaluated to determine of salinity tolerance limits were exceeded.

3.3 Results

3.3.1 Baseline conditions:

As discussed above, Mobile Bay contains a wide variety of wetland types. As a result, a total of 3525 individual features were identified based upon vegetation assemblages. The resulting map contained 41 wetland communities occurring over an area of 72505 acres (Table 3.1; Figure 3.5 and 2.6). The most abundant wetland community observed in the study area was the Baldcypress

– tupelo – bottomland mix which accounted for 30% of the total wetland area, mostly located in upper portions of the study area and along the north eastern shore of the Bay. Additionally, the Baldcypress – tupelo – swamp bay – palmetto – shrub mix and the Tidal shrub mix each comprised nearly 15% of the total wetland area, occurring in the upper to middle of the transition zone between freshwater and estuarine habitats. The distribution of wetlands within in the study area reflects a combination of elevation (Figure 3.7) and salinity tolerance (Table 3.2).

It should be noted that while the current report provides the most detailed assessment of wetland communities in the region, some wetland features likely contain inclusions of other communities. The scale of the study area places limitations on narrow, linear communities occurring at the contact between landscape features. Some vegetation types may not provide a distinct texture and/or color at all locations due to quality of available imagery and recent disturbance events. The northern Gulf Coast contains substantial areas dominated by various evergreen species (broadleaf and needle-leaf) due in part to sandy soils that are relatively low in nutrients, where retaining leaves for multiple years is advantageous, and mild day-time temperatures during winter that allow evergreens to carry out photosynthesis while deciduous species are dormant (Gilliam, 2014). These species can produce similar colors and textures in aerial imagery, making delineations problematic for some evergreen woody plant communities.

Table 3.1. Wetland classes, species names, and area of extent within the study area

Class Name	Representative species	Area (acres)
Baldcypress – black willow – Chinese tallow	*Taxodium distichum – Salix nigra – Triadica sebifera*	155
Baldcypress – tupelo	*Taxodium distichum – Nyssa aquatica/N. biflora*	2900
Baldcypress – tupelo – bottomland mix	*Taxodium distichum – Nyssa aquatica/N. biflora –* (*Acer sp. — Carya sp. — Fraxinus sp. — Quercus sp. — Ulmus sp*)	22687
Baldcypress – tupelo – slash pine	*Taxodium distichum – Nyssa aquatica/N. biflora – Pinus elliottii*	1114

Baldcypress – tupelo – slash pine – Atlantic white cedar	*Taxodium distichum – Nyssa biflora – Pinus elliottii – Chamaecyparis thyoides*	1018
Baldcypress – tupelo – swamp bay – palmetto – shrub mix	*Taxodium distichum – Nyssa biflora – Persea palustris* - (*Baccharis sp., Morella cerifera, Ilex sp.*)	10566
Big cordgrass	*Spartina cynosuroides*	31
Big cordgrass – switchgrass	*Spartina cynosuroides – Panicum virgatum*	442
Big cordgrass – switchgrass – bagpod	*Spartina cynosuroides – Panicum virgatum – Sesbania vesicaria*	83
Big cordgrass – switchgrass – sawgrass	*Spartina cynosuroides – Panicum virgatum – Cladium jamaicense*	1342
Black needlerush	*Juncus roemerianus*	569
Black needlerush – Big cordgrass	*Juncus roemerianus – Spartina cynosuroides*	763
Black needlerush – Big cordgrass – switchgrass	*Juncus roemerianus – Spartina cynosuroides – Panicum virgatum*	553
Bottomland mix	*Acer sp. — Carya sp. — Fraxinus sp. — Quercus sp. — Ulmus sp.*	5500
Bulrush	*Schoenoplectus californicus/S. tabernaemontani*	3
Chinese tallow – Black willow – tidal shrub mix	*Triadica sebifera – Salix nigra – Baccharis sp. – Morella cerifera*	971
Giant cutgrass	*Zizaniopsis miliacea*	263
Live oak – Magnolia – Pine (Hammock)	*Quercus virginiana – Magnolia grandiflora – Pinus elliottii/Pinus taeda*	440
Mexican water-lily	*Nymphaea mexicana*	1
Phragmites	*Phragmites karka*	2913
Pine flatwoods	*Pinus elliottii/P. palustris/P. taeda*	3862
Saltmeadow cordgrass	*Spartina patens*	5
Sawgrass	*Cladium jamaicense*	638
Sawgrass – tidal shrub mix	*Cladium jamaicense – Baccharis sp., Ilex sp., Morella cerifera, Persesa palustris, Sabal minor*	751

Slash pine – live oak – tidal shrub mix	*Pinus elliottii – Quercus virginiana –* (*Baccharis sp., Ilex sp., Morella cerifera, Persesa palustris, Sabal minor*)	109
Smooth cordgrass	*Spartina alterniflora*	3
Sweetbay – swampbay – yellow-poplar – netted chainfern	*Magnolia virginiana – Persea palustris – Liriodendron tulipifera – Woodwardia areolata*	61
Tidal shrub mix	*Baccharis glomeruliflora, B. halimifolia, Ilex sp., Morella cerifera, Persesa palustris, Sabal minor*	12511
Torpedograss	*Panicum repens*	54
Typha	*Typha domingensis*	164
Typha – arrowhead – alligatorweed	*Typha domingensis/T. latifolia – Sagittaria latifolia – Alternanthera philoxeroides*	24
Typha – bulltongue	*Typha domingensis – Sagittaria lancifolia*	321
Typha – bulltongue – three-square – alligatorweed	*Typha domingensis/T. latifolia – Sagittaria lancifolia – Schoenoplectus americanus – Alternanthera philoxeroides*	2525
Typha – bulltongue – wild-rice	*Typha domingensis – Sagittaria lancifolia – Zizania aquatica*	108
Typha – bulrush	*Typha domingensis – Schoenoplectus californicus/S. tabernaemontani*	5
Water hyacinth – water spangles – Cuban bulrush	*Eichhornia crassipes – Salvinia minima – Oxycaryum cubense*	24
Water lotus	*Nelumbo lutea*	78
Wild-rice	*Zizania aquatica*	153
Yellow pond-lily	*Nuphar advena/N. ulvaceae*	28
Total		73741

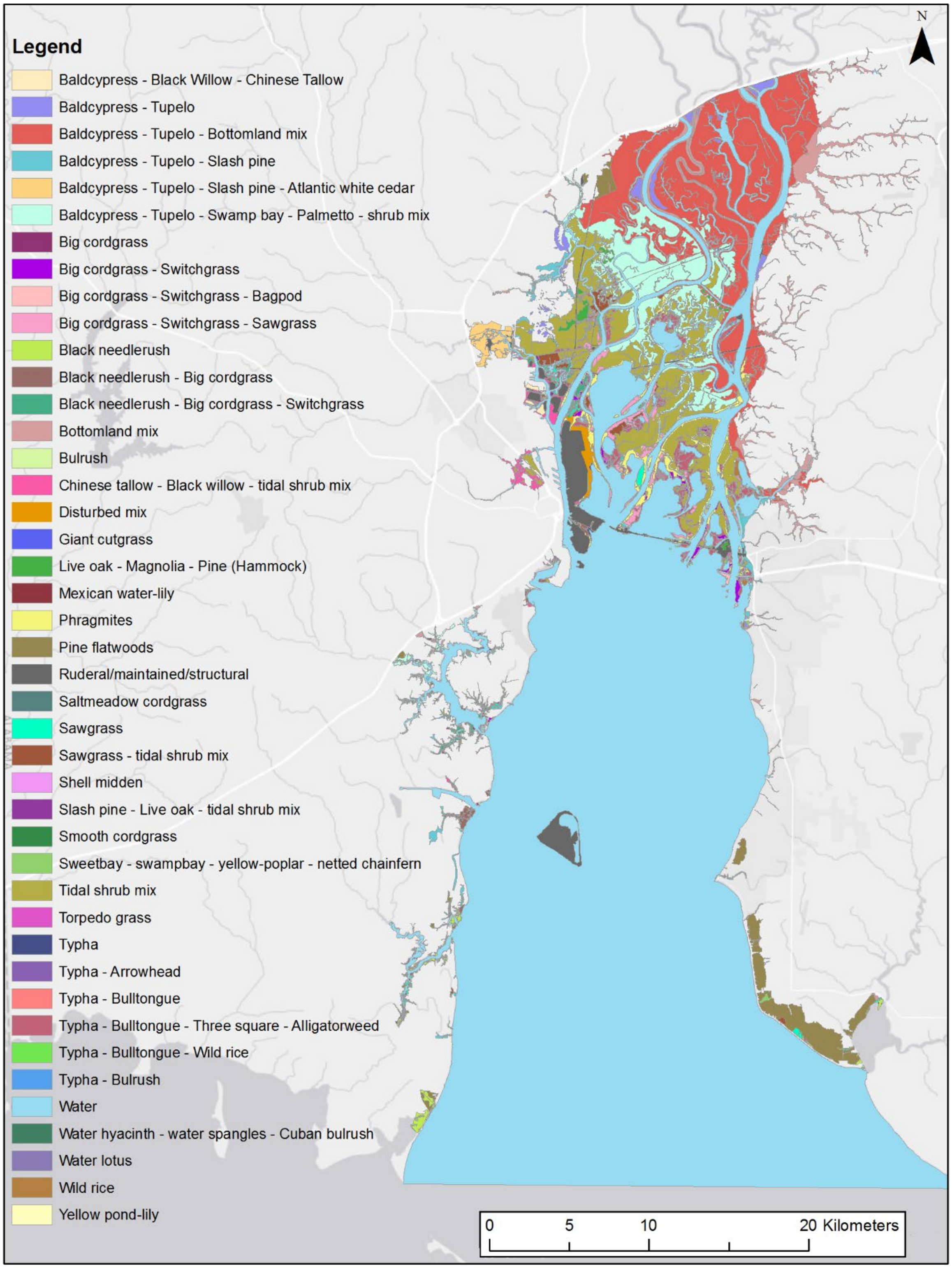


Figure 3.5. Distribution of wetland communities within the study area.

Figure 3.6. Detail of wetland community distribution within the lower Delta and upper Bay portions of the study area. The navigation channel can be seen in the center-left portion of the figure. Wetland community are identified by color using the legend provided in Figure 3.5.

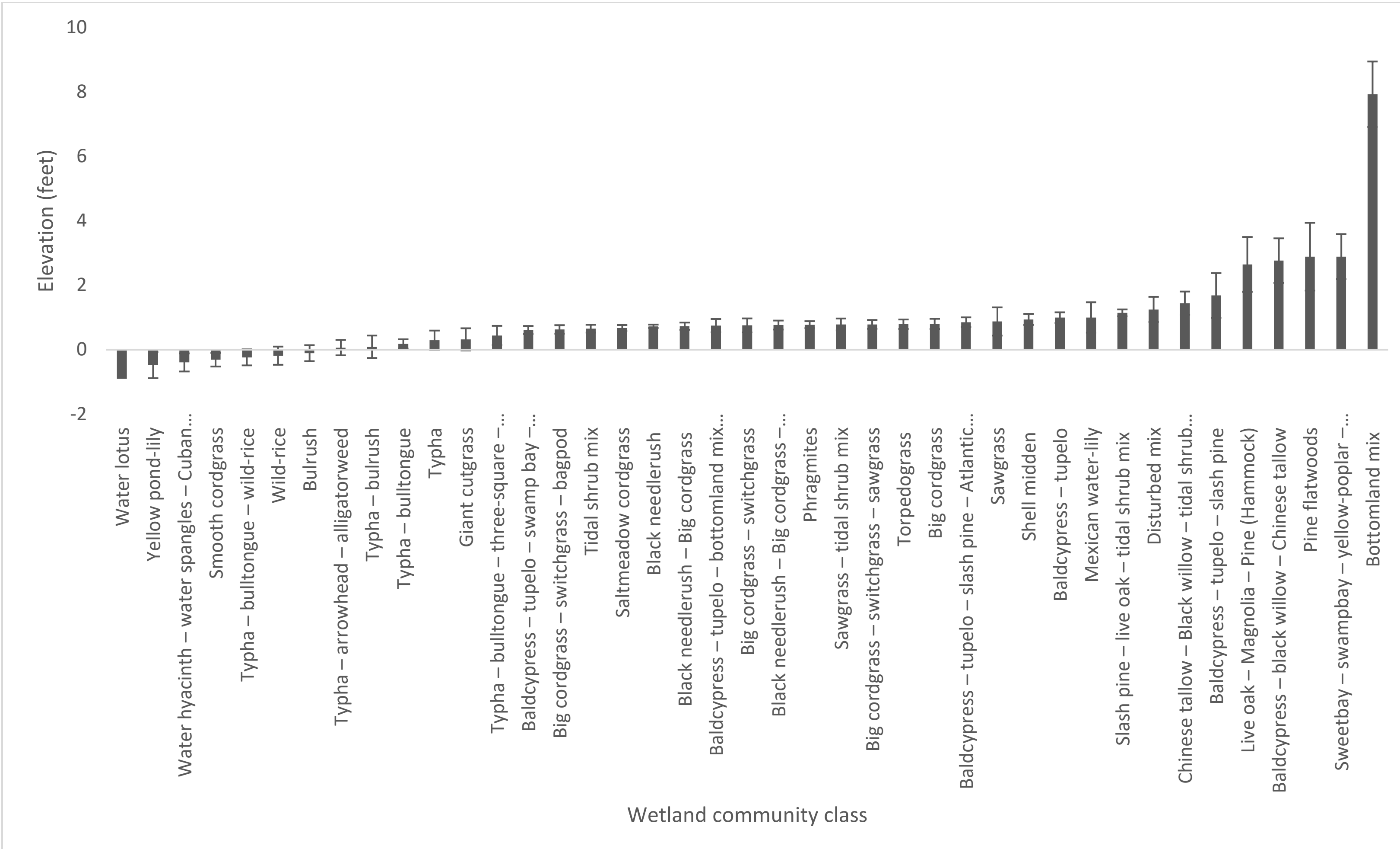


Figure 3.7. Elevation distribution (feet) of wetland community classes based upon digital elevation mapping. Error bars represent one standard deviation of the mean.

Table 3.2. Salinity tolerance ranges for each wetland plant community. Salinity thresholds are absolute values based upon ideal growth conditions and do not reflect mortality (USDA plants database).

Class name	ppt	Class name	ppt
Baldcypress – black willow – Chinese tallow	2.6-6.4	Pine flatwoods	0-1.30
Baldcypress – tupelo	1.31-2.59	Saltmeadow cordgrass	2.6-6.4
Baldcypress – tupelo – bottomland mix (Maple, Hickory, Ash, Oak, Elm)	0-1.30	Sawgrass	2.6-6.4
Baldcypress – tupelo – slash pine	1.31-2.59	Sawgrass – tidal shrub mix	2.6-6.4
Baldcypress – tupelo – slash pine – Atlantic white cedar	1.31-2.59	Slash pine – live oak – tidal shrub mix	1.31-2.59
Baldcypress – tupelo – swamp bay – palmetto – shrub mix	2.6-6.4	Smooth cordgrass	>6.4
Big cordgrass	>6.4	Sweetbay – swampbay – yellow-poplar – netted chainfern	0-1.30
Big cordgrass – switchgrass	2.6-6.4	Tidal shrub mix	2.6-6.4
Big cordgrass – switchgrass – bagpod	2.6-6.4	Torpedograss	2.6-6.4
Big cordgrass – switchgrass – sawgrass	2.6-6.4	Typha	1.31-2.59
Black needlerush	>6.4	Typha – arrowhead – alligatorweed	1.31-2.59
Black needlerush – Big cordgrass	>6.4	Typha – bulltongue	1.31-2.59
Black needlerush – Big cordgrass – switchgrass	>6.4	Typha – bulltongue – three-square – alligatorweed	1.31-2.59

Bottomland mix (Maple, Hickory, Ash, Oak, Elm)	0-1.30	Typha – bulltongue – wild-rice	1.31-2.59
Bulrush	1.31-2.59	Typha – bulrush	1.31-2.59
Chinese tallow – Black willow – tidal shrub mix	2.6-6.4	Water hyacinth – water spangles – Cuban bulrush	0-1.30
Giant cutgrass	1.31-2.59	Water lotus	0-1.30
Live oak – Magnolia – Pine (Hammock)	0-1.30	Wild-rice	0-1.30
Mexican water-lily	1.31-2.59	Yellow pond-lily	0-1.30
Phragmites	>6.4		

The following section describes of each the wetland community classes found within the study area. Common and scientific names of diagnostic species, number of features, area occupied, landscape position(s), and noteworthy co-occurring species are provided. Ruderal and non-wetland features such as hammocks that were embedded within aquatic and/or wetland features are also discussed. The diagnostic species for each class were maintained at a level that provides a recognizable assemblage based on direct visual observations, with the majority of diagnostic species having published salinity tolerance values for maximum productivity. As noted above, when conducting the wetland assessment the lowest salinity tolerance rating was applied in wetland communities exhibiting a variety of salinity tolerance classes.

<u>Baldcypress – black willow – Chinese tallow</u> (*Taxodium distichum – Salix nigra – Triadica sebifera*) occurred as eight features on approximately 154.8 acres of previously disturbed areas, typically inside berms of former disposal facilities (Figure 3.8). This community had low species richness, with the understory dominated by buttonbush (*Cephalanthus occidentalis*) and redvine (*Brunnichia ovata*).

Figure 3.8. Baldcypress – black willow – Chinese tallow forest located inside a former disposal facility, north of Mobile Harbor, Mobile County, AL.

Baldcypress – tupelo (*Taxodium distichum – Nyssa aquatica*/*N. biflora*) occurred as 72 features on 1,173.8 acres, that are freshwater to slightly brackish, and inundated seasonally to year-round. The understory was relatively sparse compared to other forest types that share these overstory species, with buttonbush and green ash (*Fraxinus pennsylvanica*) dominating the sapling/shrub stratum (Figure 3.9). Water-willow (*Justicia ovata*), arrow-arum (*Peltandra virginica*), and savanna phanopyrum (*Phanopyrum gymnocarpon*) dominated the herbaceous stratum. Sawgrass (*Cladium jamaicense*) dominated the herbaceous stratum in areas that are adjacent to slightly brackish waters, with pondcypress (*Taxodium ascendens*) frequently co-occurring.

Figure 3.9. Baldcypress – tupelo forest, dominated by water tupelo (*N. aquatica*), Baldwin County, AL.

Baldcypress – tupelo – bottomland mix (Maple, Hickory, Ash, Oak, Elm) (*Taxodium distichum* – *Nyssa aquatica*/*N. biflora* – [*Acer sp.* — *Carya sp.* — *Fraxinus sp.* — *Quercus sp.* — *Ulmus sp.*]) occurred as 72 features on 22,687.2 acres (Figure 3.10). The diagnostic species found in the tree stratum also dominated the sapling/shrub stratum. Pumpkin ash (*Fraxinus profunda*), Carolina ash (*Fraxinus caroliniana*), and swamp cottonwood (*Populus heterophylla*) frequently occurred in both the tree and sapling/shrub strata, but rarely as dominants. Dwarf palmetto typically dominated the herbaceous stratum.

This community occupies expansive areas of the northern portions of the delta. Some communities mapped as this type could potentially be separated as either "baldcypress – tupelo" or "bottomland mix"; however, broad-scale disturbances to the natural vegetation through timber harvesting have altered the corresponding texture and colors produced in both infrared and high-resolution ortho-imagery, precluding further separation based on available data.

Figure 3.10. Baldcypress – tupelo – bottomland mix adjacent to the upper Mobile River, Mobile County, AL.

Baldcypress – tupelo – slash pine (*Taxodium distichum – Nyssa aquatica/N. biflora – Pinus elliottii*) occurred as 103 features on 1,113.9 acres, often situated above tidal marshes and shrub dominated communities, or along blackwater streams (Figure 3.11). Swampbay (*Persea palustris*) and titi (*Cyrilla racemiflora*) dominated the shrub stratum. This community was mapped predominately south of I-10 and concentrated near the Dog River and Fowl River.

Figure 3.11. Baldcypress – tupelo – slash pine forest located adjacent to the Dog River, Mobile County, AL.

<u>Baldcypress – tupelo – slash pine – Atlantic white cedar</u> (*Taxodium distichum – Nyssa biflora – Pinus elliottii – Chamaecyparis thyoides*) occurred as 11 features on approximately 1,018.1 acres along acidic, blackwater streams, with the best examples adjacent to Chickasaw Creek (Figure 3.12). This community may be referred to locally as "juniper bogs" (Laderman, 1989). Sweetbay, titi, big gallberry (*Ilex coriacea*), and fetterbush (*Lyonia lucida*) dominated the shrub stratum. Royal fern (*Osmunda spectabilis*) and nettedchain fern (*Woodwardia areolata*) dominated the herbaceous stratum.

Atlantic white cedar is a distinctive component of this community and commonly occurred on stream banks, often leaning over the channel. This species is restricted to a narrow band of freshwater wetlands, typically near the coast, from Maine to Mississippi. It once covered expansive areas but is now considerably reduced due to excessive harvesting for its valuable, decay resistant wood, changes to hydrologic regime via ditching and draining, and conversion to agriculture or development (Laderman, 1989).

Figure 3.12. Baldcypress – tupelo – slash pine – Atlantic white cedar forest along Chickasaw Creek, Mobile County, AL.

Baldcypress – tupelo – swamp bay – palmetto – shrub mix (*Taxodium distichum – Nyssa biflora – Persea palustris* - [*Baccharis sp., Morella cerifera, Ilex sp.*]) occurred as 227 features occupying 10,566.2 acres. This community covered extensive areas in the central portions of the delta and as narrow bands along brackish channels on fronts and natural levees (Figure 3.13). This community is transitional to the "tidal shrub mix" community, and is defined here as having a tree stratum with ≥30 percent cover. Several species of *Ilex* were encountered in this community including yaupon (*I. vomitoria*), winterberry (*I. verticillata*), dahoon (*I. cassine*), American holly (*I. opaca*), myrtle holly (*I. myrtifolia*), and big gallberry. Dwarf palmetto typically dominated the herbaceous stratum of this community.

Figure 3.13. Baldcypress – tupelo – swamp bay – palmetto – shrub mix located adjacent to Bayou Sara, Mobile County, AL.

Big cordgrass (*Spartina cynosuroides*) occurred as 27 features on approximately 131.2 acres in the irregularly flooded zones of brackish and tidally influenced freshwater marshes (Figure 3.14). This species was typically a co-dominant component of other wetland communities and mapped here as monotypic stands in limited areas.

Figure 3.14. Big cordgrass dominated marsh, near the Dog River, Mobile County, AL.

Big cordgrass – switchgrass (*Spartina cynosuroides – Panicum virgatum*) occurred as 43 features on approximately 441.8 acres in the irregularly flooded zones of brackish and tidally influenced freshwater marshes, often above black needle rush, or co-occurring as a patchy mix.

Big cordgrass – switchgrass – bagpod (*Spartina cynosuroides – Panicum virgatum – Sesbania vesicaria*) occurred as nine features on 83.13 acres of irregularly flooded brackish marsh near the I-10 corridor (Figure 3.15). Bagpod occurred frequently as a minor component in many wetland communities throughout the study area; however, its abundance and co-dominance in the “big cordgrass – switchgrass” communities at some locations was noteworthy, and may be explained by previous disturbance activities.

Figure 3.15. Big cordgrass – switchgrass – bagpod (left) near the I-10 corridor, Baldwin County, AL; bagpod fruit (right).

Big cordgrass – switchgrass – sawgrass (*Spartina cynosuroides – Panicum virgatum – Cladium jamaicense*) occurred as 74 features on approximately 1,342.1 acres in the irregularly flooded zones of brackish and tidally influenced freshwater marshes, often above black needle rush. This community frequently transitioned upslope to the "tidal shrub mix" community.

Black needlerush (*Juncus roemerianus*) occurred as 114 features on 569.4 acres, forming monotypic stands in the irregularly flooded zones of polyhaline to oligohaline marshes (Figure 3.16). It frequently co-occurred with big cordgrass, or as a patchy mix with sawgrass and switchgrass. This species is the dominant plant of tidal marshes in the northern Gulf of Mexico (Tiner, 1993).

Figure 3.16. Black needlerush occupying the irregularly flooded zones of a brackish marsh, Mobile County, AL.

Black needlerush – Big cordgrass (*Juncus roemerianus* – *Spartina cynosuroides*) occurred as 212 features on approximately 763.1 acres in the irregularly flooded zones of polyhaline to oligohaline marshes.

Black needlerush – Big cordgrass – switchgrass (*Juncus roemerianus* – *Spartina cynosuroides* – *Panicum virgatum*) occurred as 106 features on approximately 552.9 acres in the irregularly flooded zones of polyhaline to oligohaline marshes.

Bottomland mix (Maple, Hickory, Ash, Oak, Elm) (*Acer sp.* — *Carya sp.* — *Fraxinus sp.* — *Quercus sp.* — *Ulmus sp.*) occupied 158 features on approximately 5,500.4 acres adjacent to freshwater streams (Figure 3.17). This community dominates the fronts and natural levees of large creeks and rivers, and the riparian corridors of minor tributaries to Mobile Bay. Dominant species include red maple (*Acer rubrum*), green ash, laurel oak (*Quercus laurifolia*), overcup oak, water oak (*Quercus nigra*) and American elm (*Ulmus americana*). Areas that have experienced timber harvesting within the recent past, or receive periodic natural disturbance from high flow events such as sand bars, typically included black willow, river birch (*Betula nigra*), and cottonwood (*Populus deltoides*) as dominants.

Figure 3.17. Bottomland mix adjacent to the upper Mobile River, Mobile County, AL.

Bulrush (*Schoenoplectus californicus*/*S. tabernaemontani*) occurred as six features occupying approximately 3.6 acres in the regularly flooded zones of brackish and tidally influenced freshwater marshes (Figure 3.18).

Figure 3.18. Bulrush in the regularly flooded zone of a brackish marsh near the Dog River, Mobile County, AL.

Chinese tallow – Black willow – tidal shrub mix (*Triadica sebifera – Salix nigra – Baccharis sp. – Morella cerifera*) occupied 102 features on approximately 971.3 acres, and occurred on both anthropogenic and naturally disturbed areas along channels (Figure 3.19). This community is most abundant along riparian corridors of urban and suburban areas.

Figure 3.19. Chinese tallow – black willow – tidal shrub mix near McDuffie Island, Mobile County, AL.

<u>Disturbed mix</u> occupied two features on approximately 481.8 acres near the Mobile Harbor. These sites appear to have experienced severe disturbances to the original hydrology and natural vegetation. The resultant plant community has no natural analog, and is represented by species from various communities that normally do not co-occur, especially as small disjunct patches, contrasting with the predictable zonation and large monotypic stands found in representative wetland communities.

<u>Giant cutgrass</u> (*Zizaniopsis miliacea*) occurred as 125 features on approximately 263.1 acres often forming near monotypic stands in areas of freshwater and slightly brackish marsh (Figure 3.20). This species frequently lined the margins of stream channels occurring as a narrow band (~3 ft) that could not be mapped at the scale of this effort.

Figure 3.20. Freshwater marsh dominated by giant cutgrass, Baldwin County, AL.

<u>Live oak – Magnolia – Pine (Hammock)</u> (*Quercus virginiana – Magnolia grandiflora – Pinus elliottii/Pinus taeda*) occurred as 21 features on 439.6 acres, embedded within a variety of wetland communities. These features are well-drained and often occur on deep sands (Figure 3.21). Yaupon and wax myrtle dominated the shrub stratum. Dwarf palmetto and saw palmetto (*Serenoa repens*) dominated the herbaceous stratum.

A series of dredge disposal areas located adjacent to a canal connecting the Mobile and Tensaw Rivers are included here. These sites are occupied by mature forest composed of the diagnostic species found on naturally occurring hammocks and appear to function similarly.

Figure 3.21. Live oak – Magnolia – Pine (Hammock) community located on Goat Island, Mobile County, AL.

Mexican water-lily (*Nymphaea mexicana*) occurred at a single location near Dauphin Island Parkway, and occupied 1.3 acres (Figure 3.22). This community is likely underrepresented, and may occur frequently in beaver ponds constructed on small tributaries to Mobile Bay. These open water features are conspicuous on aerial imagery but are inaccessible by boat and predominantly located on private property.

Figure 3.22. Mexican water-lily in the upper reach of Whitehouse Bayou, Mobile County, AL.

Phragmites (*P. karka*; Tropical reed) occupied 500 features on approximately 2,913.0 acres. This species often formed dense stands, frequently occurring on or near areas that appear to have been previously disturbed (Figure 3.23). The taxonomic treatment of *Phragmites* has been convoluted, with Gulf Coast populations considered to be *P. australis* (Common reed), or at the subspecific level as *P. australis ssp. berlandieri* (Subtropical reed). Ward (2010) concluded that Gulf coast populations appeared to be native and shared more morphological similarity with *P. karka* than *P. australis*. Molecular work on *Phragmites* DNA by Lambertini et al. (2012) supported Ward's findings, but suggests that there has been at least some gene flow from outside of North America, leaving its native status up for debate.

Figure 3.23. *Phragmites* along the banks of a brackish channel (left), Baldwin County, AL; *P. karka* is distinguished in part by its open, drooping inflorescence (right).

Pine flatwoods (Slash pine/longleaf pine/loblolly pine [*Pinus elliottii*/*P. palustris*/*P. taeda*]) occurred as 28 features occupying 13,862.3 acres, on level to gently sloping areas (Figure 3.24). These features were situated above high tide. In the absence of fire, most of these stands have developed a dense shrub layer dominated by yaupon, wax-myrtle, buckwheat-tree (*Cliftonia monophylla*), big gallberry, and inkberry (*Ilex glabra*). With frequent prescribed or lightning-ignited fire, the sapling/shrub stratum is reduced or sparse, with a diverse abundance of forbs and grasses. These stands represent one of the most species rich terrestrial communities found in the temperate zone (Noss, 2013).

Figure 3.24. Pine flatwoods community located near Dauphin Island Parkway, Mobile County, AL.

Ruderal/maintained/structural occurred as 160 features occupying approximately 4,715.4 acres, and consists of a variety of wetland and non-wetland features including roads, levees, utility corridors, fill, structures, and highly disturbed/managed vegetation. Utility corridors situated in naturally occurring herbaceous communities were not included here since the vegetation has the potential to develop to its natural condition.

Saltmeadow cordgrass (*Spartina patens*) occurred as five features on approximately 25.5 acres, forming near monotypic stands in the irregularly flooded zones of brackish marshes, typically above black needlerush. This community often has a distinct “cow-licked” appearance (Figure 3.25). This species did not produce a readily detectable pattern, color, or texture in aerial imagery and may occur within features mapped as other herbaceous wetland community types.

Figure 3.25. Saltmeadow cordgrass, with black needlerush in the background, adjacent to Fowl River, Mobile County, AL.

Sawgrass (*Cladium jamaicense*) occurred as 234 features occupying 638.1 acres, in the irregularly flooded zones of brackish and tidally influenced freshwater marshes (Figure 3.26). It routinely occurred immediately above stands of black needlerush, and occasionally as a mix with big cordgrass and/or switchgrass.

Figure 3.26. Monotypic stand of sawgrass in the irregularly flooded zone of a brackish marsh (left), Mobile County, AL; sawgrass inflorescence (left).

Sawgrass – tidal shrub mix (*Cladium jamaicense – Baccharis sp., Ilex sp., Morella cerifera, Persesa palustris, Sabal minor*) occurred as 29 features on 751.4 acres, as a transitional community typically between monotypic stands of sawgrass and tidal shrub communities.

Shell midden plant communities occurred on shell deposits, often embedded within various other plant communities, and at the margins of shallow bays. These areas are often small (< one hectare) and share some vegetation overlap with other adjacent communities, but are floristically unique with several species that were not recorded elsewhere (e.g., Southern flatsedge [*Cyperus thyrsiflorus*], Small-flowered buckthorn [*Sageretia minutiflora*], and Florida soapberry [*Sapindus marginatus*]). The common cultivated garden fig (*Ficus carica*) occurred on a midden near the northern shore of Grand Bay (Figure 3.27). In the absence of data, this community cannot be delineated based on aerial imagery unless the shell substrate is visible, which applied to only one site in the study area (Grand Bay). Two features totaling 3.23 acres were evaluated during this study.

Figure 3.27. Shell midden located along the northern shore of Grand Bay, Baldwin County, AL.

Slash pine – live oak – tidal shrub mix (*Pinus elliottii – Quercus virginiana* – [*Baccharis sp., Ilex sp., Morella cerifera, Persesa palustris, Sabal minor*]) occurred as 86 features on approximately 109.4 acres. This community occurred on margins and higher zones embedded in mesohaline to oligohaline marshes (Figure 3.28). Many of these features appear to be naturally occurring, but some are linear in shape and situated parallel to channels, suggesting they may be a result of minor dredging and channelization activities.

Figure 3.28. Slash pine – live oak – tidal shrub mix embedded within a mesohaline marsh, Mobile County, AL.

Smooth cordgrass (*Spartina alterniflora*) occupied eight features on approximately 3.15 acres. It occurred as monotypic stands in polyhaline marshes and as a narrow band in the regularly flooded zones of mesohaline marshes (Figure 3.29). These narrow bands could not be mapped at the scale of this effort, reducing the reported abundance and distribution of this species within the study area. This community often transitioned to black needle rush in irregularly flooded zones.

Figure 3.29. Smooth cordgrass forming a monotypic stand along the regularly flooded zone of a brackish marsh (left) at the northern shore of Polecat Bay, Mobile County, AL; smooth cordgrass inflorescence (right).

Sweetbay – swampbay – yellow-poplar – netted chainfern (*Magnolia virginiana – Persea palustris – Liriodendron tulipifera – Woodwardia areolata*) occurred as four features on approximately 61.4 acres, situated on slopes or along riparian corridors. This community may be referred to as "bayheads" locally, and likely underrepresented, as some areas encountered in the field were not mapped by USFWS-NWI (2016). Many acres of this community may be embedded in developed areas located on private property that are inaccessible. However, these wetland features are not affected by tidal events and are predominately driven by groundwater discharge to the surface, and sheetflow following rainfall events.

Yellow-poplar is widely considered a tree of mesic upland forests, but occurred frequently as a wetland component in headwater and riparian wetlands within the study area. Most of the individuals encountered in these communities appeared to be a variety that is currently undergoing taxonomic review as "Southern yellow-poplar". This variety is restricted to swamps and headwater wetlands of the outer Gulf and Atlantic coastal plain (Weakley, 2010).

Tidal shrub mix (*Baccharis glomeruliflora, B. halimifolia, Ilex sp., Morella cerifera, Persesa palustris, Sabal minor*) occurred as 266 features on approximately 12,511.8 acres, from polyhaline marshes to oligohaline areas (Figure 3.30). *Baccharis sp.* dominated areas to the near exclusion of other shrubs in areas that were polyhaline. This community was often transitional to

“Baldcypress – Tupelo – Swamp bay – palmetto – shrub mix” and is defined here as having a tree stratum with <30 percent cover. Dwarf palmetto typically dominated the herbaceous stratum but occasionally transitioned to combinations of big cordgrass, sawgrass, and/or switchgrass.

Figure 3.30. Tidal shrub mix, with scattered tree-sized individuals of swamp bay, Mobile County, AL.

Torpedograss (*Panicum repens*) occupied 20 features on approximately 53.6 acres, as near monotypic stands in the irregularly flooded zones of brackish and tidally influenced freshwater marshes (Figure 3.31). Torepdo grass is considered native to Europe but is now widely distributed across the tropics and sub-tropics. It is a pervasive weed forming dense stands and can spread rapidly by rhizomes that fragment and disperse via water (Holm et al., 1977).

Figure 3.31. Torpedograss forming a near monotypic stand in the irregularly flooded zone of a brackish marsh.

Typha (*Typha domingensis*) occurred as 77 features on approximately 163.5 acres, in the regularly flooded zones of mesohaline and oligohaline marshes (Figure 3.32). This species typically occurred as a co-dominant in other wetland communities but occupied some areas in the lower delta and along the west side of Mobile Bay, to the near exclusion of other species.

Figure 3.32. Typha dominating the regularly flooded zone of a brackish marsh, Baldwin County, AL.

Typha – arrowhead – alligatorweed (*Typha domingensis/T. latifolia – Sagittaria latifolia – Alternanthera philoxeroides*) occurred as ten features on approximately 24.2 acres in freshwater marshes near the Tensaw River (Figure 3.33).

Figure 3.33. Typha – arrowhead – alligatorweed (foreground) along the margins of the Tensaw River, Baldwin County, AL.

Typha – bulltongue (*Typha domingensis – Sagittaria lancifolia*) occupied 220 features on approximately 321.5 acres, and occurred predominantly in the regularly flooded zones of brackish and tidally influenced freshwater marshes (Figure 3.34). This zone varied considerably in width, and often formed a narrow band (<6 ft) that could not be mapped at the scale of this effort. This community is transitional to the Typha – bulltongue – three-square – alligatorweed community that dominates higher areas that flood irregularly.

Figure 3.34. Typha – bulltongue occupying the regularly flooded zone of a brackish marsh.

Typha – bulltongue – three-square – alligatorweed (*Typha domingensis/T. latifolia – Sagittaria lancifolia – Schoenoplectus americanus – Alternanthera philoxeroides*) occupied 384 features on approximately 2,524.6 acres in the irregularly flooded zones of brackish and tidally influenced freshwater marshes. This community typically has a low statured appearance due to the co-dominance of alligatorweed, and reduced abundance of Typha compared to other characteristic communities to which it has been assigned (Figure 3.35).

Figure 3.35. Typha – bulltongue – three-square – alligatorweed along the northern shore of Chuckfee Bay, Baldwin County, AL.

Typha – bulltongue – wild-rice (*Typha domingensis – Sagittaria lancifolia – Zizania aquatica*) occurred as 31 features on approximately 108.6 acres in the regularly flooded zones of brackish and tidally influenced freshwater mashes.

Typha – bulrush (*Typha domingensis – Schoenoplectus californicus/S. tabernaemontani*) occupied three features on approximately 4.6 acres, in the regularly flooded zones of brackish and tidally influenced freshwater marshes.

Water hyacinth – water spangles – Cuban bulrush (*Eichhornia crassipes – Salvinia minima – Oxycaryum cubense*) occupied 30 features on approximately 24.3 acres, forming floating rafts in slackwater areas and slow-flowing brackish and freshwater channels. Water hyacinth and water spangles are free-floating aquatics but appeared to be rafted together by the root system of the co-dominant Cuban bulrush (Figure 3.36). The formation of rafts in shallow water areas by these non-native, invasive species negatively effects habitat quantity and quality for many aquatic organisms by reducing dissolved oxygen, and altering macroinvertebrate communities (Shultz and Dibble, 2012).

Figure 3.36. Floating raft (left) composed of Cuban bulrush (right), water hyacinth, and water spangles, located in the bend of a stream channel, Baldwin County, AL.

Water lotus (*Nelumbo lutea*) occurred as 40 features on approximately 77.9 acres as an emergent aquatic in freshwater areas (Figure 3.37). Much of this community was senescent during the time of the survey, but is distinctive on growing-season aerial photography due to its relatively large, round, blue-green foliage.

Figure X-37. Water lotus (foreground) in the margins of a stream channel, Baldwin County, AL.

Wild-rice (*Zizania aquatica*) occurred as 18 features on approximately 153.0 acres in the regularly flooded zones of freshwater and brackish marshes, frequently co-occurring with the "Typha – bulltongue" community. Large stands were present on the eastern side of Mobile Bay, near the Apalachee and Blakely rivers, and D'Olive Bay. This annual species was senescent at the time of the survey, which may lead to low estimates of coverage (Figure 3.38). However, because it is an annual and relies solely on seed dispersal, its presence and abundance at a given location may be variable from year to year based on tidal events and weather-related phenomena.

Figure 3.38. A senescent stand of wild-rice near D'Olive Bay, Baldwin County, AL.

Yellow pond-lily (*Nuphar advena/N. ulvaceae*) occurred as 26 features on approximately 28.0 acres as an emergent aquatic in slackwater areas and along margins of freshwater and slightly brackish stream channels (Figure 3.39). Two distinct taxa belonging to this community are likely present in the study area. Some of the specimens that were encountered appeared to be *Nuphar ulvaceae*, a coastal plain endemic known only from Alabama, Florida, and Mississippi (Weakley, 2015). It is a state listed species in Alabama (Alabama Natural Heritage Program, 2012). Most of the specimens belonging to this community appeared to be *Nuphar advena*. This species is considered common and widely distributed throughout eastern North America (USDA 2000).

Figure 3.39. Yellow pond lily along the margin of Halls Mill Creek, Mobile County, AL.

3.3.2 Post project conditions:

General observations: The selection of appropriate water depths for the evaluation of wetland conditions is important due to season and periodic stratification that results in high salinity values at greater depths within Mobile Bay (O'Neil and Mettee 1982). Several wetland features along the eastern shore of Mobile Bay (and elsewhere) also receive freshwater inputs from seeps, groundwater discharge. And overland flow. However, the majority of wetlands within the study area exhibit surface hydrodynamic connections with adjacent open water features, with tidal fluctuations and riverine inputs driving hydrologic conditions. The water quality models utilized for the wetland assessment assessed riverine and tidal inputs, providing data for each individual cell in 10 equally spaced depth intervals. For example, if the water depth in a given cell is 10 ft, water quality data is generated in 10 – one ft increments. Similarly, if the water depth is one ft, the water quality outputs are generated in 10 – 0.1 ft increments. As a result, an analysis was conducted to evaluate differences between surface water salinities (i.e., upper increment of water quality outputs only) and the integrated upper third of the water column (i.e., top three water quality outputs). That analysis confirmed that water quality cells adjacent to wetland features displayed little or no differences in salinity between the two approaches (Figure 3.40). The close associated of the two depth intervals results from the location of wetland features in predominately shallow shoreline geomorphic positions. Where present, differences between depth intervals were associated with the navigation channel itself and other deep water areas of Mobile Bay that lack wetlands. As a result, surface water salinities were selected for all further analysis.

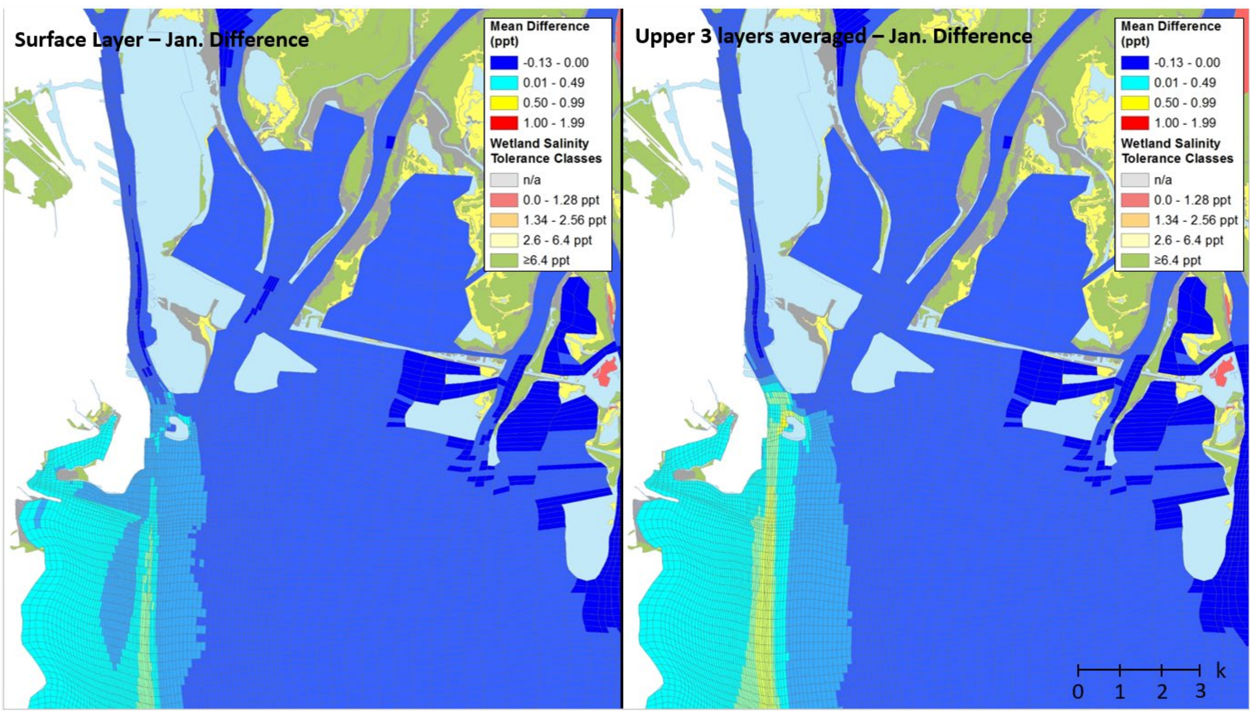


Figure 3.40. Comparison of analysis conducted using surface water salinity (left) and integrated top third of the water column (right) during January. Note that the observed differences between the two approaches is restricted to areas directly adjacent to the navigation channel (bottom left of each figure) and that no differences are observed in areas adjacent to wetland features. January data is presented, similar results occurred throughout the year.

Within the study area, species richness generally increased as salinity decreased (Gough 1994). As a result, tidally influenced freshwater marshes (≤0.5 ppt salt) in the northern portion of the study area exhibit the highest species richness found within tidal continuum. Polyhaline (18-30 ppt salt) and mesohaline (5-18 ppt salt) communities tend to have lower species richness, with several characteristic species (e.g., black needlerush, smooth cordgrass) forming predictable, abruptly zonated, monotypic stands. Oligohaline communities (0.5-5 ppt salt; "brackish") may contain a variety of species that are representative of both saline and freshwater environments (Tiner, 1993; Cowardin et al., 1979). These observations holds true within both baseline and post project conditions, as anticipated shifts in salinity remain limited. For example, within the study area most wetland features are anticipated to experience negligible increases in salinity, with only 636 (17%) of the 3525 wetland features identified displaying potential salinity increases > 0.5 ppt (herein referred to as the "potential impact area"). This represents an area of 7153 acres, or 9.8% of the 72505 acres study area. As a result, the post project conditions are not anticipated to have any potential impacts on the majority (>90 %) of wetland resources within the study area. Examining only the communities with a potential to display salinity changes > 0.5 ppt, the mean monthly surface salinity increase across all months and wetland communities was 0.68±0.38 ppt (mean ±standard deviation) with monthly minimum and maximum values of 0.2 and 1.1 ppt respectively. The text, tables 3.4 – 3.5, and figures 3.41 – 3.52 below provide data on the post project salinity conditions of wetland communities within the potential impact area, evaluating potential exceedance of mortality and productivity thresholds.

Potential mortality analysis: The wetland assessment evaluated wetland features using mortality threshold data available in the published literature (Table 3.3). Note that species specific mortality data was not available for most of the species observed. However, an examination of available mortality thresholds is provided herein for the wetland species and associated community assemblages for which data was available. Because wetlands are adapted to the conditions within the study area, the analysis evaluated potential changes in water quality as opposed to absolute water quality values. This approach accounts for local variation in salinity tolerance ranges which differ regionally and genetically across a given species or vegetation assemblage (Kozlowski 1997; Munns and Tester 2008).

To conduct the analysis, each wetland feature was linked with an adjacent water quality cell as described above to determine if the estimated changes in salinity between baseline and post project conditions would exceed the published mortality thresholds. Due to the fact that vegetative communities are adapted to local conditions (including salinity), the analysis focuses on the anticipated increase is salinity throughout. For example, if a vegetative community has a published salinity tolerance of 10 ppt and anticipated salinity increases are limited to <1 ppt, the likelihood of salinity induced mortality remains low. This approach allows for application of published mortality values, while accounting for species adaptation to local environmental conditions. Further in order to provide a conservative approach, the mortality analysis utilized the maximum estimated increase in salinity for each vegetative community. Results indicate that maximum estimated increases in salinity would not occur at the magnitude required to exceed salinity threshold ranges for the vegetation communities examined (i.e., those with available mortality data; Table 3.3). For example, across all vegetation communities containing baldcypress the maximum estimated salinity increase was 2.0 ppt (average increase of 0.7 ppt), well below the level of increase (10ppt) required to induce mortality. No cases were identified where a 2.0 ppt increase in salinity above baseline conditions would surpass the 10 ppt required to induce mortality (Table 3.4). Similarly, the under story species wax myrtle was associated with Live oak - Magnolia - Pine (Hammock) and Pine flatwoods communities and those communities exhibited a maximum estimated salinity increase was 1.5 ppt (average 0.53 ppt) and 1.3 ppt (average 0.39 ppt) respectively, below the 8.7 ppt increase required to induce mortality. This analysis suggests no wetland feature mortality thresholds would be surpassed based upon post project conditions. While the number of species with specific mortality thresholds is limited, the available species occur in a number of common wetland community types within the study area. As a result the mortality analysis accounts for 3108 acres (43%) of the 7153 acres potential impact area. Therefore the analysis provides supporting evidence that no anticipated mortality is anticipated under the post project scenario across the study area.

Table 3.3. Mortality thresholds for select species. Salinity and exposure (duration) based upon absolute values available in published literature.

Species	Salinity (ppt)	Duration (d)	Citation
Baldcypress	10	14	Conner et al. (1997)
Chinese tallow	10	42	Conner and Askew (1993)
Green ash	10	14	Conner et al. (1997)
Red maple	20-27	<5	Conner and Askew (1993)
Saltmeadow cordgrass	>60	14	Crain et al. (2004)

Smooth cordgrass	>33	Long term	USDA (2000)
Southern cattail	15	68	Glenn et al. (1995)
Water tupelo	10	14	Conner et al. (1997)
Wax myrtle	>8.7	35	Sande and Young (1992)

Table 3.4. Vegetation mortality analysis comparing the maximum estimated salinity increase with published salinity thresholds. Note that the maximum increases observed are evaluated because vegetation is adapted to local conditions throughout the study area. Because the estimated increases in salinity remain well below (<20%) the published salinity tolerance thresholds, post-project salinity increases are not anticipated to exceed the level required to induce mortality.

Species	Salinity mortality threshold (ppt)	Maximum estimated salinity increase (ppt)
Baldcypress	10	2.0
Chinese tallow	10	1.9
Green ash	10	1.5
Red maple	20-27	1.2
Saltmeadow cordgrass	>60	2.1
Smooth cordgrass	>33	2.1
Southern cattail	15	1.9
Water tupelo	10	2.0
Wax myrtle	>8.7	1.5

Wetland productivity assessment: In addition to the mortality threshold study presented above, an analysis was conducted utilizing the ideal growth tolerances developed by USDA (2000). This approach is initiated because ideal growth tolerances are available for all wetland community types occurring within the potential impact area, while only a subset of wetland plants have mortality thresholds available in published literature. These ideal growth salinity ranges available from USDA (2000) are not associated with mortality, but represent salinity levels required to induce an estimated 10% reduction in plant productivity. As a result, the assessment represents a conservative approach to evaluating potential wetland impacts. Evaluating differences in mean salinity data between baseline and post project conditions, each wetland feature within the potential impact area was assessed to determine of the growth salinity tolerance ranges were exceeded (Table 3.5). This was conducted on a monthly and seasonal basis. As noted above, the increases in salinity were evaluated to account for local adaptation to water quality conditions occurring within the study basis. For example, the Baldcypress - Black Willow - Chinese Tallow wetland community has an estimated ideal salinity tolerance range of 2.6-6.4 ppt. Estimated salinity increases are limited to 0.11, 0, 0.25, and 0.44 during winter, spring, summer and fall respectively. As a result, no negative impacts to wetland productivity are anticipated in that community. Examining the data in Table 3.5, none of the estimated salinity increases within the potential impact area were of a magnitude required to exceed the salinity tolerance threshold ranges suggesting that no impacts to wetland productivity will result under the post project conditions. To emphasize these findings figures were generated for each season within the upper (Figures 2.41-2.44), central (Figures 2.45-2.48), and southern (Figures 2.49-2.52) portions of the study area. These images provide seasonal visual representations of post project conditions representing predominantly fresh, intermediate, and estuarine wetland plant community assemblages. Note that within each figure, the estimated changes in salinity remain below the salinity tolerance threshold ranges identified for individual wetland features.

Table 3.5. Mean estimated post-project seasonal change in salinity, standard deviation for each vegetation community (all units are ppt). Salinity tolerances for optimal growth are also provided. Note that anticipated increases in salinity are utilized to account for adaptation of vegetative communities to local conditions. In no cases are salinity increases observed at a magnitude to induce salt stress.

Wetland community	Salinity tolerance	Winter	Spring	Summer	Fall
Baldcypress - Black Willow - Chinese Tallow	2.6-6.4	0.11, 0.2	0, 0	0.25, 0.18	0.44, 0.14
Baldcypress - Tupelo	1.31-2.59	1.09, 0.23	0.78, 0.21	0.98, 0.17	1.29, 0.12
Baldcypress - Tupelo - Slash pine	1.31-2.59	0.8, 0.35	0.61, 0.07	0.8, 0.11	1.19, 0.01
Baldcypress - Tupelo - Swamp bay - Palmetto - shrub mix	2.6-6.4	0.68, 0.42	0.57, 0.01	0.7, 0.05	1.05, 0.06
Big cordgrass	>6.4	0.66, 0.43	0.39, 0.1	0.86, 0.22	1.21, 0.1
Big cordgrass - Switchgrass	2.6-6.4	0.17, 0.22	0.04, 0.01	0.32, 0.19	0.53, 0.09
Big cordgrass - Switchgrass - Sawgrass	2.6-6.4	0.29, 0.27	0.16, 0.01	0.41, 0.16	0.64, 0.02
Black needlerush	>6.4	0.84, 0.26	0.61, 0.16	0.87, 0.2	1.22, 0.05
Black needlerush - Big cordgrass	>6.4	0.94, 0.35	0.65, 0.16	0.97, 0.22	1.37, 0.04
Black needlerush - Big cordgrass - Switchgrass	>6.4	0.71, 0.33	0.47, 0.11	0.84, 0.29	1.21, 0.07
Bottomland mix	0-1.30	0.63, 0.38	0.53, 0.03	0.65, 0.06	0.98, 0.05
Bulrush	1.31-2.59	0.56, 0.36	0.45, 0.01	0.56, 0.06	0.88, 0.05

Chinese tallow - Black willow - tidal shrub mix	2.6-6.4	0.6, 0.35	0.35, 0.1	0.76, 0.28	1.01, 0.09
Giant cutgrass	1.31-2.59	0.72, 0.39	0.61, 0.01	0.7, 0.07	1.05, 0.06
Live oak - Magnolia - Pine (Hammock)	0-1.30	1.13, 0.3	0.82, 0.28	1.03, 0.19	1.41, 0.13
Mexican water-lily	1.31-2.59	1.14, 0.17	0.82, 0.27	1.02, 0.21	1.27, 0.12
Phragmites	>6.4	0.48, 0.3	0.26, 0.08	0.6, 0.23	0.88, 0.06
Pine flatwoods	0-1.30	0.27, 0.09	0.2, 0.04	0.45, 0.2	0.6, 0.12
Sawgrass	2.6-6.4	0.54, 0.27	0.38, 0.04	0.59, 0.13	0.88, 0.03
Sawgrass - tidal shrub mix	2.6-6.4	0.41, 0.23	0.27, 0.03	0.49, 0.16	0.73, 0.05
Slash pine - Live oak - tidal shrub mix	1.31-2.59	0.97, 0.3	0.7, 0.18	0.99, 0.22	1.36, 0.04
Smooth cordgrass	>6.4	0.53, 0.4	0.27, 0.07	0.66, 0.25	0.99, 0.09
Sweetbay - swampbay - yellow-poplar - netted chainfern	0-1.30	0.08, 0.07	0.03, 0.03	0.32, 0.28	0.39, 0.17
Tidal shrub mix	2.6-6.4	0.68, 0.29	0.47, 0.11	0.76, 0.2	1.09, 0.03
Torpedo grass	2.6-6.4	1.14, 0.17	0.82, 0.27	1.02, 0.21	1.27, 0.12
Typha	1.31-2.59	0.53, 0.38	0.37, 0.03	0.6, 0.13	0.91, 0.03
Typha - Bulltongue	1.31-2.59	0.42, 0.32	0.31, 0.01	0.49, 0.1	0.75, 0
Typha - Bulltongue - Three square - Alligatorweed	1.31-2.59	0.13, 0.21	0.01, 0.01	0.24, 0.16	0.46, 0.07
Typha – Bulrush	1.31-2.59	0.84, 0.54	0.47, 0.15	1.08, 0.43	1.64, 0.27

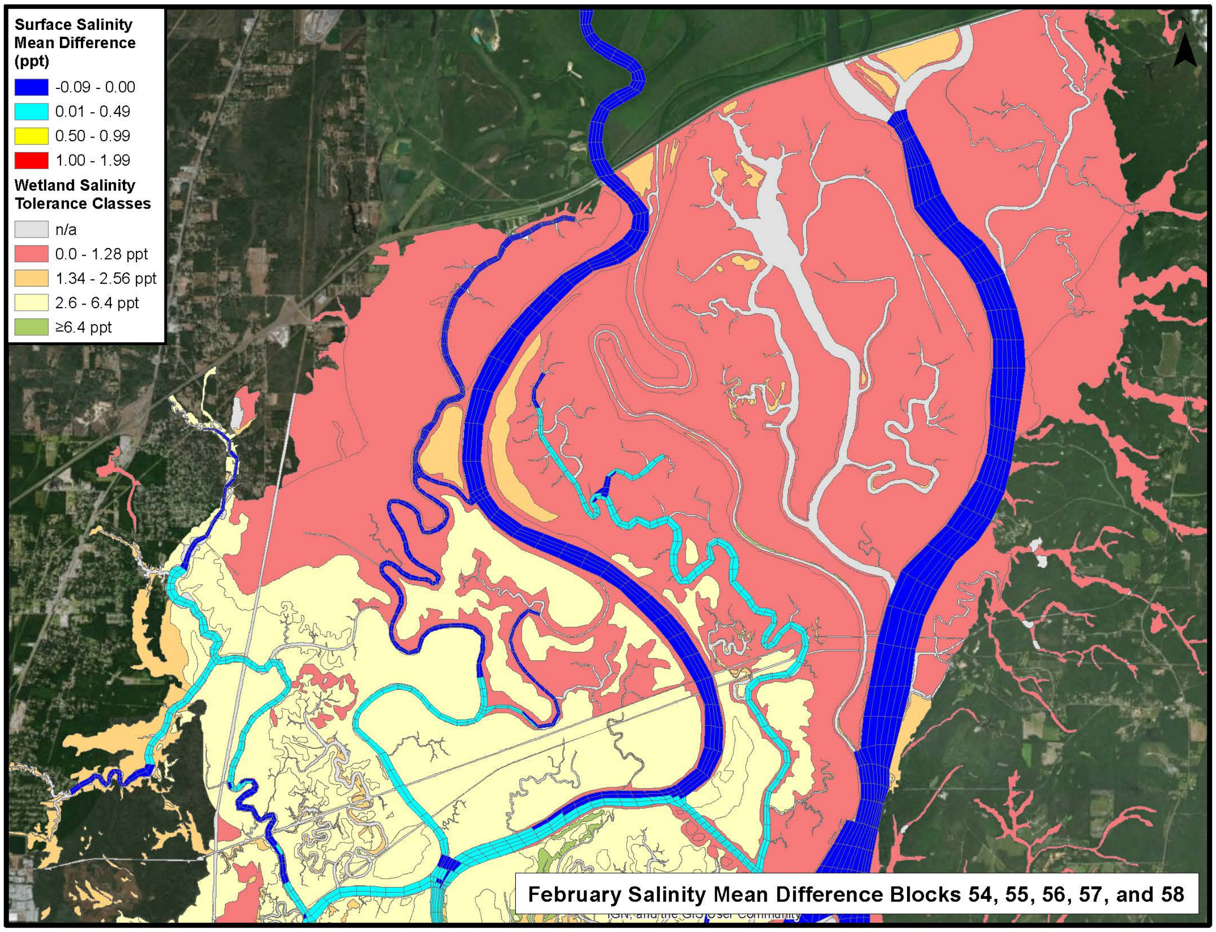


Figure 3.41. Estimated increase in salinity during the winter period (February data shown for example) within the upper (freshwater) portion of the study area. Note that estimated salinity increases are limited to 0.0, or <0.5 ppt. In areas where salinity increases may occur, wetland communities are adapted to predicted conditions. Map units designated n/a include upland habitats, highly disturbed and developed areas (e.g., historic fill, roads), and open water areas not addressed in the model domain.

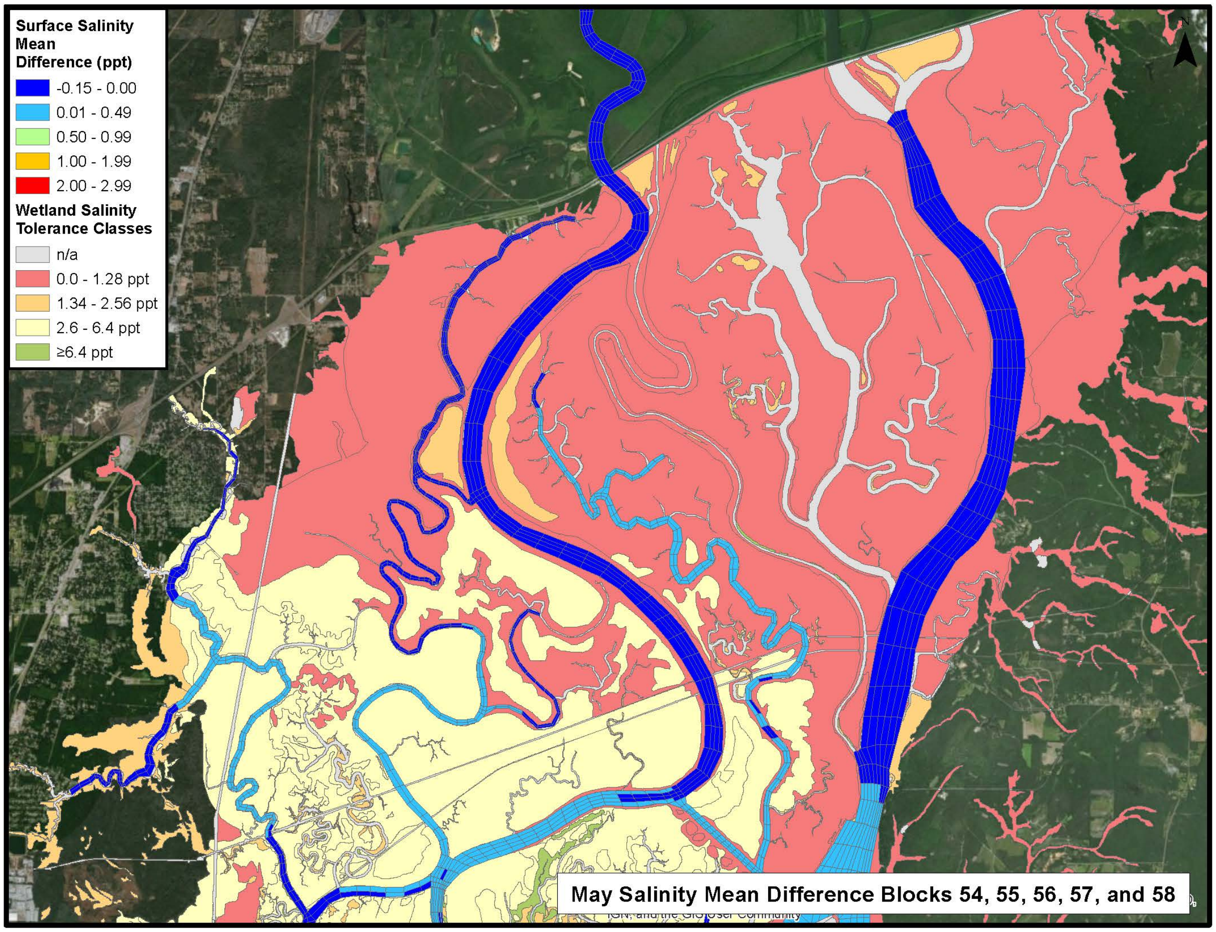


Figure 3.42. Estimated increase in salinity during the spring period (May data shown for example) within the upper (freshwater) portion of the study area. Note that estimated salinity increases are limited to 0.0, or <0.5 ppt. In areas where salinity increases may occur, wetland communities are adapted to predicted conditions. Map units designated n/a include upland habitats, highly disturbed and developed areas (e.g., historic fill, roads), and open water areas not addressed in the model domain.

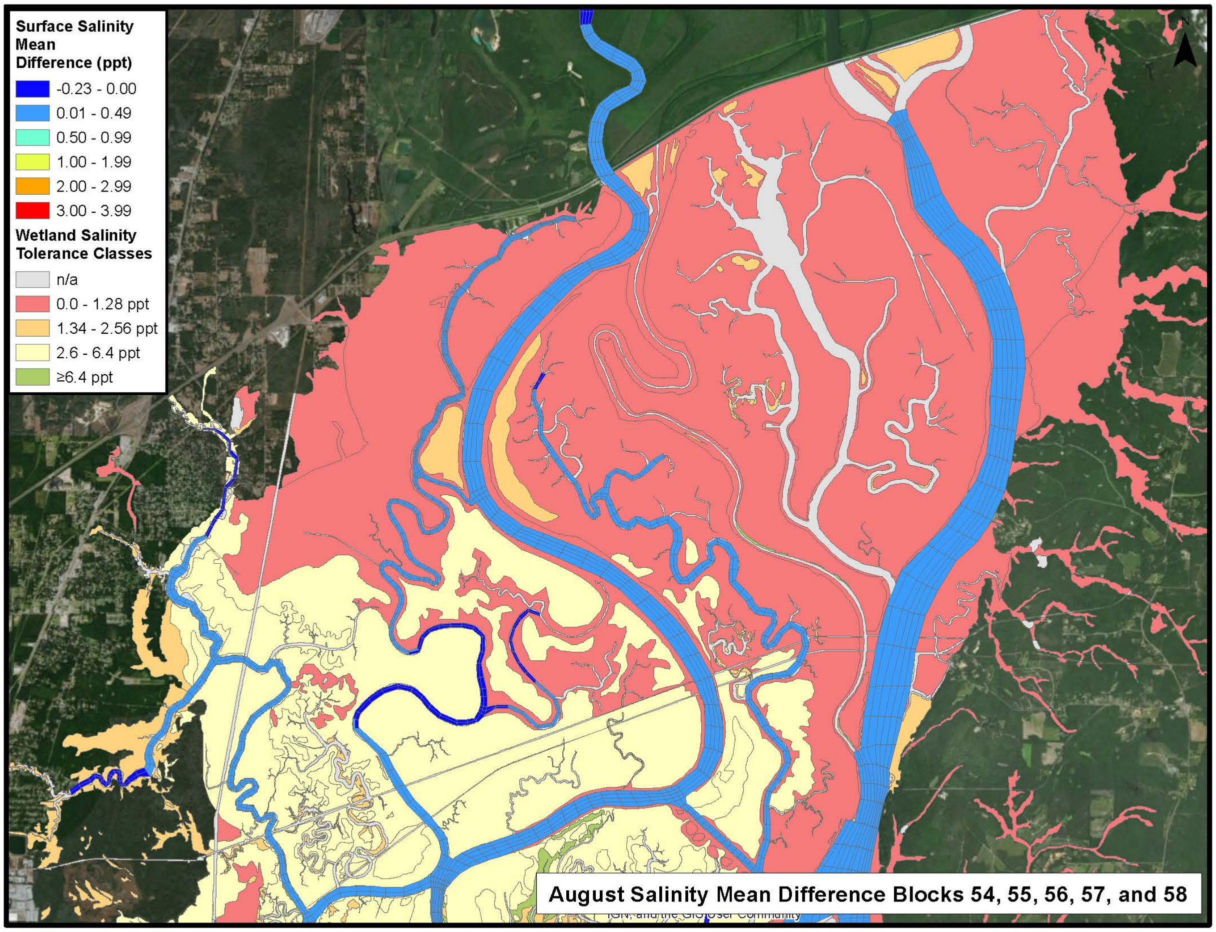


Figure 3.43. Estimated increase in salinity during the summer period (August data shown for example) within the upper (freshwater) portion of the study area. Note that estimated salinity increases are limited to 0.0, or <0.5 ppt. In areas where salinity increases may occur, wetland communities are adapted to predicted conditions. Map units designated n/a include upland habitats, highly disturbed and developed areas (e.g., historic fill, roads), and open water areas not addressed in the model domain.

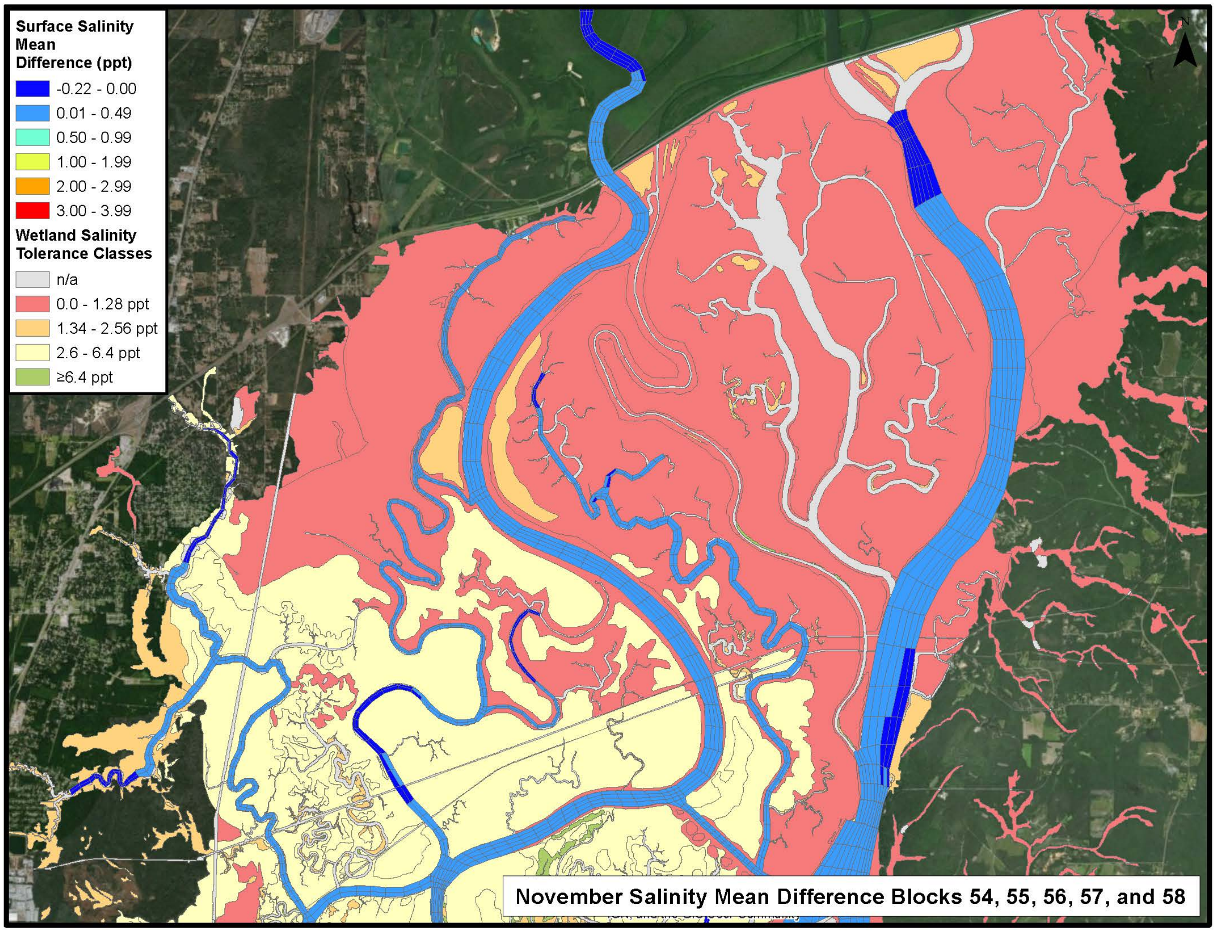


Figure 3.44. Estimated increase in salinity during the fall period (November data shown for example) within the upper (freshwater) portion of the study area. Note that estimated salinity increases are limited to 0.0, or <0.5 ppt. In areas where salinity increases may occur, wetland communities are adapted to predicted conditions. Map units designated n/a include upland habitats, highly disturbed and developed areas (e.g., historic fill, roads), and open water areas not addressed in the model domain.

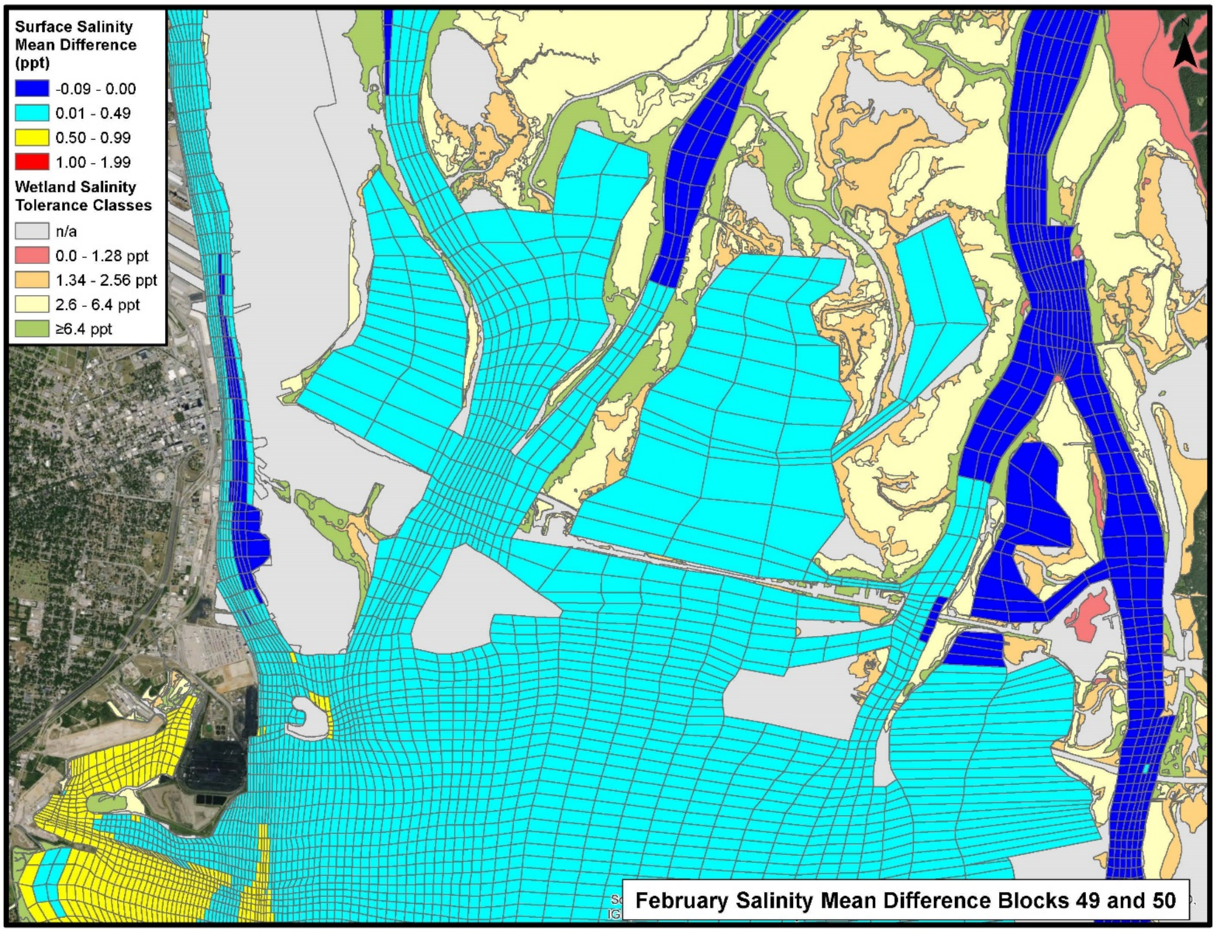


Figure 3.45. Estimated increase in salinity during the winter period (February data shown for example) within the central (transitional) portion of the study area. Note that estimated salinity increases are limited to 0.0, <0.5, or <1.0 ppt. In areas where salinity increases may occur, wetland communities are adapted to predicted conditions. Map units designated n/a include upland habitats, highly disturbed and developed areas (e.g., historic fill, roads), and open water areas not addressed in the model domain.

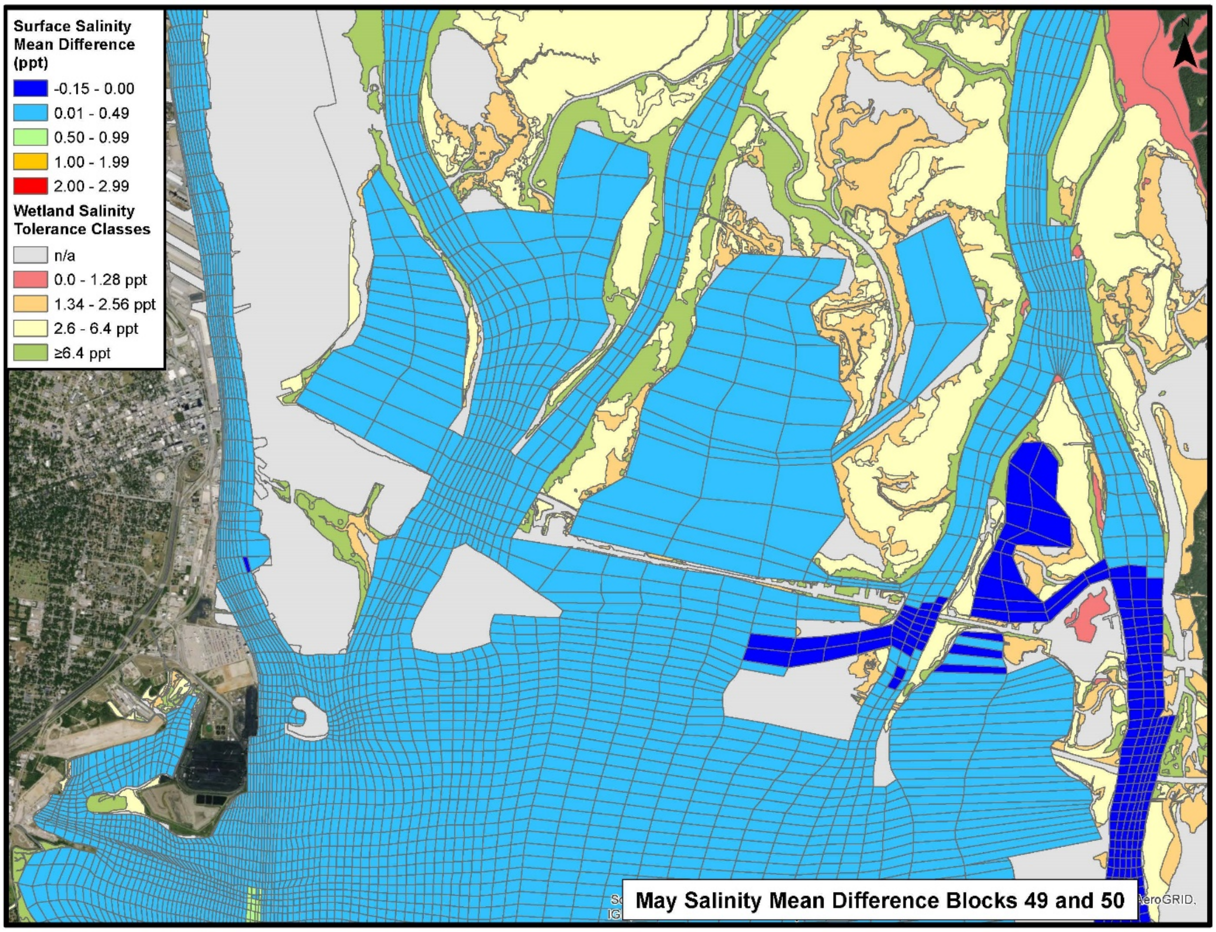


Figure 3.46. Estimated increase in salinity during the spring period (May data shown for example) within the central (transitional) portion of the study area. Note that estimated salinity increases are limited to 0.0, or <0.5 ppt. In areas where salinity increases may occur, wetland communities are adapted to predicted conditions. Map units designated n/a include upland habitats, highly disturbed and developed areas (e.g., historic fill, roads), and open water areas not addressed in the model domain.

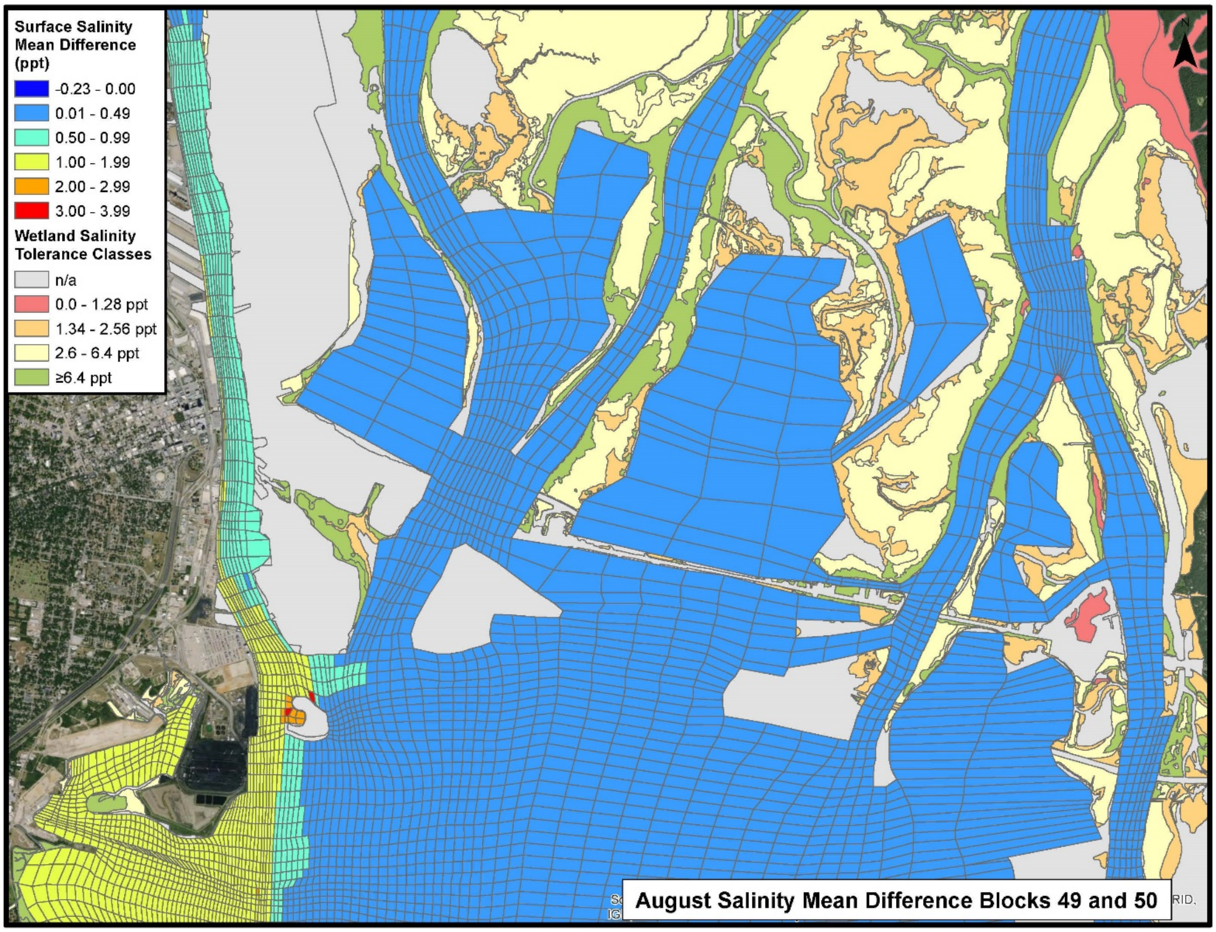


Figure 3.47. Estimated increase in salinity during the summer period (August data shown for example) within the central (transitional) portion of the study area. Note that in areas containing wetlands estimated salinity increases are limited to 0.0, <0.5, or <1.0 ppt. In areas where increases may occur, wetland communities are adapted to predicted conditions. Map units designated n/a include upland habitats, highly disturbed and developed areas (e.g., historic fill, roads), and open water areas not addressed in the model domain.

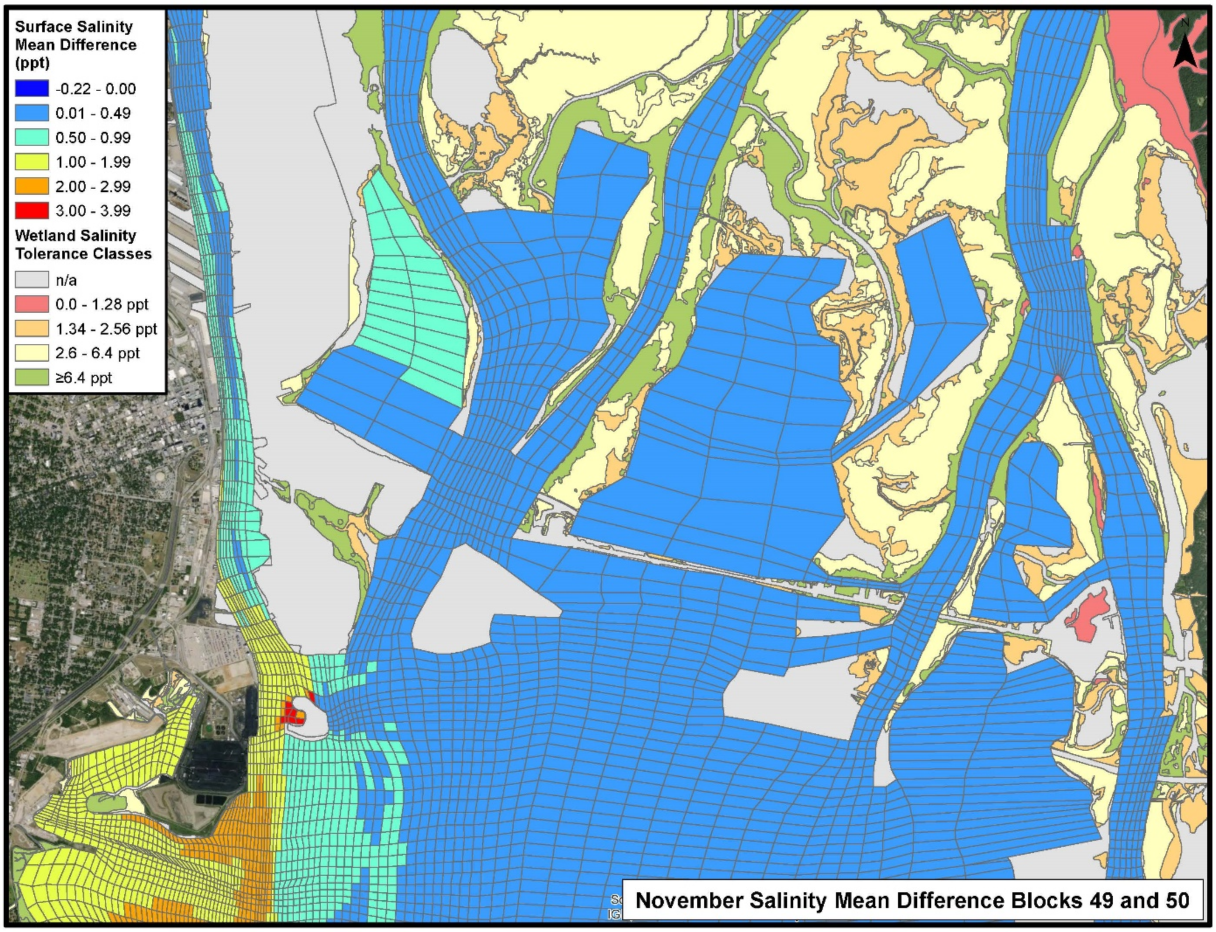


Figure 3.48. Estimated increase in salinity during the fall period (November data shown for example). Note that in areas containing wetlands estimated salinity increases are limited to 0.0, <0.5, or <1.0 ppt. In areas where increases may occur, wetland communities are adapted to predicted conditions. Higher increases in salinity (e.g., >2 ppt) may occur adjacent to the navigation channel, but no wetlands are located in those areas (bottom left). Map units designated n/a include upland habitats, highly disturbed and developed areas (e.g., historic fill, roads), and open water areas not addressed in the model domain.

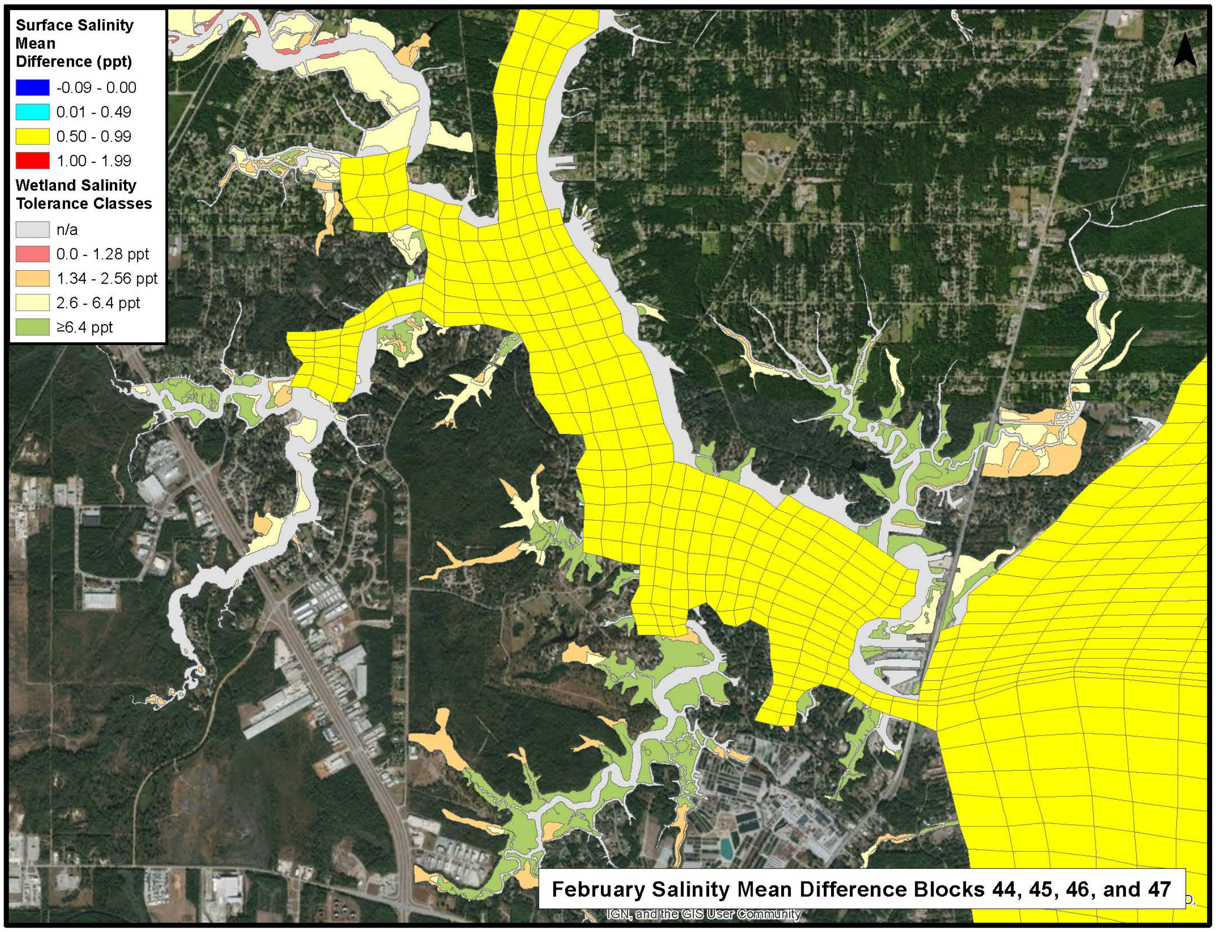


Figure 3.49. Estimated increase in salinity during the winter period (February data shown for example) within the lower (estuarine) portion of the study area. Note that in areas containing wetlands estimated salinity increases are limited to <1.0 ppt. In areas where increases may occur, wetland communities are adapted to predicted conditions. Map units designated n/a include upland habitats, highly disturbed and developed areas (e.g., historic fill, roads), and open water areas not addressed in the model domain.

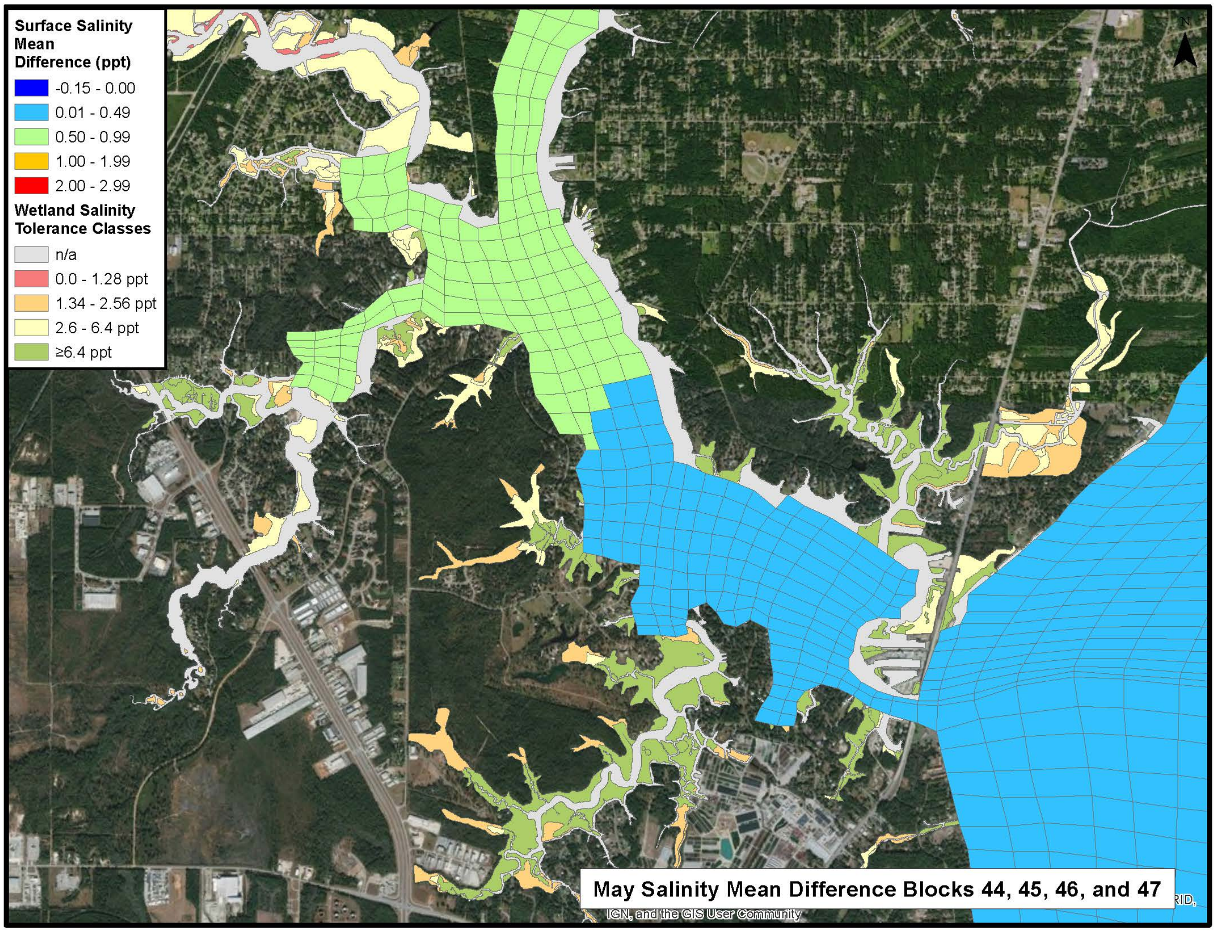


Figure 3.50. Estimated increase in salinity during the spring period (May data shown for example) within the lower (estuarine) portion of the study area. Note that in areas containing wetlands estimated salinity increases are limited to <0.5 or <1.0 ppt. In areas where increases may occur, wetland communities are adapted to predicted conditions. Map units designated n/a include upland habitats, highly disturbed and developed areas (e.g., historic fill, roads), and open water areas not addressed in the model domain.

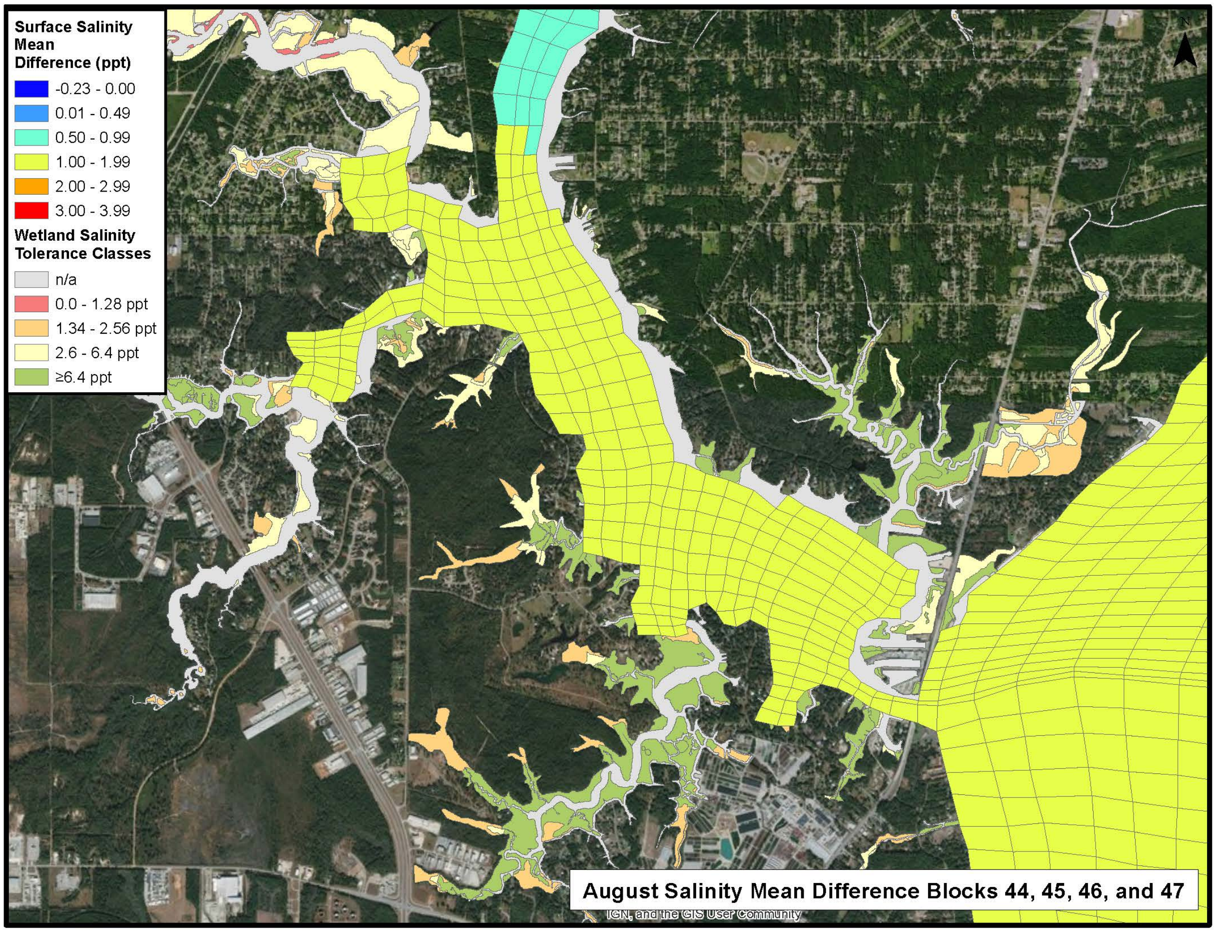


Figure 3.51. Estimated increase in salinity during the summer period (August data shown for example) within the lower (estuarine) portion of the study area. Note that in areas containing wetlands estimated salinity increases are limited to <1.0 or <2.0 ppt. In areas where increase may occur, wetland communities are adapted to predicted conditions. Map units designated n/a include upland habitats, highly disturbed and developed areas (e.g., historic fill, roads), and open water areas not addressed in the model domain.

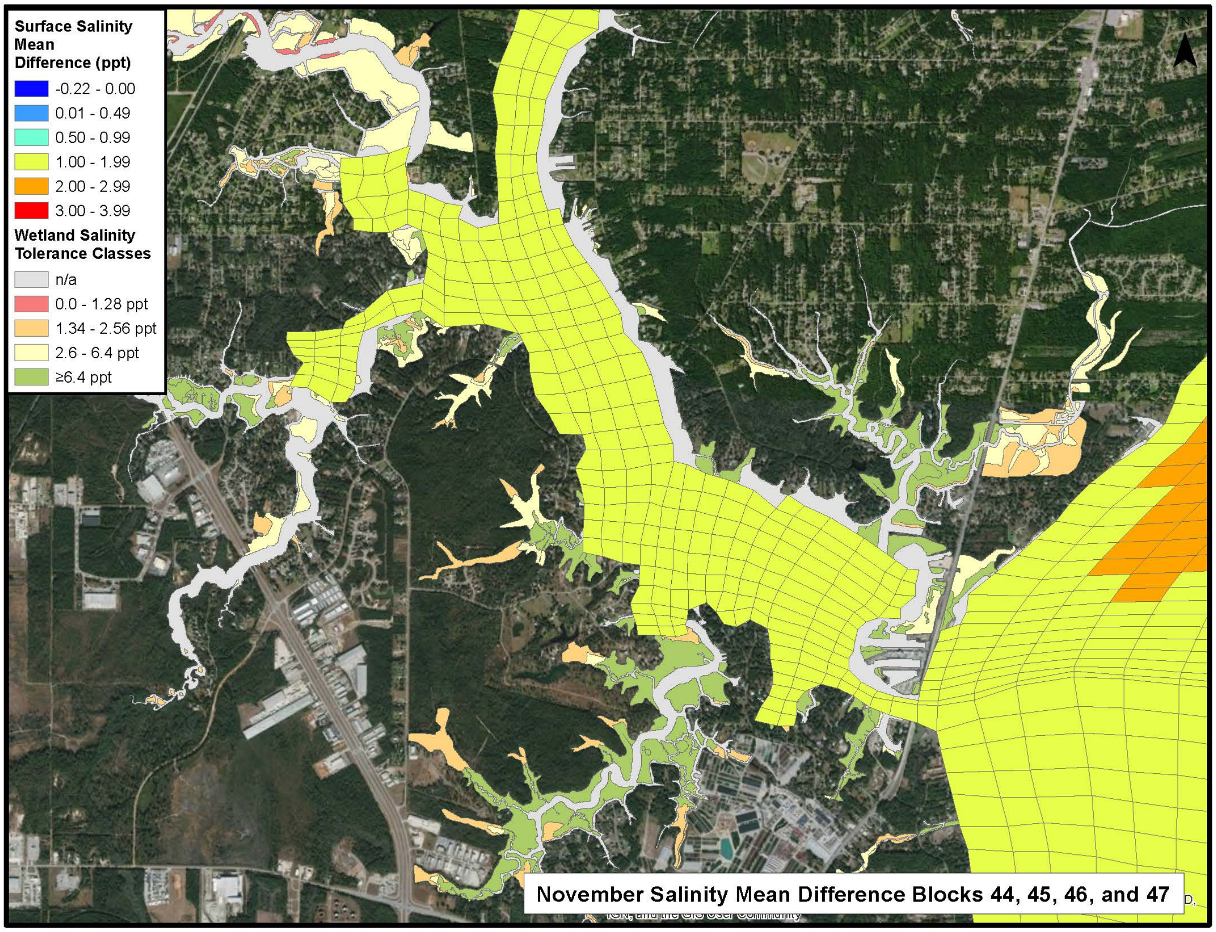


Figure 3.52. Estimated increase in salinity during the fall period (November data shown for example) within the lower (estuarine) portion of the study area. Note that in areas containing wetlands estimated salinity increases are limited to <1.0 ppt. In areas where increase occur, wetland communities are adapted to predicted conditions. Higher increases in salinity (e.g., <3.0 ppt) may occur adjacent to the navigation channel, but no wetlands are located in those areas (center right). Map units designated n/a include upland habitats, highly disturbed and developed areas (e.g., historic fill, roads), and open water areas not addressed in the model domain.

Sea level rise: The selected 0.5 m sea level rise scenario was assessed using a different approach than the one outlined above for wetland community mortality and productivity. Changes in salinity and other water quality parameters are expected to impact wetland assemblages and distributions as sea level rise occurs (Kirwan and Megonigal, 2013). However, in many regions the predominant impact of long term seal level rise will be excessive inundation leading to a conversion of wetland features to open water areas, especially in landscapes where landward retreat is restricted (USGS, others). As a result, the wetland assessment conducted as part of the proposed navigation channel expansion focuses on increased inundation, with an emphasis on determining wetland features that would become submerged following the 0.5 meter sea level rise scenario. The analysis focused on inundation because wetlands in the region currently experience prolonged periods of soil saturation, but extensive inundation can decrease productivity and prevent establishment of new plants (e.g., Baldcypress). To conduct the analysis, the water elevation provided in hydrodynamic models was appended to the wetland mapping and classification attribute table for each wetland feature. The projected elevation change in the nearest model cell was compared with the current elevation of each wetland feature. Features were considered impacted (i.e., inundated) when the projected elevation differences exceeded the current wetland feature elevation.

Results suggest that as many as 930 wetland features may be inundated as a result of the 0.5 m sea level rise projection, representing an area of 8440 acres. This includes forested areas predominantly dominated by freshwater communities (e.g., bottomland hardwoods), salt-tolerant halophytic communities (e.g., black needle rush, big cordgrass), and transitional communities (e.g., tidal shrub mix, Typha). Incorporating post project conditions into the assessment, a potential exists for inundation of four additional wetland features occupying an area of 10 acres. Notably, the inundation assessment does not account for the potential landward migration of wetlands into adjacent areas which may offset sea level rise impacts. Additionally, increased inundation may not result in the loss of wetlands but may lead to a shift of wetland types. For example, seasonally inundated wetlands may convert to more permanently saturated conditions. These changes have the potential to alter both species composition and structure, occurring over multi-years to multi-decadal timescales. As a result, predicting the end-state conditions and isolating impacts resulting from the proposed navigation project remains challenging. Given the limited estimated extent of potential project-induced impacts (10 acres) in the context of much larger potential sea level rise implications (>8000 acres) occurring over a 50 year interval suggests that any wetland impacts related to implementation of the project remain negligible within the larger sea level rise context. Additional research into sea level rise implications for wetlands in the region are needed to further account for future conditions, but remains beyond the scope of the current assessment which focuses on the proposed navigation channel expansion only.

3.4 References

Berkowitz, J. F., J. Pietroski, and D. Krenz. 2016. Identifying areas of potential wetland hydrology in irrigated croplands using aerial image interpretation and analysis of rainfall normality. ERDC/EL TR-16-6. Vicksburg, MS: U.S. Army Engineer Research and Development Center.

Berkowitz JF, Currie SJ, Pietroski JP. 2017. Evaluation of wetland hydrology in formerly irrigated areas. ERDC/EL TR-17-13. Vicksburg, MS: U.S. Army Engineer Research and Development Center.

Crain, C.M., Silliman, B.R., Bertness, S.L. and Bertness, M.D., 2004. Physical and biotic drivers of plant distribution across estuarine salinity gradients. Ecology, 85(9), pp.2539-2549.

Huckle, J.M., Potter, J.A. & Marrs, R.H. (2000) Influence of environmental factors on the growth and interactions between salt marsh plants: effects of salinity, sediment and waterlogging. Journal of Ecology, 88, 492–505.

Boesch, D.F., Josselyn, M.N., Mehta, A.J., Morris, J.T., Nuttle, W.K., Simenstad, C.A. and Swift, D.J., 1994. Scientific assessment of coastal wetland loss, restoration and management in Louisiana. Journal of Coastal Research, pp.i-103.

Brock, M.A., Nielsen, D.L. and Crossle, K., 2005. Changes in biotic communities developing from freshwater wetland sediments under experimental salinity and water regimes. Freshwater Biology, 50(8), pp.1376-1390.

Alabama Wildlife Federation (AWF). 2018. Mobile Bay: Wetland and Habitat. https://www.alabamawildlife.org/wetland-and-habitat/.

Grieve, C.M., S.R. Grattan and E.V. Maas. 2012. Plant salt tolerance. In: W.W. Wallender and K.K. Tanji (eds.) ASCE Manual and Reports on Engineering Practice No. 71 Agricultural Salinity Assessment and Management (2nd Edition). ASCE, Reston, VA. Chapter 13 pp:405-459.

Stout, J.P., Heck Jr, K.L., Valentine, J.F., Dunn, S.J. and Spitzer, P.M., 1998. Preliminary characterization of habitat loss: Mobile Bay national estuary program. MESC Contribution, (301), pp.17-40.

Kirwan ML. and P. Megonigal. 2013. Tidal wetlalnd stability in the face of human impacts and sea level rise. Nature. 504:53.

Gastaldo, R.A., 1989. Preliminary observations on phytotaphonomic assemblages in a subtropical/temperate Holocene bayhead delta: Mobile Delta, Gulf Coastal Plain, Alabama. Review of Palaeobotany and Palynology, 58(1), pp.61-83.

U.S. Army Corps of Engineers. 2010. Regional Supplement to the Corps of Engineers Wetland Delineation Manual: Atlantic and Gulf Coastal Plain Region (Version 2.0), ed. J. S. Wakeley, R. W. Lichvar, and C. V. Noble. ERDC/EL TR-10-20. Vicksburg, MS: U.S. Army Engineer Research and Development Center.

Hester, M. W., E. A. Spalding, and D. D. Franze. 2005. Biological resources of the Louisiana coast: Part 1. An overview of coastal plant communities of the Louisiana Gulf shoreline. Journal of Coastal Research Special Issue No. 44:134-145.

Glenn, E., Thompson, T.L., Frye, R., Riley, J. and Baumgartner, D., 1995. Effects of salinity on growth and evapotranspiration of Typha domingensis Pers. Aquatic Botany, 52(1-2), pp.75-91.

Downton, W.J.S. and Läuchli, A., 1984. Salt tolerance of food crops: prospectives for improvements. Critical reviews in plant sciences, 1(3), pp.183-201.

Natural Resources Conservation Service (NRCS). 2006. Land resource regions and major land resource areas of the United States, the Caribbean, and the Pacific Basin. Agric. Handbook. 296. U.S. Gov. Print. Office, Washington, DC.

Vepraskas M., Craft, C. 2016. Wetland Soils: Genesis, Hydrology, Landscapes, and Classification. CRC Press. Boca Raton, FL.

Gough, L., Grace, J.B. and Taylor, K.L., 1994. The relationship between species richness and community biomass: the importance of environmental variables. Oikos, pp.271-279.

Kozlowski, T.T., 1997. Responses of woody plants to flooding and salinity. Tree physiology, 17(7), p.490.

Cerco, C.F. and Cole, T., 1995. User's Guide to the CE-QUAL-ICM Three-dimensional Eutrophication Model: Release Version 1.0. US Army Engineer Waterways Experiment Station.

Tiner R. 2016. Wetland Indicators: A Guide to Wetland Identification, Delineation, Classification, and Mapping. CRC Press, Boca Raton, FL.

Reddy R, DeLaune R. 2008. Biogeochemistry of Wetlands. CRC Press. Boca Raton, FL.

Oliver CD, Larson BC. 1996. Forest Stand Dynamics. John Wiley & Sons, Inc. New York, NY.

Hellings, S.E. and Gallagher, J.L., 1992. The effects of salinity and flooding on Phragmites australis. Journal of Applied Ecology, pp.41-49.

Chabreck, R.H., 1989. Creation, restoration and enhancement of marshes of the northcentral Gulf coast. Wetland Creation and Restoration: The Status of the Science. US Environmental Protection Agency, Environmental Research Laboratory, Corvallis, OR, USA, pp.127-144.

Alabama Natural Heritage Program. 2012. Alabama Inventory List: the rare, threatened and endangered plants & animals of Alabama. Auburn University, Alabama.

Holm, L.G., D.L. Plucknett, J.V. Pancho, and J.P. Herberger. 1977. The world's worst weeds: distribution and biology. Honolulu: University Press of Hawaii.

Joyce, L.A. and Baker, R.L., 1987. Forest overstory-understory relationships in Alabama forests. Forest Ecology and Management. 18 (1): 49-59., 18(1), pp.49-59.

Laderman, A.D. 1989. The ecology of Atlantic white cedar wetlands: a community profile. United States Fish and Wildlife Service, National Wetlands Research Center. Report Bulletin 85(7.21).

Lambertini, C., I.A. Mendelssohn, M.H. Gustafsson, B. Olesen, T. Riis, B.K. Sorrell, and H. Brix. 2012. Tracing the origin of gulf coast *Phragmites* (Poaceae): a story of long-distance dispersal and hybridization. American Journal of Botany 99(3): 538–551.

Noss, R.F. 2013. Forgotten grasslands of the south: natural history and conservation. Island Press. Washington, D.C.

Conner, William H.; Askew, George R. 1993. Impact of saltwater flooding on red maple, redbay and Chinese tallow seedlings. Castanea. 58(3): 214-219.

Conner, W.H., McLeod, K.W. and McCarron, J.K., 1997. Flooding and salinity effects on growth and survival of four common forested wetland species. Wetlands Ecology and Management, 5(2), pp.99-109.

Sande, E. and Young, D.R., 1992. Effect of sodium chloride on growth and nitrogenase activity in seedlings of Myrica cerifera L. New phytologist, 120(3), pp.345-350.

USDA 2000. The PLANTS database. http://plants.usda.gov. National Plant Data Center, Baton Rouge, Louisiana.

Schultz, R. & E. Dibble. 2012. Effects of invasive macrophytes on freshwater fish and macroinvertebrate communities: the role of invasive plant traits. Hydrobiologia 684(1):1-14.

O'Neil PE and MF Mettee. 1982. ALABAMA COASTAL REGION ECOLOGICAL CHARACTERIZATION VOLUME 2 A SYNTHESIS OF ENVIRONMENTAL DATA. FWS/OBS-82/42.

Tiner, R.W. 1993. Field guide to coastal wetland plants of the southeastern Unites States. University of Massachusetts Press.

USFWS – National Wetland Inventory Mapper. 2016. https://www.fws.gov/wetlands/Data/Mapper.html

Ward, D.B. 2010. North America has two species of *Phragmites* (Gramineae). Castanea. 75(3):394-401.

Weakley, A.S. 2015. Flora of the southern and mid-Atlantic states. Working draft of 29 May 2015. University of North Carolina Herbarium, Chapel Hill, North Carolina.

Munns, R. and Tester, M., 2008. Mechanisms of salinity tolerance. Annu. Rev. Plant Biol., 59, pp.651-681.

Chapter 4: Submerged Aquatic Vegetation

Summary

This Chapter describes the potential impacts of the proposed channel deepening and widening of the Mobile Bay Federal Navigation Channel on the Submerged Aquatic Vegetation (SAV) within the Mobile Bay system as a consequence of project related salinity changes. We used field verified SAV distribution maps to determine seasonal species distribution and determined species specific salinity thresholds through literature reviews. Using hydrodynamic model predictions of salinity change due to project implementation, we were able to assess increases in salinity above relative SAV salinity threshold ranges. We focused the analysis on the estuarine transition zone, and determined that the largest increase in salinity was 1.5 ppt above species-specific salinity threshold values. Four species of SAV, Eurasian Watermilfoil, Wild Celery, Southern Naiad, and Widgeon Grass, were predicted to experience an increase in salinity up to 1.5ppt above threshold values due to proposed project implementation. None of these increases are expected to significantly impact SAV habitat. No impact due to dissolved oxygen changes

resulting from the project are expected. Predicted salinity impacts of sea level rise (SLR) are greater than those predicted under project implementation.

4.1 Introduction

General context: Submerged Aquatic Vegetation (SAV) refers to a subset of vascular plants that have adapted to live underwater, in marine, estuarine and freshwater conditions. Healthy SAV beds are important habitats that are beneficial in many ways. By buffering wave energy, modifying wave currents, preventing erosion, consolidating sediment and influencing deposition, SAV can help to maintain and shape coastal landscapes (Biber and Cho 2017). In addition, coastal seagrass beds represent one of the most productive ecosystems on the planet and provide food, shelter and nesting grounds to many commercially and ecologically important invertebrate and vertebrate communities.

SAV diversity and distribution are limited by a number of water quality parameters. Light attenuation and water clarity, as measured through Photosynthetically Active Radiation (PAR) and Turbidity, are critical as these are vascular plants that require light. In addition to light, predominant limiting factors to SAV distribution and diversity are salinity and temperature. For this impact assessment, the parameters that were available for evaluation of impacts from the accompanying hydrodynamic and water quality models (described in detail in supplemental report and in sections below addressing assessment of model results) were salinity and dissolved oxygen.

Problem statement: The proposed channel deepening and widening of the Mobile Bay Federal Navigation Channel may cause changes in the salinity regime within the Mobile Bay system. If there is an influx in saltwater into upstream habitats, increased salinities may have impacts on SAV communities, depending on where the salinity changes occur (geographic location), how long they last (duration) and how these changes align spatially with existing SAV habitat.

Model Purpose: The current chapter focuses on groundtruthing and utilizing baseline maps of SAV habitat within the system, identifying variation in SAV distribution across several years and seasons, and assessing potential species specific impacts of increased salinity resulting from hydrodynamic and water quality models of the proposed widening and deepening of the Mobile Bay Federal Navigation Channel.

Model summary: Baseline data, leveraged from existing maps of SAV distribution initially developed in conjunction with the Mobile Bay National Estuary Program (MBNEP) and Alabama Department of Conservation and Natural Resources State Lands Division (SLD), were field verified to check accuracy and temporal variation in order to establish baseline distribution, within Mobile Bay. Salinity tolerance thresholds were identified for local SAV species through a review of published literature to use to determine impacts of potential salinity change due to project implementation. Following establishment of salinity thresholds and ranges, we used the output of the hydrodynamic and water quality model results to 1) estimate salinity values for SAV polygons within the estuarine transition zone but outside of model domain, 2) assess change in depth averaged mean and 75th percentile salinity monthly during 2015 due to project implementation (with/without project salinity), and 3) identify SAV patches that would be impacted with above threshold salinity values due to project implementation. We focused on the estuarine transition zone because this is where larger changes in salinity are expected if changes are to occur as a result of project implementation. The impact of salinity changes with and without project under a sea level rise scenario were also assessed. Finally we looked at predicted changes in dissolved oxygen (DO) as a result of the project and assessed the potential impacts due to DO.

4.2 Methods - Model Development Process

Study Area

To assess potential impacts of the Mobile Harbor Channel Deepening on SAV coverage and distribution, we used SAV survey maps developed by the environmental and research consulting group Barry A. Vittor and Associates, Inc (Vittor). These surveys were supported by the MBNEP and Alabama Department of Conservation and Natural Resources SLD. The surveys focused on near-shore estuarine and marine aquatic ecosystems in coastal Alabama including the entire coastline (Vittor, 2004, Figure 4.1). The northern boundary of these surveys was the Louisville and Nashville (L & N) Railroad north of Mobile Bay, with the exception of the streams and bays of the waterway north of the L&N Railroad (i.e., McReynolds Lake/The Basin).

Existing SAV surveys

SAV surveys of Mobile Bay have been completed by the environmental and research consulting group Barry A. Vittor and Associates Inc. for several years to support the Mobile Bay National Estuary Program and the Alabama Department of Conservation and Natural Resources. These SAV surveys used a combination of aerial imagery mapping and field verification. As described in their reports, Vittor used the following methodology:

> " Ortho imagery was created from true color aerial photography acquired with a digital mapping camera. The orthorectification process relied on the aerial imagery, camera calibration data, aerotriangulation data, and a digital elevation model. The procedure was performed in a fully digital workflow environment, using measurements obtained from airborne global positioning system and an inertial measurement unit to provide accurate exterior orientation of the imagery. Outlines of SAV signatures in the ortho imagery were digitized in a GIS environment, using the seasonal mosaics as base maps. Digitized areas were field verified to document habitat characteristics at the surface level." (Vittor et al. 2004)

Through the on the ground field surveys, Vittor identified species composition of the SAV beds. Surveys were conducted in 2002, 2009, and the summer (July/August) and fall (October) of 2015 (Vittor and Associates, Inc. 2004, 2010, 2016). To our knowledge, the Vittor surveys provide the best available SAV mapping data for the Mobile Bay region and we focused on their mapping efforts from the fall of 2015 to address potential impacts to SAV species as a result of the proposed channel deepening (Figures 4.2 [entire study area], 4.3 [estuarine transition zone], and 4.4). We used maps developed in other seasons and years to assess natural variation in species distribution (aerial coverage and composition).

Field verification and assessment of variation

For additional QA/QC of the baseline maps developed by Vittor, ERDC ran a hydroacoustic survey in October of 2016 to groundtruth and compare to the 2015 Vittor et al survey. ERDC's SAV hydroacoustic survey utilized the Submersed Aquatic Vegetation Early Warning System

(SAVEWS Jr.) which incorporated a boat mounted Humminbird high-frequency sonar that can detect SAV in high turbidity water and is integrated with a GPS system (Sabol et al. 2014). The transducer is synced with a GPS enabling estimation of the edges of SAV beds within 1 m resolution. Variation in SAV coverage by year was examined by comparing mapped SAV polygon size using ArcGIS 10.3.1.

Salinitiy tolerance estimates

Salinity tolerances of SAV were estimated using a literature review of published salinity thresholds for local SAV species. In cases in which salinity threshold data were not available, reports of species distribution coupled with known salinity conditions were used to estimate the salinity range. Salinity range refers to the expected salinity conditions a species is exposed to within a given location, whereas salinity threshold tolerance refers to the lowest and highest salinity values a species can withstand. For most species, even when a salinity threshold has been identified, the impact of duration or length of time of exposure to that threshold value is not known. Where more than one tolerance threshold was published, we used both the report with the closest geographic proximity (i.e., nearest study sites to Mobile Bay) and the lowest reported maximum threshold value in an effort to provide conservative estimates of tolerance.

When we intersected the Vittor fall 2015 SAV coverage map with the modeled baseline salinity data, we found that a number of species were persistent in areas with modelled salinity above reported threshold values. To adjust to modeled salinity output, we estimated relative tolerance thresholds for Mobile Bay SAV. To do so, we intersected SAV survey maps from the fall 2015 Vittor aerial survey with seasonal (Fall: October, Winter: February, Spring: May, Summer: August) baseline model mean, depth averaged salinity data using ArcGIS 10.3.1. Although we present results from all seasons, we focused on the Fall (October data) because it has the highest salinity values, and represents the month in which plants are exposed to the most saline conditions in the year. Salinity values predicted from the hydrodynamic model that were higher than published maximum threshold values were assigned as relative maximum threshold values. Any predicted increase in salinity above this relative maximum threshold as a result of project implementation was considered a salinity value above the species specific relative maximum. SAV salinity tolerance estimates were only taken where the water quality model overlapped the SAV beds, not where we estimated salinity values for SAV beds (i.e., not in unmodeled beds). Relative maximum salinity threshold values are species specific and were applied to the entire survey area (beds that were within and outside of the model domain).

Assessing impact of hydrodynamic and water quality modelling results

Hydrodynamic and water quality data were modelled for Mobile Bay, estimating baseline (i.e., existing, without project) conditions as well as conditions post-project implementation using the Geophysical Scale Multi-Block (GSMB) system, the Curvilinear Hydrodynamic in three-dimension Waterways Experiment Station (CH3D-WES) approach, and the CE-QUAL-ICM water quality component developed and maintained by the US Army Corps of Engineers Engineer Research and Development Center (Cerco and Cole 1995), as described in chapters that supplement the current one. The hydrodynamic and water quality models were used to predict baseline conditions, conditions following project implementation, and baseline and project conditions under a 0.5m sea level rise projection scenario. The 0.5m sea level rise projection is considered the intermediate projection for the Mobile Bay area. Specifically, the monthly depth averaged mean salinity value was calculated for each individual model cell, under baseline and post project conditions and with and without sea level rise. Because the depth in which SAV occur is so shallow, we used the depth averaged model outputs for parameters of interest as it was most relevant to what the entire plant (roots to shoots) would experience (as opposed to the top or bottom three depth layers). To estimate the changes due to project implementation, baseline salinity values were subtracted from post-project salinity values. This process was completed on a cell by cell basis, so that salinity change could be determined for the entire model domain. Once predicted salinity change was estimated for the whole model domain, we intersected the mapped SAV beds within the domain using ArcGIS software to isolate salinity output to regions where SAV were present. We then compared the change in mean, depth averaged salinity from baseline to project as predicted by the hydrodynamic model to the relative salinity threshold values established for local SAV species an reported any predicted increases. In cases in which an SAV bed contained multiply species, we used the salinity tolerance of the species most intolerant of increased salinity (i.e., the species with the lowest salinity tolerance values) to evaluate impacts. In addition to the mean monthly salinity values, we also investigated the 7^{5th} percentile hydrodynamic model outputs for salinity, following the same methodology. We included an analysis of the 75th percentile to provide an indication and assessment of the variation in modelled salinity that were similar, but slightly more conservative than a standard deviation approach (i.e. reporting 1 standard deviation from mean measurements). The 75th percentile results provide an indication of the variation around mean values, and highlight that in this case, variation from mean estimates are small. Note that higher salinity values predicted using the 75th percentile have very short durations and small geospatial footprints. We used the same approach in determining the potential impacts of salinity change due to project implementation in combination with 0.5m modelled Sea Level Rise scenario. In addition to salinity, we also assessed DO outputs from the Water Quality model to determine whether we could predict any impact of decreased DO on submerged plants from baseline to post project conditions.

Assigning water quality to SAV beds outside of model domain

SAV beds in the Mobile Bay delta tend to be in relatively shallow water (<1m). In some cases, the hydrodynamic and water quality model domains did not overlap with shallow regions that contained SAV. Of the 6300 acres of SAV beds in the 2015 fall surveys, 2376 acres did not have overlapping water quality data from either model (Figure 4.5). In order to assign estimated water quality parameters values to the 2376 "unmodeled" acres of SAV, the mean water quality value of interest of all adjacent model polygons touching the unmodeled SAV bed was assigned to that unmodeled bed (Figure 4.6). In cases in which there were no adjacent model water quality polygons (e.g. SAV beds were far up a creek), we 1) measured the distance from the mouth of the creek to the SAV beds, 2) applied that distance in an upstream direction in the nearest adjacent polygons that were within the model domains, and 3) assigned the value obtained at the distance and location identified in step 2 to the unmodeled SAV beds in question (Figure 4.6B). This approach likely overestimates some salinity values that will reach distant SAV beds. This, in effect, makes the interpretation of project impacts more conservative.

4.3 Results – Application

Field verification and assessment of variation

The SAVEWs survey covered a distance of 64 km throughout the Mobile Bay, with the goal of mapping the edges of various SAV beds to compare to beds recently mapped by Vittor (Figure 4.7, 4.4). A total of 31,684 points were mapped and 1788 of these points (~0.06%) detected the presence of SAV. Because of variance in SAV coverage seasonally and annually, we compared our October 2016 hydroacoustic survey against the fall 2015 shape file data supplied by Vittor. Of the 1788 points, the hydroacoustic survey detected SAV about 85% overlapped with the SAV polygons mapped by Vittor (Figure 4.8). The remaining 15% of hydroacoustic SAV detections were within 10 meters of the Vittor SAV polygons. The 15% difference can likely be attributed to annual variation. The hydroacoustic survey could only determine absence or presence of SAV and not species composition. During the hydroacoustic survey, a rake was used to collect SAV for species identification and the GPS position was recorded for every rake sample. The species identification for each rake sample location had 100% agreement with the Vittor fall 2015 survey. The agreement of the two techniques shows the SAV coverage of Mobile Bay is accurately portrayed in the Vittor fall 2015 survey and is suitable for the use of potential impacts that the Mobile Bay deepening project may have on SAV. Another benefit to using the fall 2015 SAV aerial survey is that the salinity results from the hydrodynamic and water quality models estimate the greatest salinity differences between the no project and project salinity values in Mobile Bay to occur in October. The model also estimates that salinities are naturally highest during October so this is when plants will be most susceptible to salinity stress.

Year to year and seasonal variation in SAV coverage by year is both common and extensive (Table 4.1). The species with both the most coverage and the most temporal variation in coverage were Eurasian Watermilfoil *(Myriophyllum spicatum*), Water Celery (*Vallisneria neotropicalis*), Southern Naiad *(Najas guadalupensis*), Water stargrass (*Heteranthera dubia*), and Coons Tail (*Ceratophyllum demersum*). These species ranged in mean acreages of ~1600 to 4000 with high variance (standard deviation ranged from ~1300-2000 acres). In comparison, on average, the rest of the common species covered less than 1000 acres each and all but Widgeon Grass (*Ruppia maritima*) covered less than 400 acres each.

Salinitiy tolerance estimates

Species specific salinity tolerance thresholds and range estimates, as compiled from published reports and peer reviewed literature is presented in Table 4.2. As is expected in a geographic region that encompasses fresh water, brackish, and estuarine conditions, the SAV species found in the region have tolerance ranges that vary considerable depending on whether the plant is adapted to variable salinity exposure or not. For example, Water Stargrass, *Heteranthera dubia*, is a predominantly freshwater species with a limited salinity tolerance of 0-3.5 ppt. In contrast, Shoal grass, *Halodule wrightii*, has a very broad salinity tolerance of 0-60+ ppt. These species specific differences provide critical information for evaluating potential impacts of increased salinity due to projects implementation. Spatial alignment of project related salinity increases with SAV species occurrence makes it possible to evaluate impacts. For example, an increase in salinity from 2ppt to 10ppt would not indicate potential impacts if this increase occurred in an SAV bed made up of Shoal grass. If the bed were composed of Water Stargrass, this same increase in salinity would likely have negative effects on the species.

Assessing impact of hydrodynamic and water quality modelling results

Salinity

Results of the hydrodynamic model indicate that predicted depth averaged salinity changes due to project implementation are less than 2 parts per thousand (ppt) during the months of January-June (Figure 4.9). There is an increased range in predicted depth averaged mean salinity starting in July, and peaking in October, with a range above 5 ppt (Figure 4.9). Summaries of the 75th percentile results show similar trends, with a larger range of increased predicted salinity in October and November (Figure 4.10). These results indicate the October is the most critical month to examine in terms of potential impact of salinity increases on SAV distribution and coverage. In fact, our analysis indicated that there are no increases in salinity above relative threshold values due to the proposed project in the Spring, Summer or Winter months (Figures

4.11, 4.12 and 4.13).Therefore, we focused our impact analysis on the month of October. In addition, we found that there were minimal changes that impacted salinity threshold values for SAV in the lower bay, and focused our results on the estuarine transition zone, where larger changes in salinity are expected (see mapped domain extent in figures).

When predicted increases in salinity above the species-specific SAV threshold values were evaluated, we found that the majority of SAV habitat was not predicted to experience an increased salinity regime or be impacted by salinity changes due to the channel deepening project (Figure 4.14). Eighty-three percent of the mapped fall 2015 SAV habitat is predicted to experience a negligible (≤0.5ppt) monthly mean change in salinity (Table 4.3). The range in mean salinity threshold increases were from 0-1.5ppt. Similar patterns were seen when evaluating the monthly 75th percentile hydrodynamic model output. In this case, post-project impacts were predicted to be ≤0.5ppt for 80.7% of all mapped SAV and increases in salinity thresholds were from 0-1.5ppt (Table 4.3). There was a total of 52 (mean) and 58 (75th percentile) acres of SAV habitat that showed predicted increases above 1 ppt in October salinity threshold values following project implementation (Table 4.3). Although there were cases in which the salinity increased up to 1.5ppt above relative salinity threshold values, these elevated salinities did not persist in time, with durations on the order of hours, as opposed to days or months.

In order to get a better understanding and evaluate these potential impacts further, we ran a species specific analysis for potentially impacted species with low salinity thresholds. These species include Water Star Grass, Eurasian Watermilfoil, Southern Naiad, Widgeon Grass, Wild Celery, Carolina Fanwort and Coon's Tail. Of these, only four species, Eurasian Watermilfoil, Wild Celery, Southern Naiad, and Widgeon Grass were predicted to experience an increase in salinity up to 1.5ppt above threshold values (Tables 4.4 & 4.5).

The majority of the potentially impacted SAV habitat is made up of Widgeon Grass, followed by Southern Naiad. Widgeon Grass can tolerate hypersaline conditions up to 100ppt, so an increase in salinity of 1.5ppt of up to 22 acres of Widgeon Grass does not a represent an impact to this species (Table 4.2 and references therein, Table 4.4). Southern Naiad has a salinity range up to 10ppt, with best growth occurring in a salinity range of 0-5 ppt and decreasing growth up to salinities of 10ppt (Moore 2012). However, mortality does not occur until plants experience an exposure duration of 10 ppt for a month or more (Moore 2012). Therefore, the duration of high salinities is critical. An increase of 1.5ppt above relative threshold values is unlikely to impact the 21 acres of Southern Naiad in question, unless these increased salinities have extended (i.e. multiple weeks) duration.

Two to twenty-six acres of Wild Celery were also predicted to experience elevated salinities 1-1.5ppt above threshold values (mean, 75th percentile, respectively) due to project implementation (Tables 4.4 & 4.5). At a maximum reported salinity threshold of 18 ppt (Table 4.2), post-project estimates suggest salinity exposure to increase to 20.5ppt. These results do not contain duration information, despite the importance of exposure time to elevated salinity. A short exposure (< 4hrs) to elevated salinity will likely have a smaller impact than a long (>24 or 48 hrs) exposure time. The extent of the impact is due to both magnitude of salinity increase, duration of exposure, and the specific species of interest. For many SAV species, duration data are not reported. Fortunately, studies have been conducted using Wild Celery, showing that this species can survive salinity up to 25ppt in pulses of less than 7 days (Frazer et al. 2006). As the predicted salinity impact due to project implementation are lower than this, we expect that the predicted salinity increases should have a minimal impact Wild Celery, if any.

Eurasian Watermilfoil, an aquatic invasive species native to Europe, Asia and North Africa. This species was introduced to the U.S. and first sighted in the early 1940s. It is now introduced nationwide. Eurasian Watermilfoil reproduces through fragmentation, grows quickly and outcompetes native species. Due to its invasive status, impacts to this species are unlikely to require mitigation or have a negative impact on local SAV species.

Sea Level Rise and Salinity

Results from the hydrodynamic model indicate that a 0.5m sea level rise projection will contribute to salinity changes in the Mobile Bay region. Changes from existing baseline condition to baseline conditions (i.e., no project) with sea level rise show an increase in relative salinity tolerance thresholds for mapped SAV species ranging from -1 to 3 ppt (Figure 4.15). This is a greater range of change seen post-project without sea lever rise conditions, and the distribution of change is different (Figures 4.15 & 4.16). A larger proportion of SAV habitat will be exposed to higher salinities due to sea lever rise impacts than project implementation impacts. To illustrate this point further, the increase in salinity above relative SAV salinity thresholds due to project implementation under a 0.5 sea level rise scenario shows the same range in salinity increases and distribution as those with sea level rise under baseline conditions (Figures 4.15 & 4.16).

Dissolved Oxygen

While low levels of dissolved oxygen (DO) in the water column can cause mortality of invertebrates and fish, and can have a devastating impact within a bay system, SAV, like all vascular plants, produce oxygen and some release oxygen from their roots under low oxygen conditions (Sand-Jensen et al, 1984). In order for DO conditions to create stressful condition for SAV, the conditions would need to be very low, persistent DO. As reported in other chapters, the lowest post-project DO levels predicted in the water quality model were minimum summer (June-September) DO concentrations ranging from 6.7-7.1 mg/L. These concentrations of DO would not have an impact on the SAV species present.

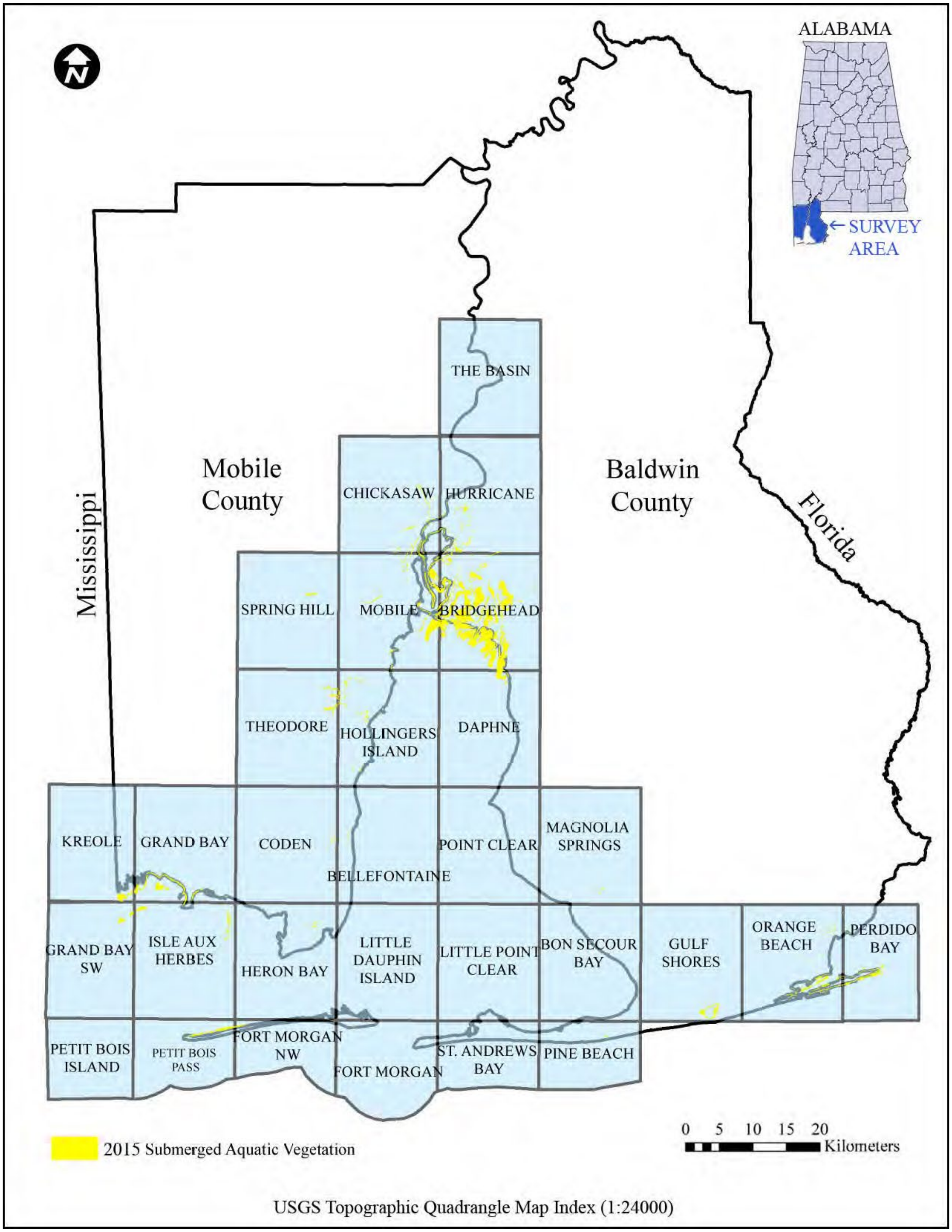


Figure 4.1. Map of surveyed region used to map SAV via remote sensing techniques. From B.A. Vittor and Associates, Inc. (2016).

Figure 4.2. Spatial Distribution of SAV beds (Fall 2015) within the entire study area using Vittor & Associates data.

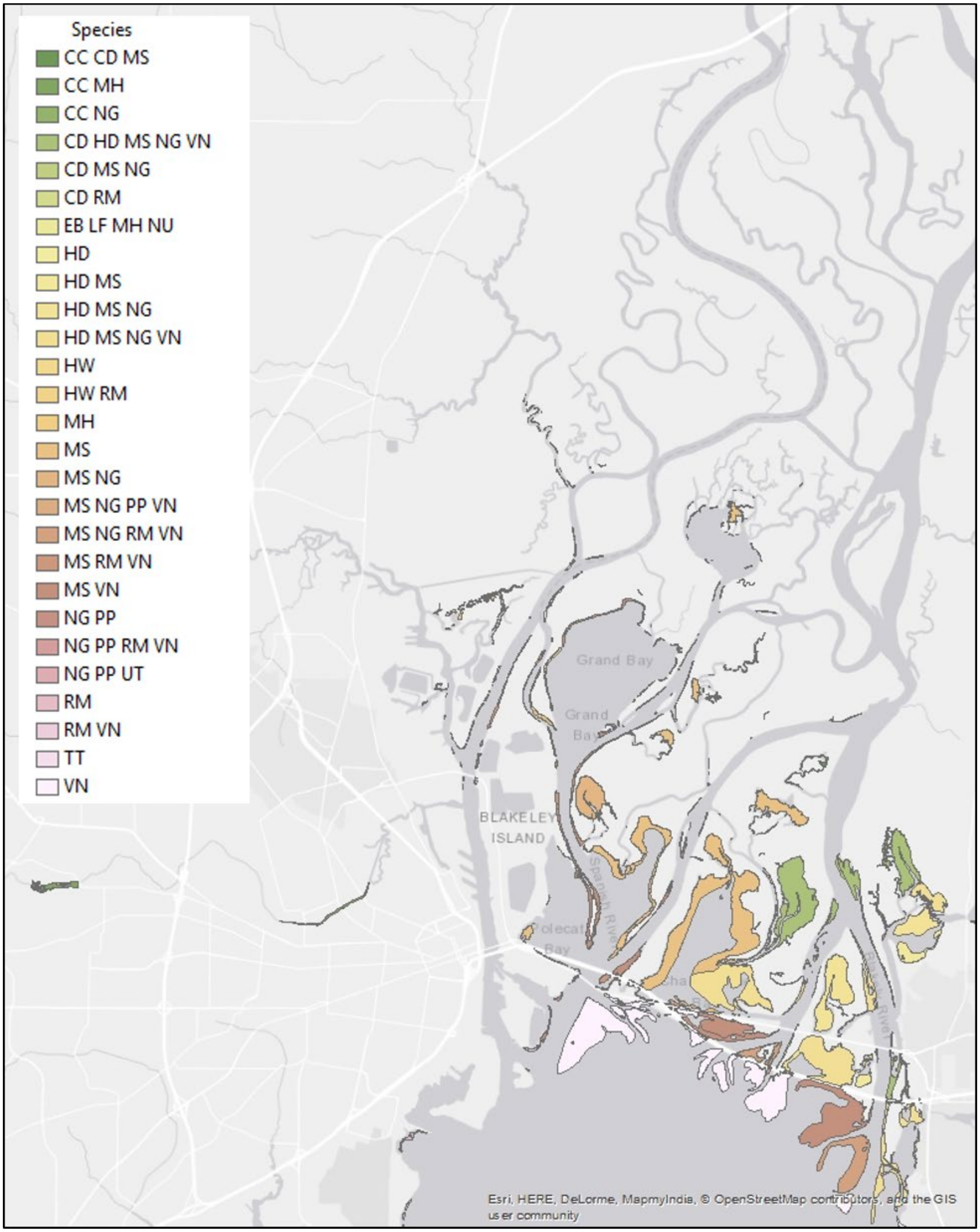


Figure 4.3 Fall 2015 SAV distribution within Mobile Bay as mapped by Vittor & Associates. Species codes can be found in Figure 4.4 and Table 4. 1.

Figures 4.3 and 4.6 Species legend

Species	
CC CD MS	*Cabomba caroliniana,Ceratophyllum demersum, Myriophyllum spicatum*
CC MH	*C. demersum, Myriophyllum heterophyllum*
CC NG	*C. demersum, Najas guadalupensis*
CD HD MS NG VN	*C. demersum, Heteranthera dubia,M. spicatum, N. guadalupensis, Valissneria neotropicalis*
CD MS NG	*C. demersum, M. spicatum, N. guadalupensis*
CD RM	*C. demersum, Ruppia maritima*
EB LF MH NU	*Eleocharis baldwinii, Luziola fluitans, M. heterophyllum, Nuphar ulvacea*
HD	*Heteranthera dubia*
HD MS	*H. dubia, M. spicatum*
HD MS NG	*H. dubia, M. spicatum, N. guadalupensis*
HD MS NG VN	*H. dubia, M. spicatum, N. guadalupensis, V. neotropicalis*
HW	*Halodule wrightii*
HW RM	*H. wrightii, R. maritima*
MH	*M. heterophyllum*
MS	*M. spicatum*
MS NG	*M. spicatum, N. guadalupensis.*
MS NG PP VN	*M. spicatum, N. guadalupensis, Potamogeton pusillis, V. neotropicalis*
MS NG RM VN	*M. spicatum, N. guadalupensis R. maritima, V. neotropicalis*
MS RM VN	*M. spicatum, R. maritima, V. neotropicalis*
MS VN	*M. spicatum, V. neotropicalis*
NG PP	*N. guadalupensis, P. pusillis*
NG PP RM VN	*N. guadalupensis, P. pusillis, R. maritima, V. neotropicalis*
NG PP UT	*N. guadalupensis, P. pusillis, Utricholaria inflata*
RM	*R. maritima*
RM VN	*R. maritima, V. neotropicalis*
TT	*Thalassia testinudum*
VN	*V. neotropicalis*

Figure 4.4. Species specific legend for SAV patches mapped in figures 4.3 and 4.6.

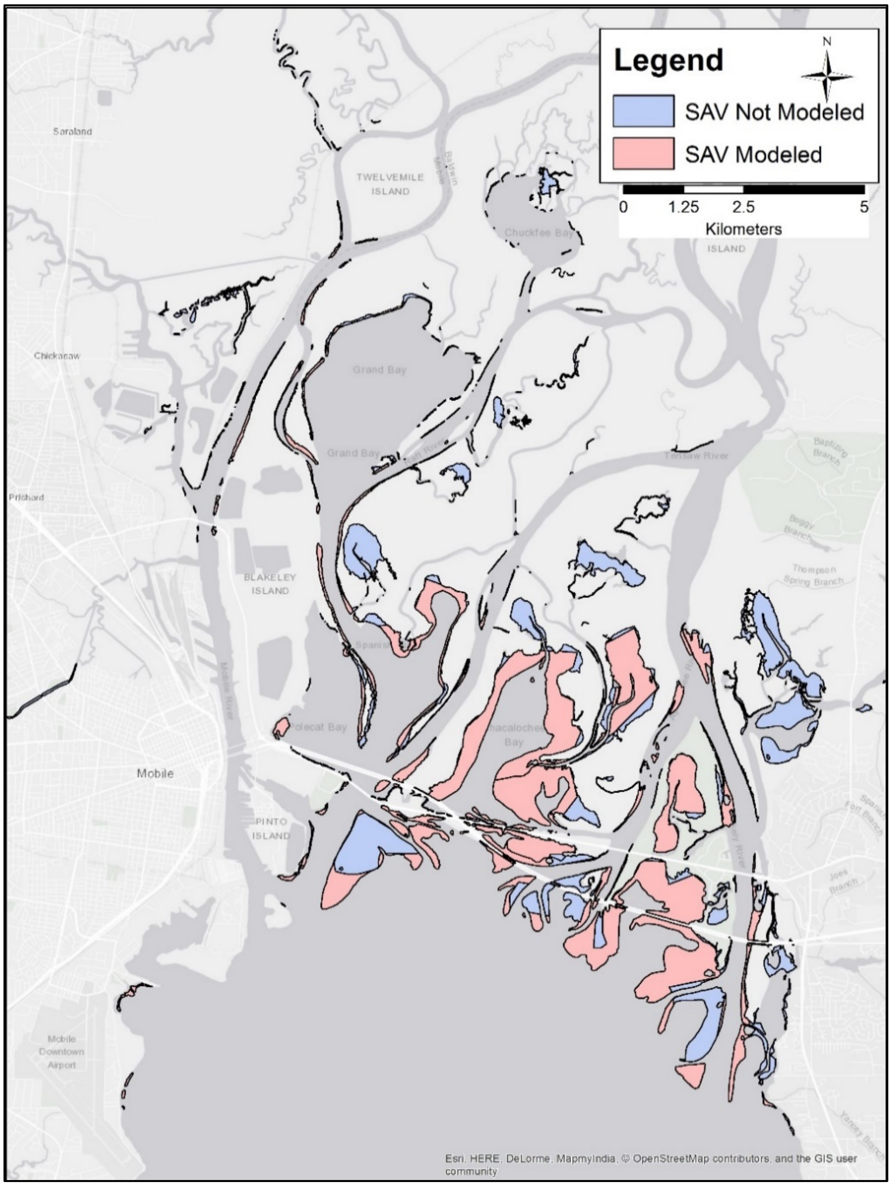
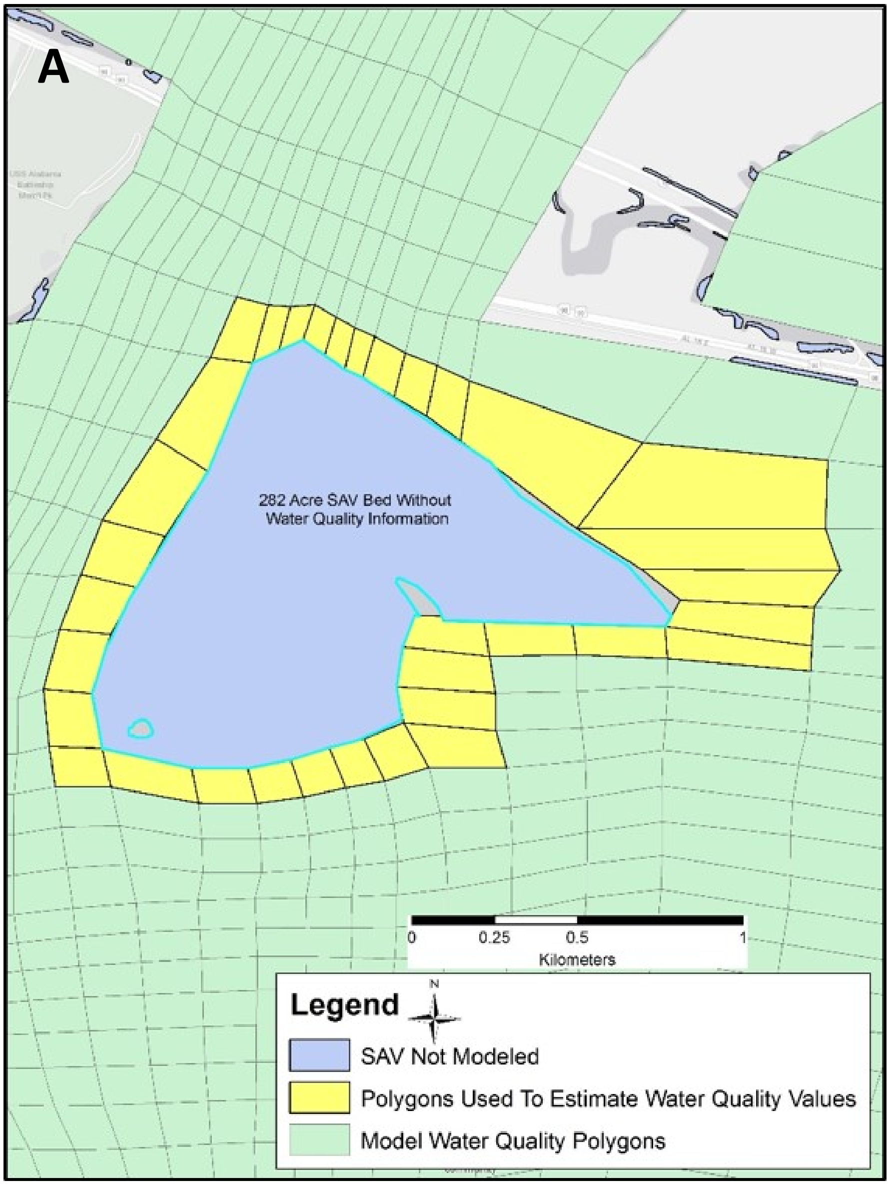


Figure 4.5. Mapped SAV beds in and out of the hydrodynamic and water quality model domains within the estuarine transition zone. The red areas are where the hydrodynamic and water quality model domains overlapped the fall 2015 Vittor SAV coverage. The blue areas are SAV beds where hydrodynamic and water quality data were not modeled so values were estimated from neighboring polygons as described in the text.


A
282 Acre SAV Bed Without
Water Quality Information
0
0.25
0.5
1
Kilometers
Legend
N
SAV Not Modeled
Polygons Used To Estimate Water Quality Values
Model Water Quality Polygons

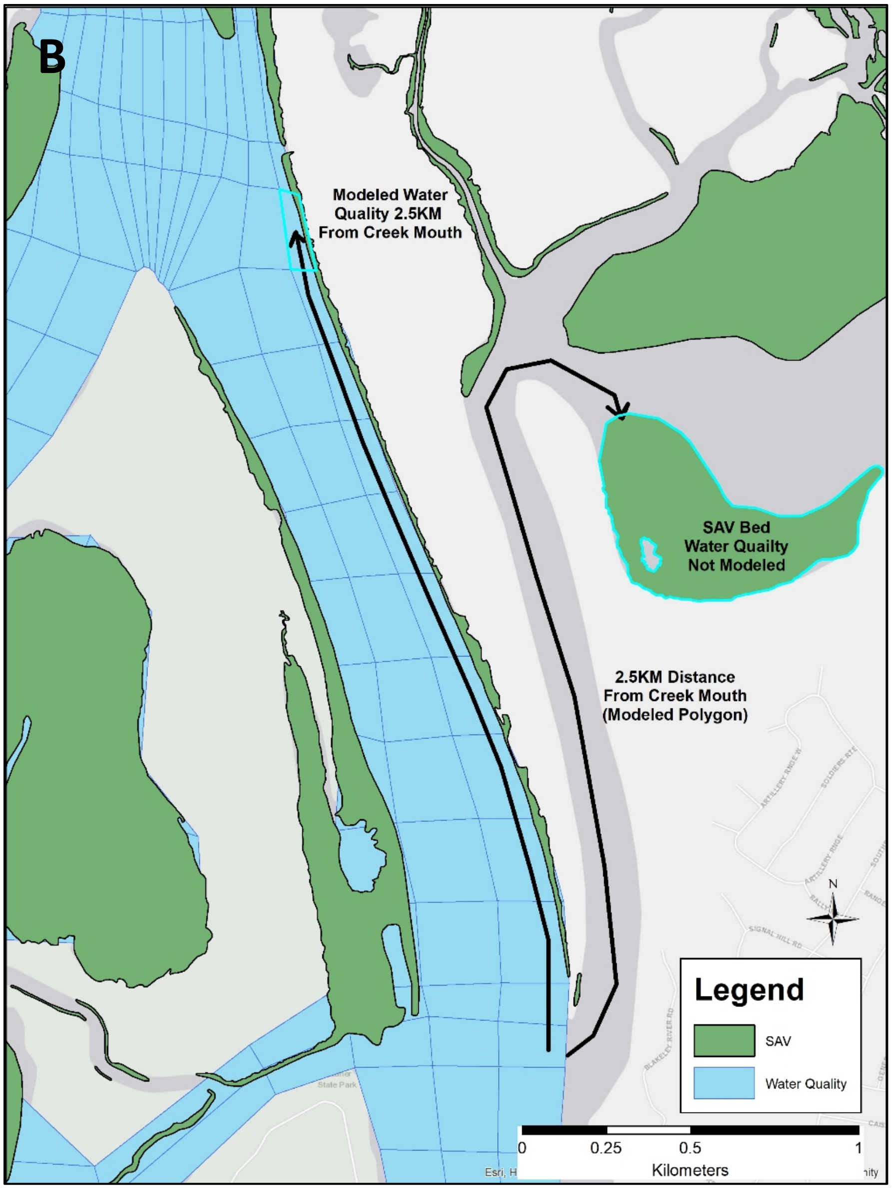


Figure 4.6. Assigning water quality values to SAV beds within the estuarine transition zone but outside of model domain. Example using adjacent water quality polygons (A) in which the blue area is a SAV bed where water quality data were not modeled and values were estimated using the mean value of neighboring polygons (yellow). The second example uses a case where there were no adjacent model water quality polygons (B). The salinity value used from the measured distance up the main river will likely still be high because it does not taking into account freshwater inputs into the creek. As a result, assigned water quality values are conservative and likely represent over estimates (i.e. higher salinities).

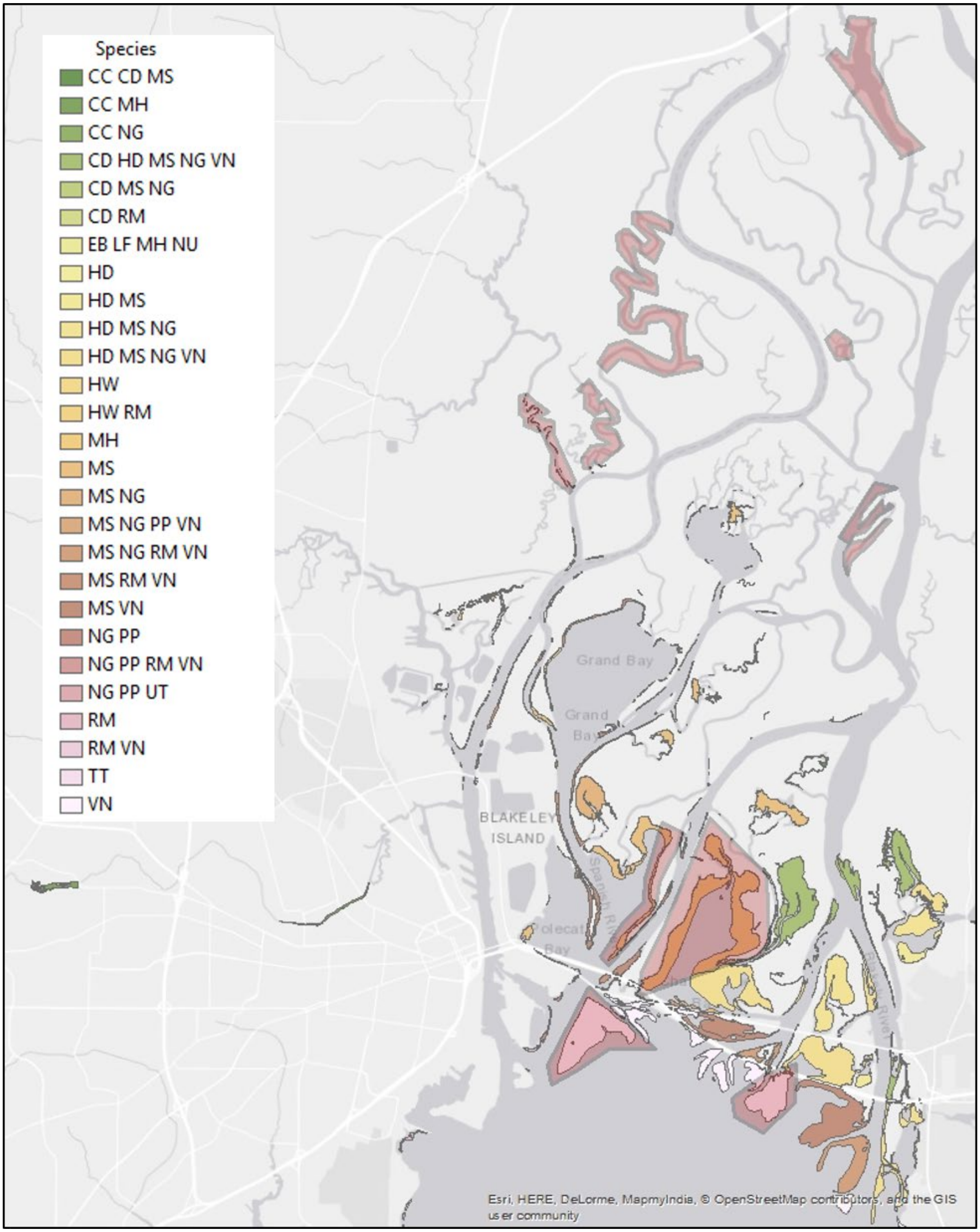


Figure 4.7. Fall 2016 Field verification sites (highlighted red polygons) and Fall 2015 SAV distribution within Mobile Bay as mapped by Vittor & Associates. Species codes can be found in Figure 4.4 and Table 4. 1.

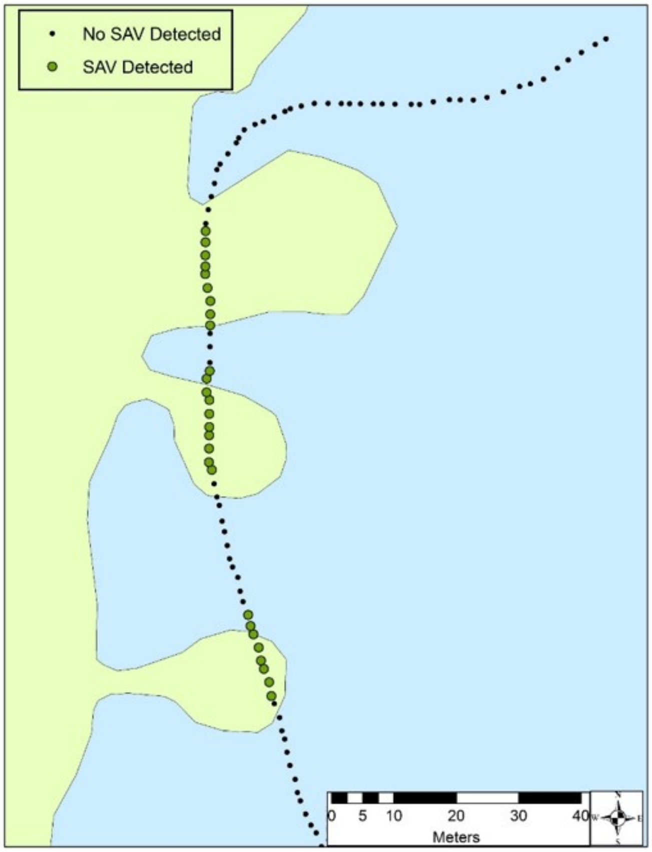


Figure 4.8. Hydroacoustic field verification of Vittor 2015 SAV maps. The light green area is SAV coverage reported by fall 2015 Vittor aerial survey and the points are hydroacoustic locations surveyed by ERDC.

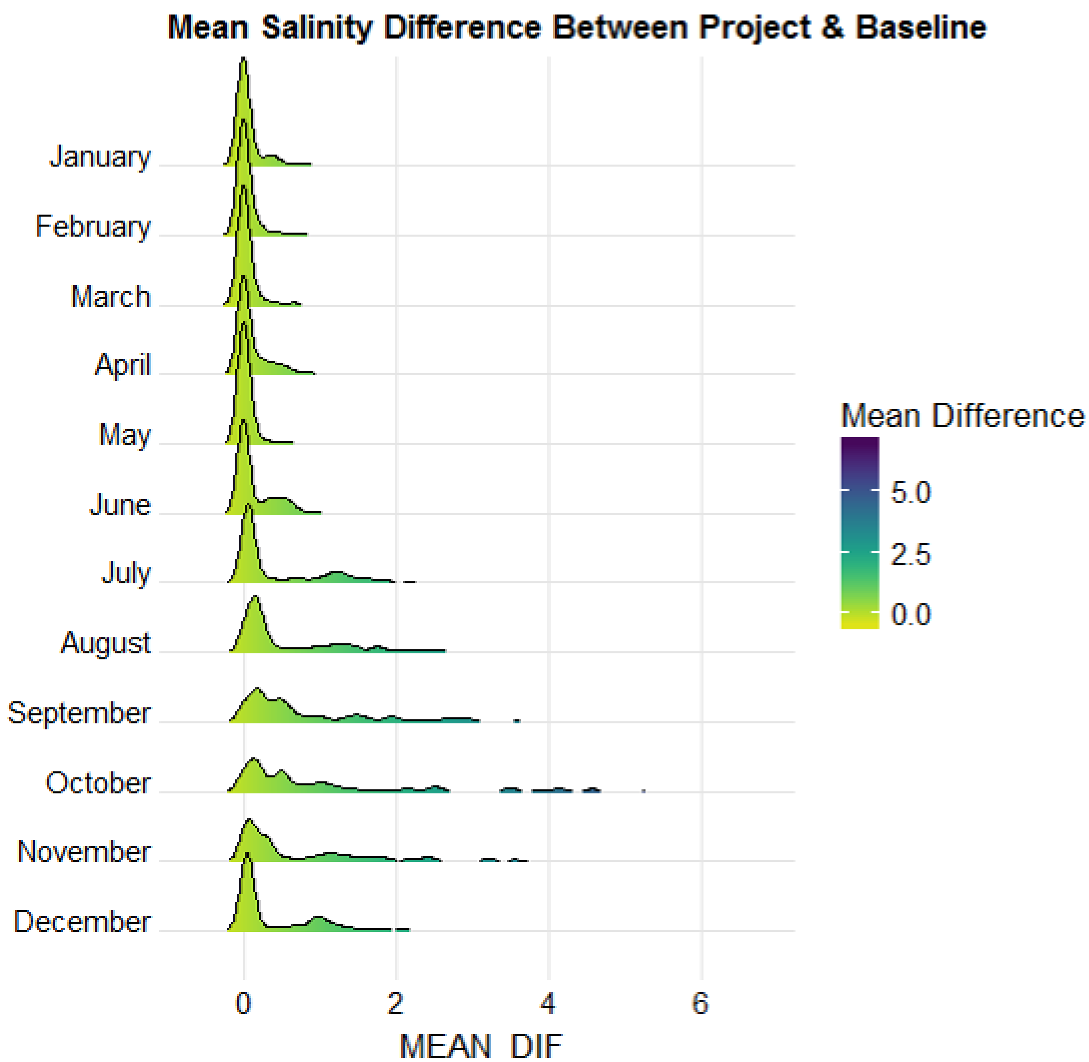


Figure 4.9 Mean depth averaged salinity differences resulting from project implementation as predicted by the hydrodynamic model (CH3D). Note largest range is in October.

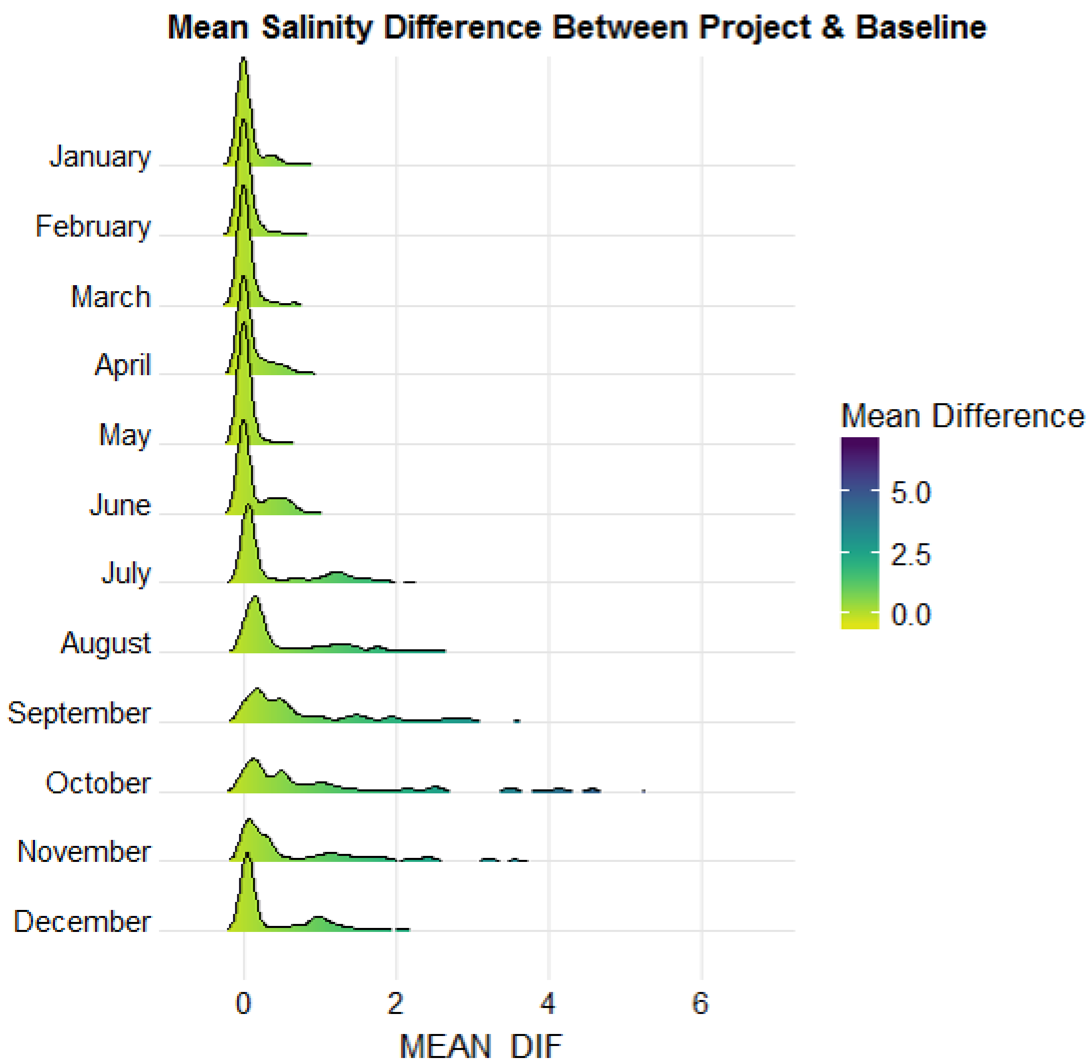


Figure 4.10. Seventy fifth percentile depth averaged salinity differences resulting from project implementation as predicted by the hydrodynamic model (CH3D). Note largest ranges are in October and November.

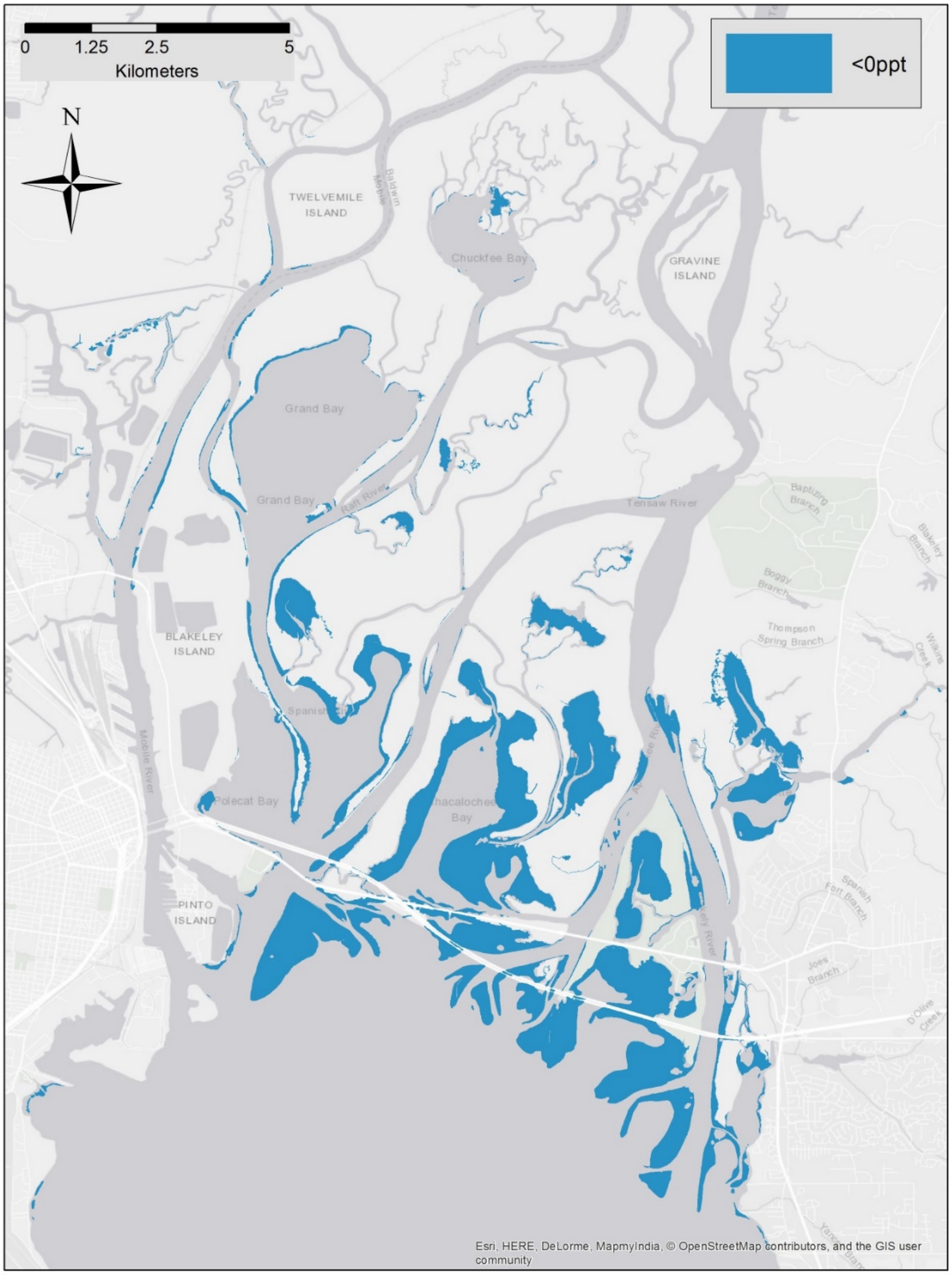


Figure 4.11.Change in Spring (May) salinity (ppt) above relative species specific thresholds values due to project implementation (i.e., post-project – baseline salinity) within the estuarine transition zone.

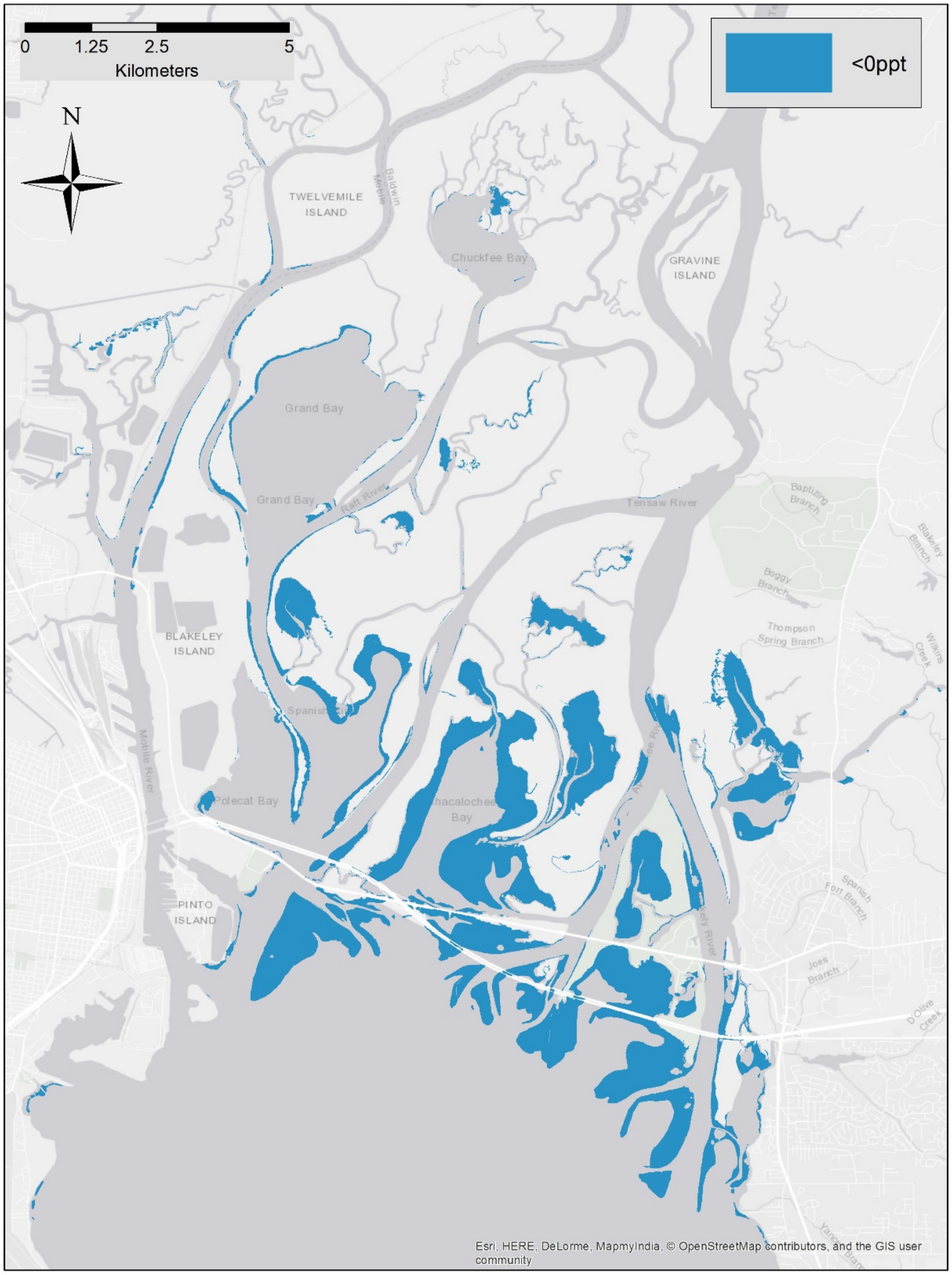


Figure 4.12. Change in Summer (August) salinity (ppt) above relative species specific thresholds values due to project implementation (i.e., post-project – baseline salinity) within the estuarine transition zone.

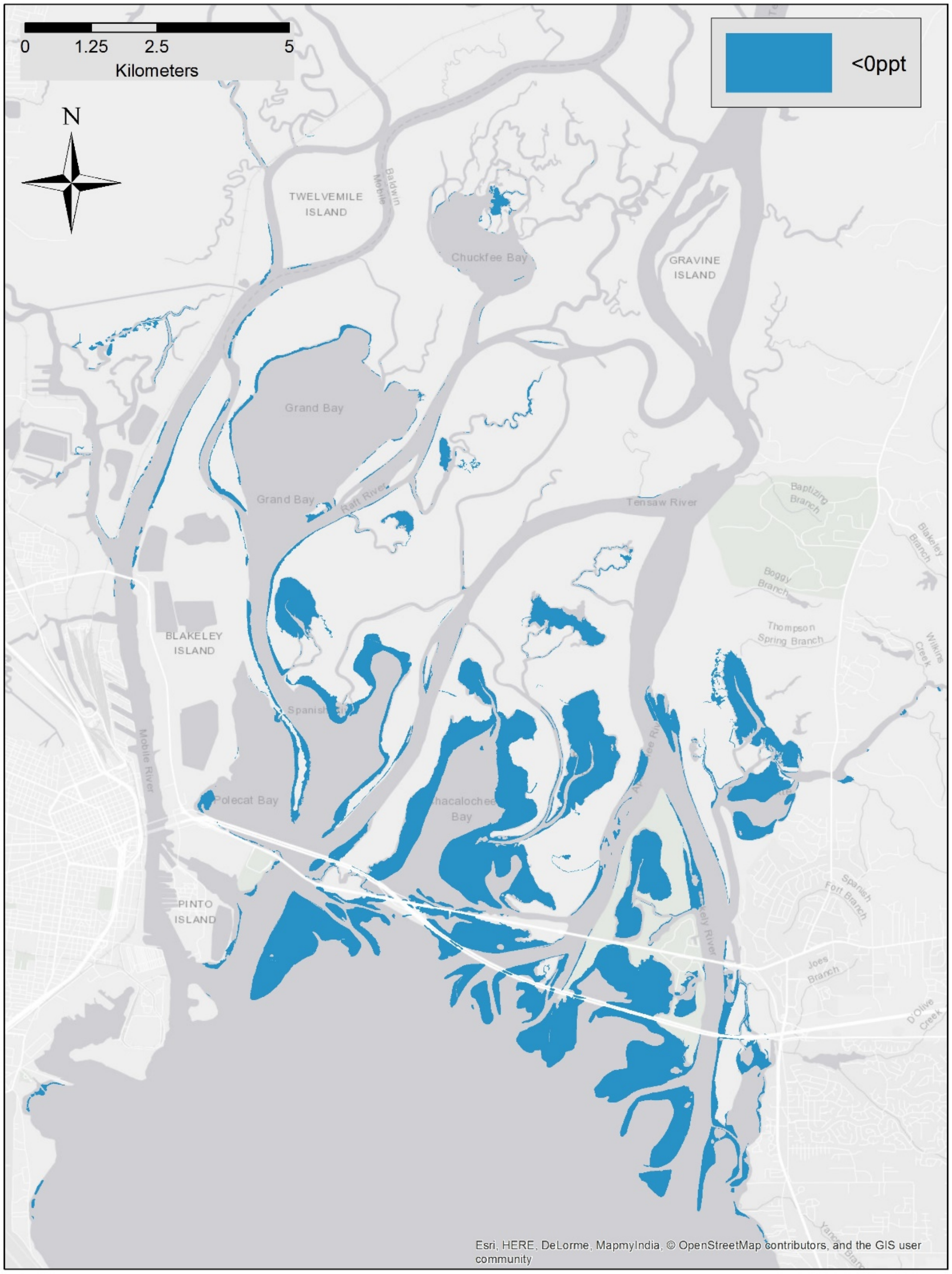


Figure 4.13..Change in Winter (February) salinity (ppt) above relative species specific thresholds values due to project implementation (i.e., post-project – baseline salinity) within the estuarine transition zone.

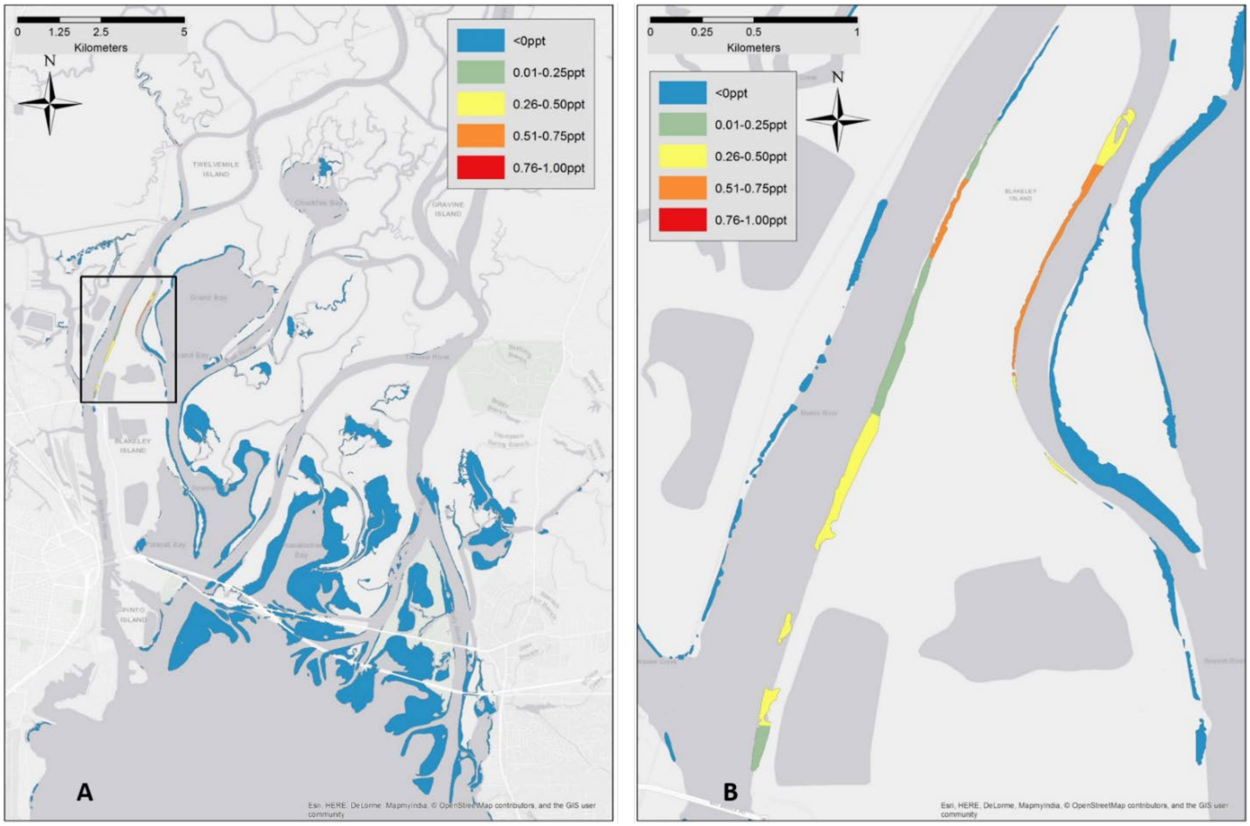


Figure 4.14. Change in Fall (October) salinity (ppt) above relative species specific thresholds values due to project implementation (i.e., post-project – baseline salinity) within the estuarine transition zone (A) and detailed within region of higher predicted salinity change (B, region outlined in black in A).

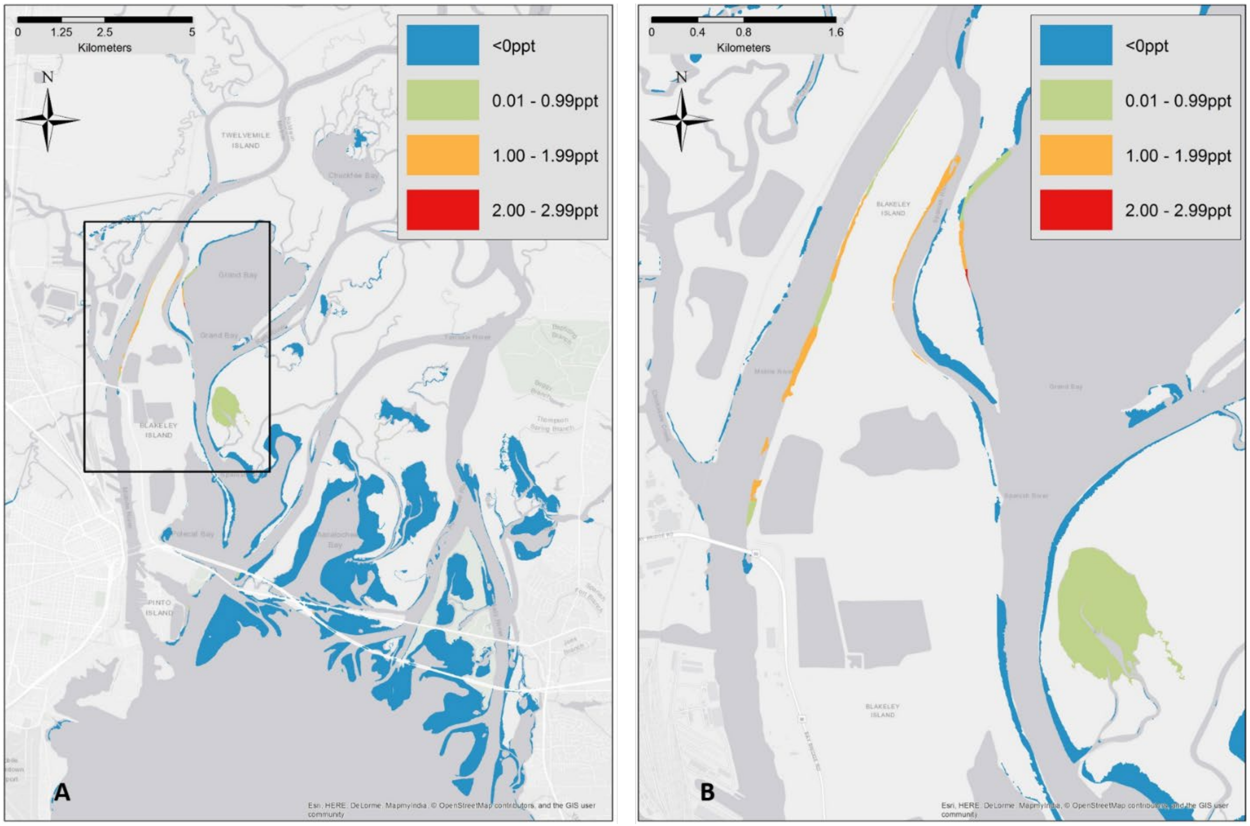


Figure 4.15. Change in salinity (ppt) above relative species specific thresholds values from current baseline conditions to projected 0.5m sea level rise conditions with no project implementation (i.e., SLR baseline – current baseline) within estuarine transition zone (A), and detailed within region of higher predicted salinity change (B, region outlined in black in A). SLR projections predict higher salinity increase than salinity increase due to project implementation alone.

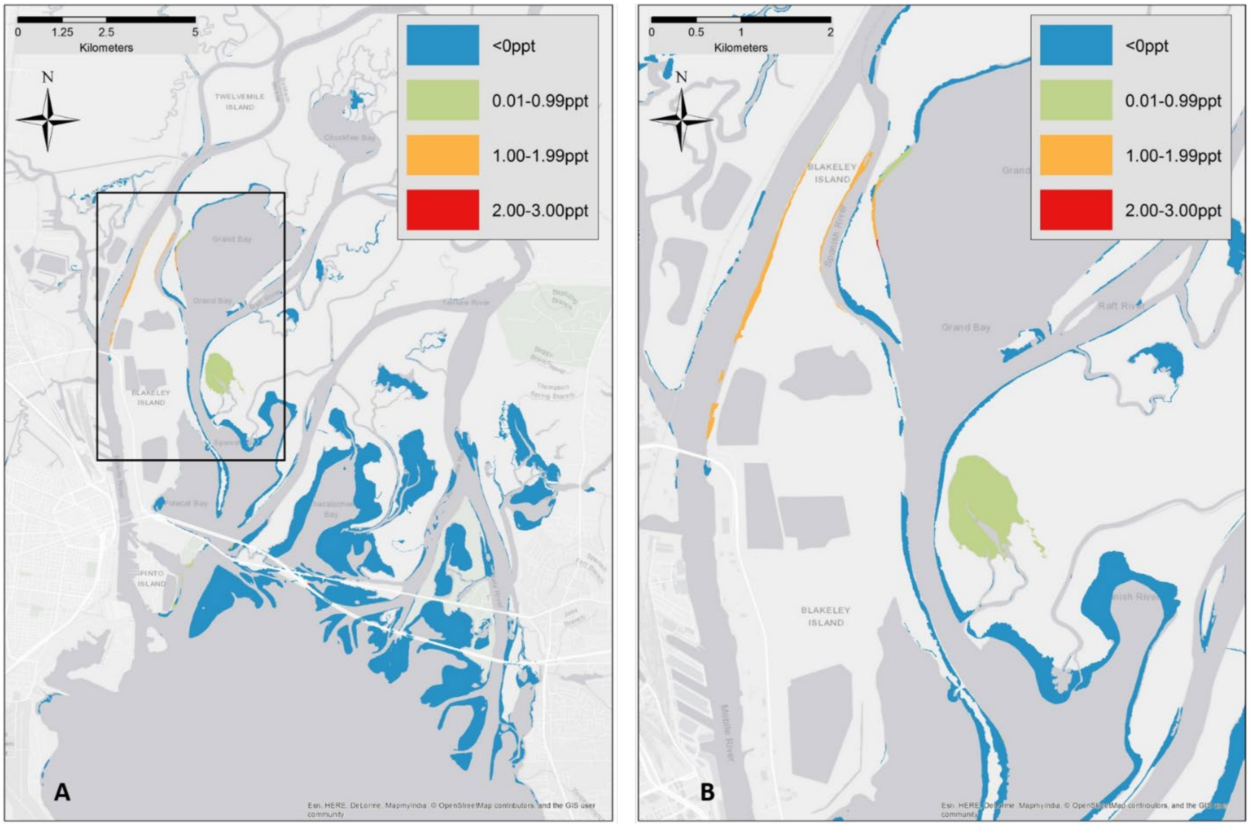


Figure 4.16. Change in salinity (ppt) above relative species specific thresholds values from current baseline conditions to projected 0.5m sea level rise conditions with project implementation (i.e., SLR post project – current baseline) within estuarine transition zone (A), and detailed within region of higher predicted salinity change (B, region outlined in black in A). SLR projections predict higher salinity increase than salinity increase due to project implementation alone.

Table 4.1. Variation in acreage over time. Values are obtained from Vittor SAV survey maps. Highlighted species are those predicted to experience increased salinities above 1ppt due to project implementation.

Species	Acres						
	2003	2009	Summer 2015	Fall 2015	Seasonal Change 2015	Mean	Standard Deviation
Myriophyllum spicatum	2318.5	2955.2	6734.8	4647.3	2087.5	4163.9	1975.7
Vallisneria neotropicalis	2610.4	2499.7	5304.3	2851.1	2453.2	3316.4	1333.4
Najas guadalupensis	762.2	1773.6	4832.9	2041.2	2791.7	2352.5	1742.9
Heteranthera dubia	427.8	312.0	3540.0	3075.9	464.1	1838.9	1707.5
Ceratophyllum demersum	954.6	188.8	2002.1	3329.4	-1327.3	1618.7	1361.3
Ruppia maritima	475.2	293.1	1767.6	632.1	1135.5	792.0	665.0
Stuckenia pectinata	0	238.9	1280.2	5.7	1274.4	381.2	609.6
Potamogeton pusillus	0	17.1	1115.1	131.2	983.8	315.8	536.0
Cabomba caroliniana	0	1.9	28.1	768.8	-740.7	199.7	379.6
Potamogeton crispus	0	27.9	375.3	9.8	365.5	103.2	181.7
Utricularia foliosa	0	5.7	213.4	114.1	99.3	83.3	101.4
Zannichellia palustris	0	0	198.8	0.2	198.6	49.8	99.4
Hydrilla verticillata	0	76.1	16.7	91.2	-74.5	46.0	44.4
Nuphar ulvacea	0	46.0	5.7	29.9	-24.3	20.4	21.4
Myriophyllum heterophyllum	0	0	5.7	29.9	-24.3	8.9	14.3
Myriophyllum aquaticum	0	0	0	0.1	-0.1	0	0.1

Table 4.2. Reported Salinity tolerance thresholds and ranges for local SAV species. Where threshold information was not available, published salinity range in known locations is reported and designated as 'Range'.

Species	Abbreviation	Common Name	Reported Salinity Tolerance or Range (ppt)	Citations	Notes
Cabomba caroliniana	*CC*	carolina fanwort	0-0.5	Poirrer et al. 2010	Rare in study area, mostly in the side creeks
Ceratophyllum demersum	*CD*	coon's tail	0-0.7 0-5	Poirrer et al. 2010 Izzati 2015	Present throughout the delta, very abundant
Halodule wrightii	*HW*	shoal grass	0-60 0-70 5-80	Texas Parks and Wildlife 1999 Kock et al. 2007 McMahan 1968, McMillian 1974	All along the Gulf of Mexico, likely not affected by project
Heteranthera dubia	*HD*	water stargrass	0-3.5 0-5	Poirrer et al. 2010 Izzati 2015	Very abundant on the east side of the delta
Hydrilla verticillata	*HV*	hydrilla	0-6.6 0-10 0-12 0-13	Haller et al. 1974 Poirrer et al. 2010 Twilley et al. 1990 Steward and Van 1987	invasive, only at 5 points up creeks in the right side of the delta
Myriophyllum aquaticum	*MA*	parrot's feather	0-10	Haller et al. 1974	Very rare in study area, in upland areas, invasive
Myriophyllum heterophyllum	*MH*	southern watermilfoil	0-5 (Range) ~6 (Range)	Sivaci et al. 2008 Eggleston et al. 2008	Very rare in study area, one patch far up a creek
Myriophyllum spicatum	*MS*	Eurasian watermilfoil	0-13 0-15 0-15 0-20	Haller et al. 1974 Aiken et al 1979 Izzati 2015 Poirrer et al. 2010	Present throughout the delta, invasive
Najas guadalupensis	*NG*	southern naiad	0-3.5 0-10 0-10	Poirrer et al. 2010 Texas Parks and Wildlife 1999 Haller et al. 1974	Present throughout the delta, very abundant
Potamogeton crispus	*PC*	curly pondweed	0-8 (Range)	Vincent 2001	Rare but spread throughout the delta, invasive
Potamogeton diversifolius	*PD*	water thread pondweed	0 (Range)	USDA, NRCS 2018	Present in the bay far downstream of areas of salinity change
Potamogeton nodosus	*PN*	longleaf pondweed	0 (Range) 0-1.3 (Mean, Range)	USDA, NRCS 2018 Castellanos and Rozas 2001	Present in the bay far downstream of areas of salinity change
Potamogeton pusillus	*PP*	small pondweed	0-3.5	Poirrer et al. 2010	Present in the bay far downstream of areas of salinity change
Ruppia maritima	*RM*	widgeon grass	0-60 0-70 0-100	Phillips 1960 Kock et al. 2007 Kantrud 1991	Present throughout the entire study region
Stuckenia pectinata	*SP*	sago pondweed	0-15, can likely handle above 20	Borgnis and Boyer 2014	Present only in lower part of the delta, not likely to be affected by project
Thalassia testudinum	*TT*	turtle grass	5-45 20-40 36-70	Lirman and Cropper 2003 Zieman 1982 Kock et al. 2007	One patch by the Gulf of Mexico, out of project area
Utricularia foliosa	*UF*	leafy bladderwort	0-5 (Range) 1-3.5 (Range)	Camargo and Florentino 2000 Ross et al. 2000	A few patches up the creeks on the east side of the lower delta
Utricularia inflata	*UI*	floating bladderwort	0-0.02 (Range)	de Roa et al. 2002	Rare, one patch miles away from the lower delta
Vallisneria neotropicalis	*VN*	wild celery	0-18 0-18 0-18 0-18	Doering et al. 2001 Kraemer et al. 1999 Boustany et al. 2010 Lauer et al. 2011	Widespread, species observed in areas higher than 18 ppt
Zannichellia palustris	*ZP*	horned pondweed	0-6	Greenwood and DuBowy 2005	A few patches present up creeks at the mouth of the Bay, not likely to be affected by the project

Table 4.3.Number of SAV acres predicted to experience a change in salinity exposure, displayed by range of predicted salinity change.

Post Project Salinity (ppt) above SAV tolerance threshold		
Range	**Mean Acres**	**75th Percentile Acres**
<0	0	82
0-0.25	5249	5235
0.25-0.5	774	556
0.5-0.75	1080	601
0.751-1.0	120	742
1-1.25	50	58
1.25-1.5	2	1

Table 4.4. Number of SAV acres, by most vulnerable species, predicted to experience a change in mean monthly salinity exposure, displayed by range of predicted salinity change.

	Species within SAV Bed with lowest Salinity Tolerance						
Post-Project Monthly Mean Salinity (ppt) above SAV tolerance threshold	Water Star Grass	Eurasian Watermilfoil	Southern Naiad	Widgeon Grass	Wild Celery	Carolina Fanwort	Coon's Tail
<0							
0-0.25	3288	561	284	5	401	82	41
0.25-0.5	18	257	60	12	106	15	
0.5-0.75		313	164		412		25
0.75-1.0		1		1	9		
1-1.25		3	21	20	2		
1.25-1.5				2			

Table 4.5. Number of SAV acres, by most vulnerable species, predicted to experience a change in monthly 75th percentile salinity exposure, displayed by range of predicted salinity change.

	Species within SAV Bed with lowest Salinity Tolerance						
Post-Project Monthly 75th Percentile Salinity (ppt) above SAV tolerance threshold	Water Star Grass	Eurasian Watermilfoil	Southern Naiad	Widgeon Grass	Wild Celery	Carolina Fanwort	Coon's Tail
<0							
0-0.25	3285	557	281	16	386	82	41
0.25-0.5		171	185	4	62	15	
0.5-0.75	25	380	52	14	66		25
0.75-1.0		32	11	3	309		
1-1.25		4	21	3	25		
1.25-1.5					1		

4.4 References

Aiken, S.G., P.R. Newroth and I. Wile. 1979. The biology of Canadian weeds. 34. Myriophyllum spicatum L. Canadian Journal of Plant Science 59:201-215.

Biber, P. and H.J. Cho. 2017. Coastal seagrass and submerged aquatic vegetation

Borgnis, E. and Boyer, K.E., 2016. Salinity tolerance and competition drive distributions of native and invasive submerged aquatic vegetation in the Upper San Francisco Estuary. *Estuaries and coasts*, *39*(3), pp.707-717.

Boustany, R.G., Michot, T.C. and Moss, R.F., 2010. Effects of salinity and light on biomass and growth of Vallisneria americana from Lower St. Johns River, FL, USA. *Wetlands ecology and management*, *18*(2), pp.203-217.

Camargo, A.F.M. and Florentino, E.R., 2000. Population dynamics and net primary production of the aquatic macrophite Nymphaea rudgeana CF Mey in a lotic environment of the Itanhaém River basin (SP, Brazil). *Revista Brasileira de Biologia*, *60*(1), pp.83-92.

Castellanos, D., & Rozas, L. (2001). Nekton Use of Submerged Aquatic Vegetation, Marsh, and Shallow Unvegetated Bottom in the Atchafalaya River Delta, a Louisiana Tidal Freshwater Ecosystem. Estuaries, 24(2), 184-197.

de Roa, E.Z., Gordon, E., Montiel, E.D.I.E., Delgado, L.A.U.R.A., Berti, J. and Ramos, S., 2002. Association of cyclopoid copepods with the habitat of the malaria vector Anopheles aquasalis in the peninsula of Paria, Venezuela. Journal of the American Mosquito Control Association-Mosquito News, 18(1), pp.47-51.

Doering, P.H., Chamberlain, R.H. and McMunigal, J.M., 2001. Effects of simulated saltwater intrusions on the growth and survival of wild celery, Vallisneria americana, from the Caloosahatchee Estuary (South Florida). *Estuaries*, *24*(6), pp.894-903.

Eggleston, D.B., Johnson, E.G., Kellison, G.T., Plaia, G.R. and Huggett, C.L., 2008. Pilot evaluation of early juvenile blue crab stock enhancement using a replicated BACI design. Reviews in Fisheries Science, 16(1-3), pp.91-100.

Frazer, T. K., Notestein, S. K., Jacoby, C. A., Littles, C. J., Keller, S. R., & Swett, R. A. 2006. Effects of storm-induced salinity changes on submersed aquatic vegetation in Kings Bay, Florida. *Estuaries and Coasts*, *29*(6), 943-953.

Greenwood, M.E. and DuBowy, P.J., 2005. Germination characteristics of Zannichellia palustris from New South Wales, Australia. *Aquatic botany*, *82*(1), pp.1-11.

Haller, Sutton and Barlowe 1974. Effects of salinity on growth of several aquatic macrophytes. Ecology 55:891-894

Hinojosa-Garro, D., Mason, C. F., & Underwood, G. J. 2008. Macrophyte assemblages in ditches of coastal marshes in relation to land-use, salinity and water quality. *Fundamental and Applied Limnology/Archiv für Hydrobiologie*, *172*(4), 325-337.

Izzati, M., 2016, September. Salt Tolerance of Several Aquatic Plants. In *Proceeding International Conference on Global Resource Conservation* (Vol. 6, No. 1).

Kantrud, H.A. 1991. Wigeongrass (Ruppia maritima L.): A literature review. U.S. Fish and Wildlife Service, Fish and Wildlife Research 10. Jamestown, ND: Northern Prairie Wildlife Research Center online: http://www.npwrc.usgs.gov/resource/plants/ruppia/index. htm. Accessed 1 August 2011.

Koch, M.S., Schopmeyer, S.A., Kyhn-Hansen, C., Madden, C.J. and Peters, J.S., 2007. Tropical seagrass species tolerance to hypersalinity stress. *Aquatic Botany*, *86*(1), pp.14-24.

Kraemer, G.P., Chamberlain, R.H., Doering, P.H., Steinman, A.D. and Hanisak, M.D., 1999. Physiological responses of transplants of the freshwater angiosperm Vallisneria americana along a salinity gradient in the Caloosahatchee Estuary (Southwestern Florida). *Estuaries*, *22*(1), pp.138-148.

Lauer, N., Yeager, M., Kahn, A.E., Dobberfuhl, D.R. and Ross, C., 2011. The effects of short term salinity exposure on the sublethal stress response of Vallisneria americana Michx.(Hydrocharitaceae). *Aquatic botany*, *95*(3), pp.207-213.

Lirman, D. and Cropper, W.P., 2003. The influence of salinity on seagrass growth, survivorship, and distribution within Biscayne Bay, Florida: field, experimental, and modeling studies. *Estuaries*, *26*(1), pp.131-141.

McMahan, C.A., 1968. Biomass and salinity tolerance of shoalgrass and manateegrass in Lower Laguna Madre, Texas. *The Journal of Wildlife Management*, pp.501-506.

McMillan, C., 1974. Salt tolerance of mangroves and submerged aquatic plants. In *Ecology of halophytes* (pp. 379-390).

Moore, K., 2012. St. Joohns River Water Supply Impact Study, Appendix 9.B. Submerged Aquatic Vegetation (SAV) in the Lower St. Johns River and the Influences of Water Quality Factors on SAV. Prepared for the St. Johns River Water Management District.

Phillips, R.C. 1960. Observations on the ecology and distribution of the Florida seagrasses. Florida Board of Conservation Professional Paper Series 2: 1-72.

Poirrier, M.A., Burt-Utley, K., Utley, J.F. and Spalding, E.A., 2010. Submersed Aquatic Vegetation of the Jean Lafitte National Historical Park and Preserve. *Southeastern Naturalist*, *9*(3), pp.477-486.

Ross, M., Meeder, J., Sah, J., Ruiz, P., & Telesnicki, G. (2000). The Southeast Saline Everglades Revisited: 50 Years of Coastal Vegetation Change. Journal of Vegetation Science, 11(1), 101-112.

Sabol, B., D. Shafer, E. Melton, J.Jarvis, S. Evert and R Loyd. 2014. SAVEWS Jr. User's Manual, Version 1.0. Final Report prepared for US Army Corps of Engineers, ERDC/EL TR-14-8.

Sand-Jensen, K., Prahl, C. and Stokholm, H., 1982. Oxygen release from roots of submerged aquatic macrophytes. *Oikos*, pp.349-354.

Sivaci, A., Elmas, E., Gümüş, F. and Sivaci, E.R., 2008. Removal of cadmium by Myriophyllum heterophyllum Michx. and Potamogeton crispus L. and its effect on pigments and total phenolic compounds. Archives of environmental contamination and toxicology, 54(4), pp.612-618.

Steward, K.K. and Van, T.K., 1987. Comparative studies of monoecious and dioecious hydrilla (Hydrilla verticillata) biotypes. *Weed Science*, pp.204-210.

Texas Parks and Wildlife. 1999. Seagrass conservation plan for Texas. Final Report. 84pp.

Twilley, R.R. and Barko, J.W., 1990. The growth of submersed macrophytes under experimental salinity and light conditions. *Estuaries*, *13*(3), pp.311-321.

USDA, NRCS. 2018. The PLANTS Database (http://plants.usda.gov, 9 April 2018). National Plant Data Team, Greensboro, NC 27401-4901 USA.

Vincent, W.J., 2001. Nutrient partitioning in the upper Canning River, Western Australia, and implications for the control of cyanobacterial blooms using salinity. Ecological Engineering, 16(3), pp.359-371.

Vittor, B.A. and Associates, Inc. 2016. Submerged aquatic vegetation mapping in Mobile Bay and adjacent waters of coastal Alabama in 2015. Report prepared for Mobile Bay National Estuary Program, Mobile, AL and Alabama DCNR State Lands Division Coastal Section.

Vittor, B.A. and Associates, Inc. 2010. Submerged aquatic vegetation mapping in Mobile Bay and adjacent waters of coastal Alabama in 2008 and 2009. Report prepared for Mobile Bay National Estuary Program, Mobile. AL.

Vittor, B.A. and Associates, Inc., 2004. Mapping of Submerged Aquatic Vegetation in Mobile Bay and Adjacent Waters of Coastal Alabama in 2002. Prepared for the Mobile Bay National Estuary Program, Mobile, AL.

Zieman, J 1982. The ecology of the seagrasses of south florida: A community profile. USFWS, Office of Biological Services, Washington, D.C. FWS/OBS-82 (reprinted September 1985).

Chapter 5: Oysters

Summary

This chapter examines the potential effects of the harbor deepening project on oyster larvae movement and mortality. Oyster larvae dynamics within the Mobile Bay were simulated by integrating modeled hydrodynamic data, water quality data, and a particle-tracking model

(PT123) incorporating the physical behavior of oysters. Oyster modeling occurred in two phases (A) an initial phase that included simulating oyster larval releases from the Brookley Reef under four scenarios: 1) a baseline scenario of future-without-project and without projected sea level rise (SLR), 2) a project involving the implementation of deepening Mobile Harbor via dredging the navigation channel within Mobile Bay and without projected SLR conditions, 3) a scenario of future Without-Project with projected SLR, and 4) a project involving the implementation of harbor deepening with projected SLR conditions, and (B) a detailed analysis of the spatial distribution of oyster larvae that was specifically designed to address public comments; this analysis simulated larval releases from 18 reefs (the Brookley reef and 17 additional reefs) under future-without-project and future-with-conditions.

In the initial phase, differences in larval transport and survival were conducted using a single release location (the Brookley reef in upper Mobile Bay) with 42 particles released for each scenario. Under the assumptions used for this model parameterization, settlement locations of simulated larvae were all within the boundaries of Mobile Bay. The scenarios with SLR (i.e., Scenarios 3 & 4) resulted in a much higher mortality of oyster larvae when released at Brookley reef, although that was not the case for the scenarios without SLR. The model results specific to Brookley Reef did not project an increase in larvae flushing out of Mobile Bay under the with channel modification project scenarios (i.e., Scenarios 2 & 4), however, this analysis is limited to a small number of particles being released from the Brookley Reef.

In the second phase of oyster larvae modeling detailed analysis, 5400 particles were released from 18 identified oyster reef locations and their movements were tracked and analyzed. Larvae were considered to be flushed from the system if their final locations were located south of the two barrier islands at the bay’s outlet and oyster mortality was dictated by oyster exposure to water quality below or above a specified mortality threshold. Overall, 20% and 18% of the oyster larvae were flushed from the system for the baseline and project scenarios, respectively. Further analysis revealed that final larvae locations and trajectories were statistically similar for both the baseline and project scenarios. Each of the 18 specific reefs were further analyzed and results showed no significant changes in particle locations and flushing. Larvae were unaffected by dissolved oxygen as it stayed above the mortality threshold for both scenarios. However, 33% of the oyster larvae in the baseline simulation and 28% of the larvae in the project simulation suffered mortality due to exposure to low salinity for an extended duration. Results collectively indicate that the project would have minimal impacts on oyster movement and mortality.

5.1 Introduction

Eastern oyster (*Crassostrea virginica*) recruitment is the key driver for maintaining oyster population over time. However, this process is poorly understood due to the difficulty in tracking oyster larvae over time. Recruitment occurs through the settlement of larvae from their natal reef (intra-reef recruitment), or from other reefs within the system (inter-reef recruitment). Intra-reef recruitment has been shown to be relatively low, indicating that inter-reef recruitment is crucial for sustaining oyster populations in hydrodynamically-driven systems. Oyster larvae have limited swimming abilities so their movement is controlled in large part by hydrodynamic transport. Oyster larvae have a maximum swim speed on the order of two to three mm/s (North et al., 2006, 2008), which is negligible in comparison to the horizontal velocities typically observed in most estuarine systems. However, vertical velocities are much lower, and veligers are able to overcome vertical velocity gradients to change their vertical position in the water column. In addition to hydrodynamic forcings, oyster veligers also respond to changes in water quality (e.g., temperature, salinity, DO). Salinity is a recognized driver for both larval and adult oyster dynamics (Gunter 1955, Kennedy et al. 1996), with the optimal range of salinities being in mesohaline conditions, which facilitates oyster growth in disease-prone waters (Carnegie & Burreson 2011, Levinton et al. 2011).

Understanding the oyster larvae movement and reef recruitment dynamic is critical towards understanding how potential project actions will impact oyster populations within a project footprint. Specifically, local oyster recruitment within the Mobile Bay area could be negatively impacted if a higher percentage of oyster larvae are flushed out of the bay due to hydrodynamic changes caused by alterations to the navigation channel.

The complexity of the oyster life cycle, coupled with the difficulty in tracking oyster larvae in the field, facilitates an integrated ecological modeling approach for understanding system dynamics. Eulerian-Lagrangian particle tracking models developed for visualizing flow fields, estimating contaminant transport paths, or estimating sediment transport can be adapted for tracking biological particles by applying behavior rules that can supersede physical rules (e.g., Tate et al. (2012) successfully modified it to simulate various fish egg behaviors in the Mississippi River Gulf Outlet). A common particle tracker, PT123 (Cheng et al. 2011), uses water level and current estimates from two- and three-dimensional hydrodynamic models to predict where sediments or other discrete constituents are transported. We modified PT123 with biologically-based behaviors to simulate and track oyster larvae within the system (conceptualized in Figure 5.1).

The main objectives were to assess oyster larvae movement and survival under two different scenarios for Mobile Bay: 1) a baseline scenario of without-project and 2) a scenario of with-project involving the implementation of deepening Mobile harbor via dredging the navigation channel within Mobile Bay.

5.2 Methods

Model Development Process

The model simulates the responses of oyster larvae to physical processes and biological behavior. The model produces a veliger particle transport success rate, which must be combined with a veliger particle mortality rate dependent upon simulated local water quality conditions to provide an estimate of recruitment rates.

The model was implemented as a library that was added to PT123, an existing particle tracking/engineering model. In this case, biological behaviors were parameterized as a set of rules governing growth, swimming ability, and fall velocity that represent the current state of knowledge of larval life history strategies (e.g., growth, settling rate) and how oyster larvae respond to the physical environment. Mortality was assessed in a separate analysis based on known larval response to environmental conditions such as salinity and dissolved oxygen (DO). The model and mortality analysis were based on models developed by North et al., (2008), Kim et al., (2010) and Kjelland et al., (2015).

Model Description

The oyster model is driven by the Geophysical Scale Transport Modeling System (GSMB) hydrodynamic code, which simulates water level, current velocities, and constituent transport in the system of interest using the Curvilinear-grid Hydrodynamics 3D Multi-Block (CH3D-MB). CH3D-MB uses a horizontal boundary-fitted curvilinear grid with a vertical sigma grid, and is suitable for application to coastal and near shore waters (Chapman and Luong, 2009). The integrated compartment model (ICM), i.e., water quality model, was coupled with CH3D to provide water quality parameters for the oyster model. Both CH3D and ICM are mature codes that have been thoroughly documented in other studies (Cerco and Noel 2004). We focus on providing details for the model we developed for quantifying the processes and dynamics of the biological behaviors of pelagic oyster veligers. The model was applied to 18 reefs in the Mobile Bay system (Figures 5.1).

An existing Eulerian-Lagrangian particle tracking model, i.e., PT123, (Cheng et al. 2011), was modified with oyster larval behaviors to simulate oyster reef connectivity and recruitment within Mobile Bay. Given velocities, PT123 can track massless particles in 1-, 2-, and 3-D unstructured or converted structured meshes. The elements used to construct PT123 meshes are line elements in 1-D, triangular and/or quadrilateral elements in 2-D, and tetrahedral, triangular prism, and/or hexahedral elements in 3-D (Cheng et al. 2011). One adaptive (embedded 4th- and 5th-order) and three non-adaptive (1st-, 2nd-, and 4th-order) Runge-Kutta (RK) methods are included in PT123 to solve the ordinary differential equations describing the motion of particles (Cheng et al. 2011). Particles are tracked along the closed boundary and stops tracking when a particle encounters the boundary. The start and end times of tracking are flexible as long as their corresponding velocities can be computed via temporal interpolation using the given velocities (Cheng et al. 2011).

Model Rules

Veliger density and swimming ability changes with age so the basic behavior rules were simplified to best approximate the vertical distribution of veligers in a well-mixed system. We developed a rule set to achieve a temporally varying vertical distribution of veliger particles consistent with North et al. (2008) and Kim et al. (2010). Attachment was assumed once the veligers matured to an age at which settlement could occur (assumed to be 14 to 21 days) and reached the bay bottom. For this iteration of PT123, we assumed that oyster larvae could settle anywhere within the bay, although attachment success was not accounted for, and no recruitment entered the system from outside the modeled reefs. Three instantaneous particle releases were simulated (consistent with Kim et al., 2010) from the modeled reefs, with a total of 1800 particles per release.

The particles were modified to capture the behavior of oyster larvae using the following rules:

1) Particle size increases linearly from 50 to 300 µm over a three-week period after release into the system (i.e., a constant growth rate of 12.5 µm/day)

2) Vertical biological velocity (m/s) is the sum of the vertical fall velocity of the larvae and the positive swim speed, both size dependent. The fall velocity is calculated as

$$w_{fall} = -\frac{0.0304*size+1.099}{1000} \tag{1}$$

where *size* is the particle size in µm and the swim velocity is calculated as

$$w_{swim} = \frac{0.0089*size-0.076}{1000} \tag{2}$$

The fall and swim velocities are combined to determine the total biological velocity. Particles are assumed to be actively swimming only a fraction of each time step so the net velocity is calculated as

$$w_{bio} = w_{fall}(1-r) + w_{swim} * r \tag{3}$$

where *r* is the fraction of each time step that a particle is assumed to be swimming. Dekshenieks et al. (1996) determined larvae swim between 64 to 83% of the time so *r* is a random number between 0.64 and 0.83. Since fall and swim velocities increase at different rates as the larva grow, the net result is particles initially remain near the surface. Particles migrate downward based on their size until a larva reaches the bottom or until the maximum time span allotted for larval mobility is reached, at which point they then settle to the bottom.

4) Time span oyster larvae are competent to settle: 14 to 21 days.

The three-dimensional velocity output from 42868-cell CH3D-MB grid (sigma-stretched grid) were reduced to three sigma layers and used to drive the PT123 oyster behavior model at a time step of 200 seconds. Model variables, corresponding parameters, and mortality analysis thresholds can be found in Table 5.1.

Larvae release locations were located at the reef locations shown in Figure 5.1 (listed in Table 5.1). Overall, 5400 larval particles were released. On each release date, 100 particles were released from 18 reef locations (Figure 1), making up a total of 1800 particles released per release date. Larvae were released on 1 April, 14 June, and 27 August. Larva location was captured at 200 second increments and tracked over 21 days post-release. Final locations were recorded and mapped, larval densities were calculated and heat maps were created to identify hotspots for larval settlement.

The juvenile and adult mortality analysis consisted of comparing larval trajectories with the location of adverse salinity and dissolved oxygen (DO) conditions for each particle release based on larval tolerances (from Kjelland et al., 2015 and Kennedy et al., 2009). For each scenario, salinity from CH3D-MB and DO from ICM were summarized--monthly statistics were calculated for mean, standard deviation, minimum, maximum, and the following percentiles 1, 5, 10, 25, 50, 75, 90, 95, and 99. For vertical reference, statistical values were assessed for depth-averaged, top, top 3 layers, bottom 3 layers, and bottom layers, consistent with the vertical resolution of PT123. Larval mortality was assumed to occur if oyster larvae were exposed to salinity conditions outside of the threshold for 50 time steps (i.e., 10,000 seconds) based on its position in the water column. Larval mortality was assumed to occur if oyster larvae were exposed to DO conditions below the minimum threshold at the 1st-percentile, which represent the lowest 1% of modeled DO conditions for each month.

5.3 Results and Discussion

Model Evaluation and Application

The final larval particle locations are shown in Figure 5.2. To better visualize the distribution of the larval particles, point density maps of the final larval positions were constructed to show the relative density of the particles across the model domain and are shown in Figure 5.3. The overall pattern between the without and with project conditions are similar and agree well with the behavioral particle tracking results of Kim et al. (2010) despite the use of different hydrodynamic and particle tracking models as well as different parameterizations of oyster behavior. Most of the larvae end up in the Cedar Point and Eastern Mississippi Sound (southwestern portion of the model domain) with another concentrated area of particle settlement in Bon Secour Bay. PT123 oyster larval tracking is probabilistic in nature; the vertical position of each particle is a function of how much time it spends actively swimming producing a more realistic representation of the natural variability of larval distribution. Consequently, the final positions of the larval particles are not the same between model runs so the point density map similarity was assessed using the Warren similarity index, calculated using the R SDMTools package (Warren et al., 2008; v1.1-221 VanDerWal et al., 2014). The Warren similarity index is used to determine ecological niche model distribution overlap and has been used in a variety of habitats; 1 is the similarity index value if the spatial distributions are identical and 0 is the similarity index value if there is no overlap between the spatial distributions. Comparing the without and with project point density maps, the similarity index value is 0.977 and indicates the larval particle distributions are similar.

To better ascertain whether the observed larval particle distributions are statistically significant, the particle final locations were analyzed using a spatial hot spot analysis in ArcMap 10.6.1 (ESRI, Redlands CA). The results of the hot spot analysis indicate if more particles end up in an area than would be expected due to chance. Since the distribution of larval particles is controlled by a combination of system hydrodynamics and swim behavior, some hot spots are expected. Comparison of the hot spot locations between without and with project conditions is used to analyze if the hot spots change due to the deepening of the channel. Figure 5.4 shows some changes in the distribution of the hot spots may be due to the channel deepening; however, the overall pattern of hot spots, where more particles than expected by chance occur, are similar between without and with project conditions.

To add more granularity to the analysis, the results of the hot spot analysis at each reef location were analyzed without and with project to see if the proposed channel deepening affected the likelihood of a reef being a hot spot or not. Note that since not all larval sources are simulated, there may be more hot spots within Mobile Bay that are not represented in these results. Of the 18 reefs that were included in the analysis, seven are predicted to be particle settlement hot spots (>50% of the area identified as a hot spot at α=0.05) under without project conditions and 8 are predicted to be particle settlement hot spots with the deepening (Table 5.3). A greater proportion of Kings Bayou Reef and Whitehouse/Denton Reef were identified as hot spots for larval settlement in the with-project condition than the without-project condition. While Shell Banks Reef was not identified as a hot spot based on the >50% of the total area criteria, 13.2% of the reef area was located in a hot spot region under without-project conditions. Under with project conditions the location of the hot spot in that region shifted so that Shell Banks reef was no longer in the hot spot. This does not indicate that larva are not settling on Shell Banks reef, but that the criteria for classifying it as a hot spot were not met. Of the 3262 acres of oyster reef included in the analysis, 2700 acres were identified as larval settlement hot spots under without project conditions and 2761 acres were identified as larval settlement hot sport with project conditions.

As the model did not explicitly account for particle settling, additional analysis of the larval trajectory data were used to determine if competent larvae passed over reef areas before finally settling. ArcMap 10.6.1 was used to visualize the particle trajectories 10 days from release to the end of the simulation and the line density of the trajectories was mapped. Like the final larval particle location point density maps, the overall pattern of line density was similar between without and with project conditions, indicating that a similar density of competent larvae passed over reef areas (Figure 5.5). The Warren similarity index between the line density maps was 0.980, indicating a high degree of similarity between the two datasets.

Final particle locations were used to quantify the number of particles being flushed out of the system. Results indicate a similar number of particles were flushed from the Mobile Bay system with the project than without. Although fewer particles were flushed under with project conditions, the total number were similar and it is unclear if the difference was due to the project or the probabilistic nature of the model. Breaking down the number of particles flushed by the spawning reef produced similar results (Table 5.4).

The larval trajectories without and with project were compared with monthly water quality model outputs at all water depths to determine if a simulated larval particle was exposed to adverse salinity or DO conditions before settling. The limiting conditions are summarized in

Table 5.1, but briefly, exposure to DO < 2.4 mg/L or salinity values less than 6 ppt or greater than 37 ppt for over 10,000 seconds resulted in mortality. Although the water quality model results show minimum DO concentrations as low as 3.3-3.7 mg/L during some months (i.e., August through November), these conditions did not exceed the mortality threshold value of < 2.4 mg/L at any time during the simulated larval releases or the full spawning season. Consequently, no mortality due to low DO was indicated (Figure 5.6 and 5.7). The average differences between the with and without project DO conditions across the model domain ranged from 0.02 to 0.06 mg/L with standard deviations ranging from 0.08 to 0.22 mg/L which is considered within the range of modeling error. Given the small magnitude in DO differences attributable to the project, larval mortality during future instances when DO could exceed the mortality threshold should likewise be similar between without and with project conditions.

Some of the simulated salinity values were lower than the oyster larvae threshold of 6 ppt (Figure 5.8), especially during the April and June releases. Simulated oyster larvae experienced mortality due to spending more than 10,000 seconds in low salinity zones (Table 5.5). Overall, there was a 33% loss in oyster larvae due to low salinity values in the without project condition and a 28% loss in the with project scenario (Table 5.5).

<u>Communication</u>

The results from the oyster model are intended to be presented to an audience with a general technical background, particularly environmental planners, operations personnel, and natural resource managers. Results should facilitate a deeper understanding of the relative impact of project alternatives on inter-reef dynamics of oyster larvae in Mobile Bay.

5.4 References

Carnegie, R. B. & E. M. Burreson. 2011. Declining impact of an introduced pathogen: *Haplosporidium nelsoni* in the oyster *Crassostrea virginica* in Chesapeake Bay. *Mar. Ecol. Prog. Ser.* 432:1–15

Cerco, C. F. and M. R. Noel. 2004. The 2002 Chesapeake Bay Eutrophication Model. EPA 903-R-04-004 July 2004 U.S. Army Engineer Research and Development Center.

Chapman, R. and P.V. Luong. 2009. Development of a Multi-Block CH3D with a Wetting, Drying, and CLEAR Linkage Capability. Report prepared for Louisiana Coastal Area (LCA) Ecosystem Restoration Plan S&T Office, Vicksburg, MS.

Cheng, Hwai-Ping. Farthing, Matthew W. Winters, Kevin D. Howington, Stacy E. Cheng, Jing-Ru C., 1963- Hines, Amanda M. PT123 a multi-dimensional particle tracking computer program: version 1.0. Report Number ERDC TR-11-10, October 2011, Information Technology Laboratory (U.S.), System-Wide Water Resources Program (U.S.), Coastal and Hydraulics Laboratory (U.S.), Engineer Research and Development Center (U.S.).

Dekshenieks, M. M., Hofmann, E. E., Klinck, J. M., & Powell, E. N. (1996). Modeling the vertical distribution of oyster larvae in response to environmental conditions. *Marine Ecology Progress Series*, 136, 97-110.

ESRI 2018. ArcGIS Desktop: Release 10.6.1 Redlands, CA: Environmental Systems Research Institute.

Gunter, G. 1955. Mortality of oysters and abundance of certain associates as related to salinity. *Ecology* 36:601–605.

Kennedy, V. S., R. I. E. Newell & A. F. Eble, editors. 2009. The eastern oyster *Crassostrea virginica*. College Park, MD: Maryland Sea Grant. 734 pp.

Kim, C. K., Park, K., Powers, S. P., Graham, W. M., & Bayha, K. M. (2010). Oyster larval transport in coastal Alabama: Dominance of physical transport over biological behavior in a shallow estuary. *Journal of Geophysical Research: Oceans*, *115*(C10).

Kjelland, Michael E., Candice D. Piercy, Tahirih Lackey, Todd M. Swannack. 2015. An integrated modeling approach for elucidating the effects of different management strategies on Chesapeake Bay oyster metapopulation dynamics. *Ecological Modelling* 308: 45-62.

Levinton, J., M. Doall, D. Ralston,A. Starke & B. Allam. 2011. Climate change, precipitation and impacts on an estuarine refuge from disease. *PLoS One* 6:e18849.

North, E.W., Schlag, Z., Hood, R., Zhong, L., Li, M., Gross, T. 2006. Modeling dispersal of *Crassostrea ariakensis* oyster larvae in Chesapeake Bay. In: Final report to Maryland Department of Natural Resources. Maryland Department of Natural Resources (55 p.).

North, E. W., Z. Schlag, R. R. Hood, M. Li, L. Zhong, T. Gross, V. S. Kennedy. 2008. "Vertical swimming behavior influences the dispersal of simulated oyster larvae in a coupled particle-tracking and hydrodynamic model of Chesapeake Bay. *Marine Ecology Progress Series* 359: 99-115. doi: 10.3354/meps07317

Tate, J., Savant, G., McVan, D. 2012. Rapid response numerical modeling of the 2010 Pakistan flooding. *Leadersh. Manage. Eng.* 12(4): 315-323.

VanDerWal, J., L. Falconi, S. Januchowski, L. Shoo, and C. Storlie. 2014. SDMTools: Species Distribution Modelling Tools: Tools for processing data associated with species distribution modelling exercises. R-package version 1.1-221. https://cran.r-project.org/web/packages/SDMTools/index.html

Warren, D. L., Glor, R. E., & Turelli, M. (2008). Environmental niche equivalency versus conservatism: quantitative approaches to niche evolution. *Evolution: International Journal of Organic Evolution*, 62(11), 2868-2883.

Table 5.1. Overview of oyster model components including: input variables and environmental parameters

PARAMETER	VALUE (Status/Unit of measure)
Spatial scale	42,868 cells
Adaptive time step	Seconds (s)
Length of simulation	April through September
Initial oyster larvae	5400 particles
Depth (# of layers)	Averaged to 3 layers
Low Dissolved Oxygen (DO) threshold	2.4 mg/l
High Dissolved Oxygen (DO) threshold	N/A
Low Salinity threshold	6 ppt
High Salinity threshold	37 ppt
DO mortality threshold duration	10,000 s to live outside threshold
Salinity mortality threshold duration	10,000 s to live outside threshold
Temperature mortality threshold duration	10,000 s to live outside threshold

Table 5.2. Source reef locations used as release points for oyster larvae simulations. Area, geographic coordinates and percent contribution of each reef to overall recruitment within Mobile Bay are also included.

NAME	ACRES	Latitude	Longitude	Percent Contribution
Area VI Natural	17.9	30.46253	-88.09981	0.5
Bender-Austal Reef	3.21	30.52829586	-88.04907171	0.5
Bon Secour Reef	30.7	30.29039975	-87.77714232	0.5
Brookley Reef	88.4	30.60005312	-88.04149181	3.5
Buoy Reef A	212.0	30.32572768	-88.11199004	5
Cedar Point East 2014 Plant	585.4	30.31660719	-88.13051762	11
Cedar Point East Bridge	292.8	30.30077478	-88.1327984	11
Cedar Point Gullies	637.4	30.29746944	-88.1399407	11
Cedar Point Pass-aux-Huite	373.8	30.30730306	-88.13873889	11
Fish River Reef	109.1	30.32822442	-87.83445264	0.5
Heron Bay Cedar Point Beach	497.9	30.3141351	-88.14204291	14
Heron Bay Pass-aux-Bar	264.1	30.32325461	-88.15540528	14
Kings Bayou Reef	66.1	30.34213812	-88.10862831	0.5
Klondike Reef	166.2	30.4511418	-87.93237655	0.5
Portersville Bay Hard Reef	35.4	30.35057894	-88.23129115	0.5
Portersville Bay Middle Ground	33.5	30.34702079	-88.207512	0.5
Shell Banks Reef	155.6	30.25970518	-87.85898376	0.5
Whitehouse/Denton Reef	70.6	30.41160401	-88.06768474	0.5

Table 5.3 Results of the hot spot analysis at each reef location were analyzed without and with project to see if the proposed channel deepening affected the likelihood of a reef being a hot spot. Note all Cedar Point reefs are combined for the purpose of this analysis.

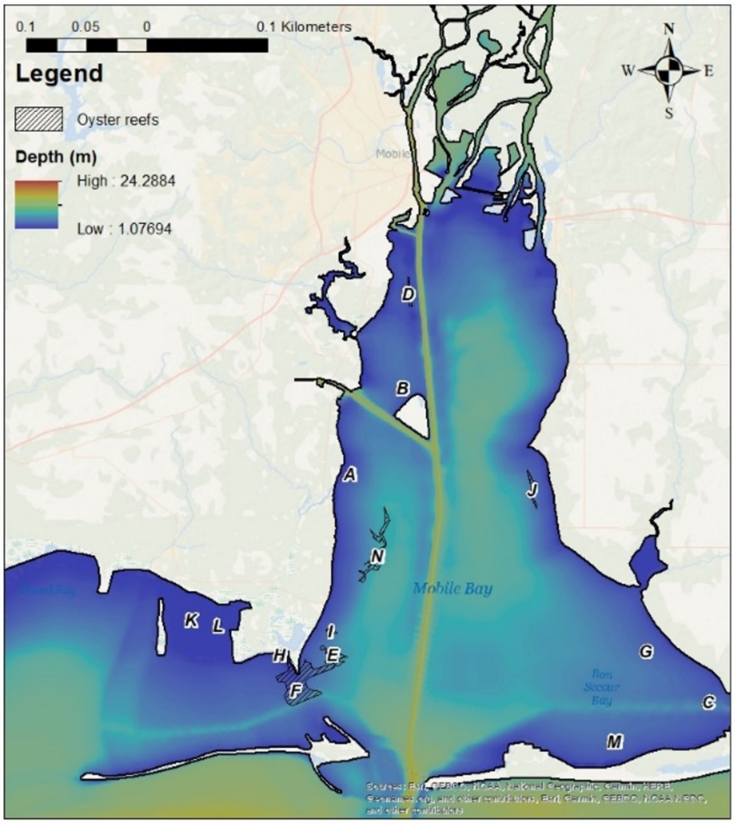
Reef name	Label	Total Reef Area (acres)	Proportion of reef identified as hot spot		Area of reef identified as hot spot	
			Without	With	Without	With
Area VI Natural	A	17.9	0.0%	0.0%	0.0	0.0
Bender-Austal Reef	B	3.2	0.0%	0.0%	0.0	0.0
Bon Secour Reef	C	30.7	100.0%	100.0%	30.7	30.7
Brookley Reef	D	88.4	0.0%	0.0%	0.0	0.0
Buoy Reef A	E	212.0	99.4%	95.7%	210.7	203.0
Cedar Point - all	F	2009.2	97.9%	99.6%	1966.6	2001.8
Fish River Reef	G	109.1	100.0%	100.0%	109.1	109.1
Heron Bay Pass-aux-Bar	H	264.1	100.0%	100.0%	264.1	264.1
Kings Bayou Reef	I	66.1	43.4%	100.0%	28.7	66.1
Klondike Reef	J	166.2	0.0%	0.0%	0.0	0.0
Portersville Bay Hard Reef	K	35.4	100.0%	100.0%	35.4	35.4
Portersville Bay Middle Ground	L	33.5	100.0%	100.0%	33.5	33.5
Shell Banks Reef	M	155.6	13.2%	0.0%	20.5	0.0
Whitehouse/Denton Reef	N	70.6	0.7%	24.2%	0.5	17.1
Total		**3262**			**2700**	**2761**

Table 5.4. Simulated number of particles were flushed from the Mobile Bay system for the without and with project scenarios for each of the simulated reefs.

	Without Project		With Project	
Reef Name	Particles flushed	% flushed	Particles flushed	% flushed
Area VI Natural	1	0%	1	0%
Bender-Austal Reef	40	13%	46	15%
Bon Secour Reef	85	28%	58	19%
Brookley Reef	26	9%	19	6%
Buoy Reef A	124	41%	129	43%
Cedar Point East 2014 Plant	86	29%	69	23%
Cedar Point East Bridge	103	34%	82	27%
Cedar Point Gullies	157	52%	111	37%
Cedar Point Pass-aux Huite	76	25%	79	26%
Fish River Reef	9	3%	1	0%
Heron Bay Cedar Point Beach	0	0%	0	0%
Heron Bay Pass-aux-Bar	0	0%	0	0%
King Bayou Reef	107	36%	108	36%
Klondike Reef	7	2%	12	4%
Portersville Bay Hard Reef	62	21%	27	9%
Portersville Bay Middle Ground	3	1%	0	0%
Shell Banks Reef	164	55%	170	57%
Whitehouse/Denton Reef	5	2%	62	21%
Total	1055	20%	974	18%

Table 5.5. Simulated number of oyster larvae exposed to simulated salinity values less than 6 ppt for 10,000 seconds. Overall a there was a 5% reduction in simulated larval particles being exposed to adverse salinities during the time period simulated.

	Without Project			With Project		
Release Date	April	June	August	April	June	August
# of Dead Particles	675	765	318	602	657	277
Total			1758			1536
Percentage of overall total released			33%			28%



Label	Reef name
A	Area VI Natural
B	Bender-Austal Reef
C	Bon Secour Reef
D	Brookley Reef
E	Buoy Reef A
F	Cedar Point - all
G	Fish River Reef
H	Heron Bay Pass-aux-Bar
I	Kings Bayou Reef
J	Klondike Reef
K	Portersville Bay Hard Reef
L	Portersville Bay Middle Ground
M	Shell Banks Reef
N	Whitehouse/Denton Reef

Figure 5.1. Map of the reef system used for modeling oyster larval dispersal in Mobile Bay. Note Cedar Point reef encompasses all five Cedar Point reef areas.

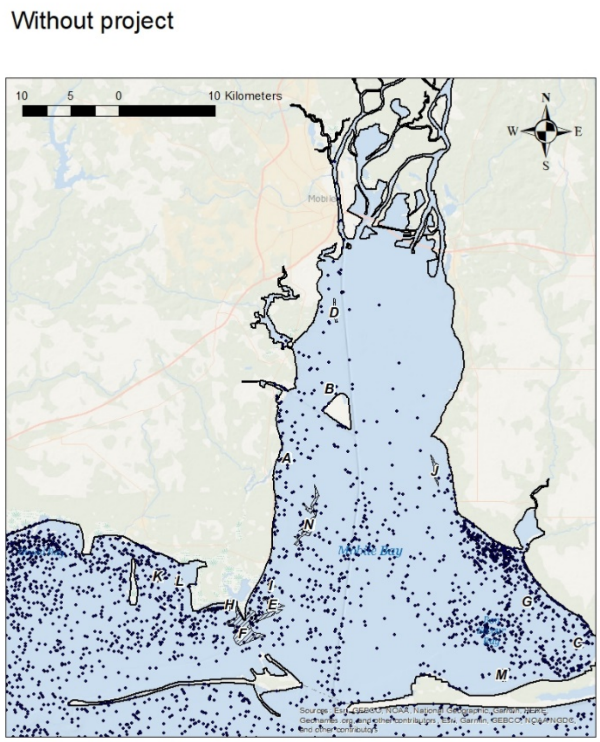


Figure 5.2. Final location of simulated oyster larvae for the without (left) and with (right) project scenarios.

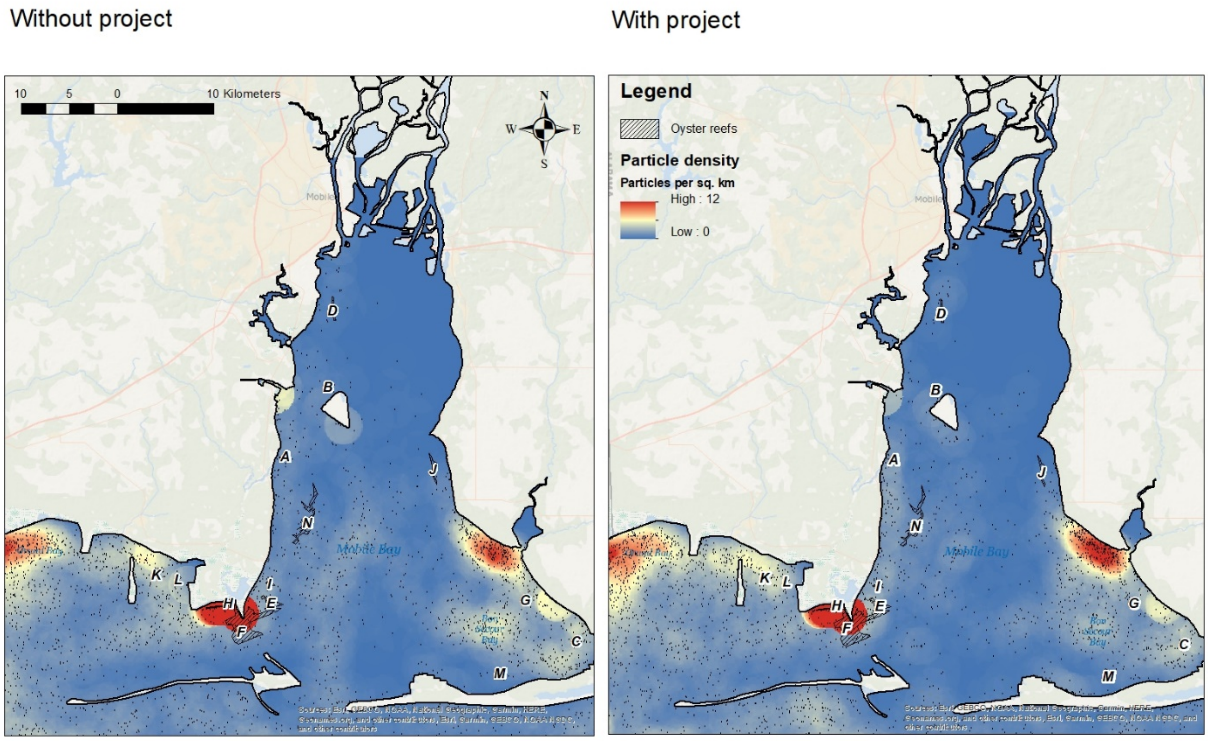


Figure 5.3. Point density heat map representing oyster densities of final locations of simulated oyster larvae for the without (left) and with (right) project scenarios. Comparing the without and with project point density maps, the Warren similarity index value is 0.977 (indicating particle distributions are similar).

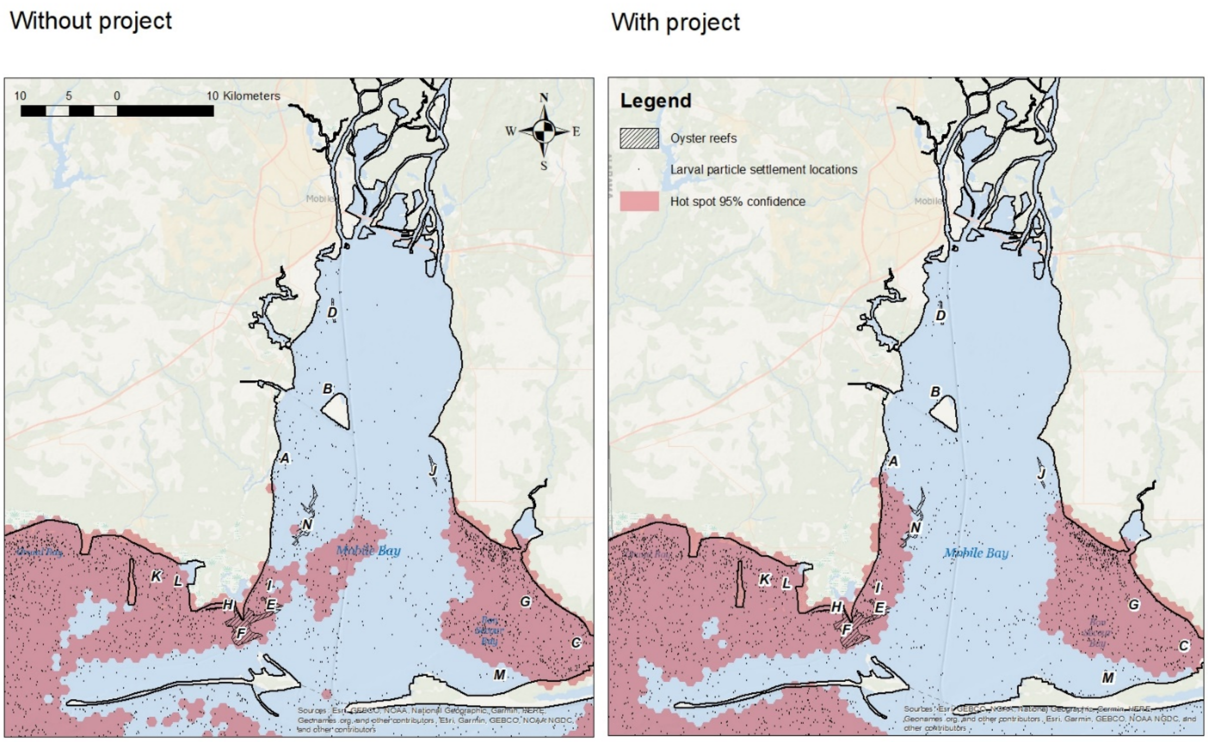


Figure 5.4. Results of hotspot analysis indicating if more particles end up in an area than would be expected due to chance for the without (left) and with (right) project scenarios. Red shaded areas represent the locations that were considered hotspots with 95% confidence.

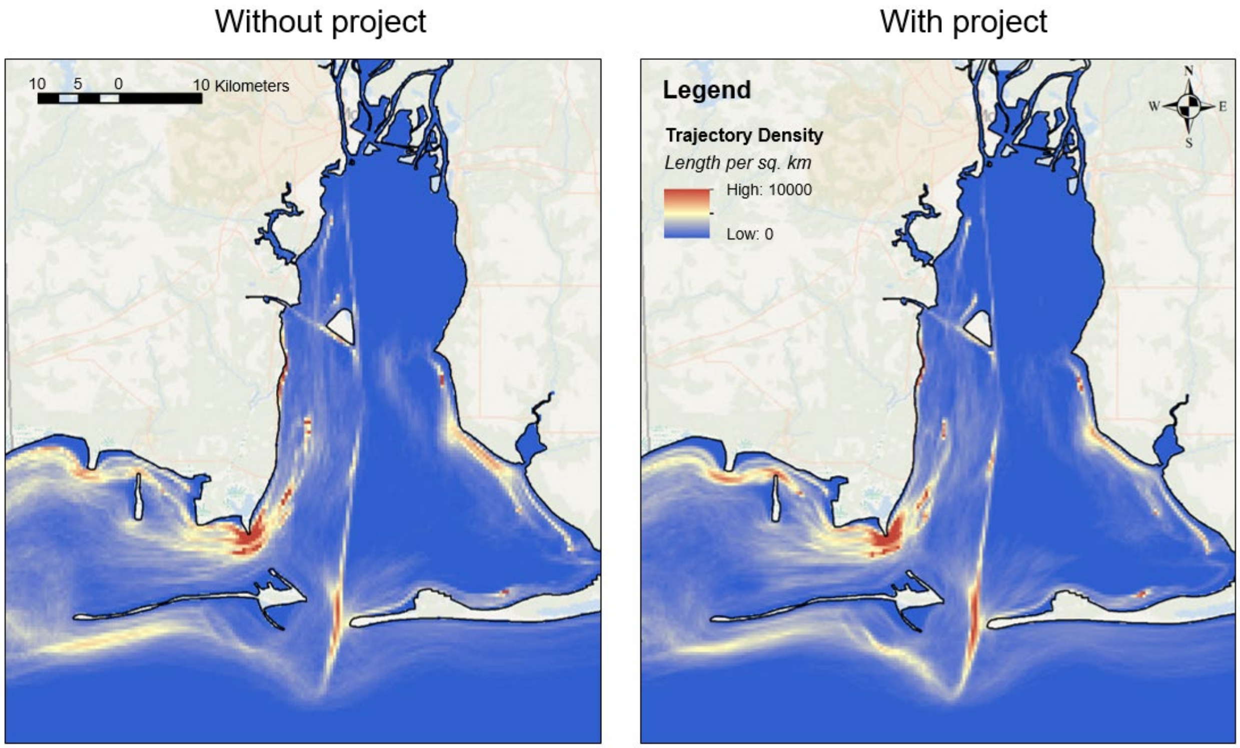


Figure 5.5. Line density heat map representing line densities of simulated oyster larvae trajectories for the without (left) and with (right) project scenarios. Comparing the without and with project line density maps, the Warren similarity index value is 0.980 (indicating trajectory line distributions are similar).

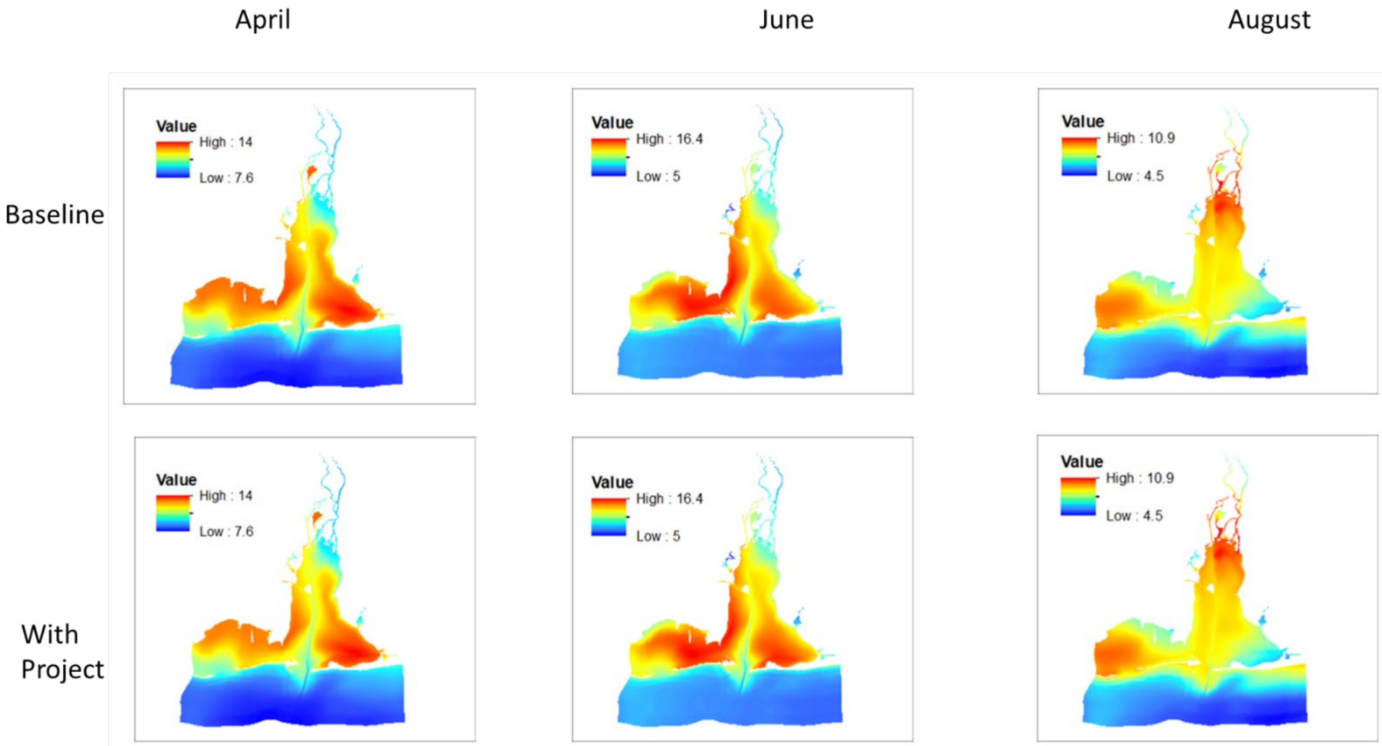


Figure 5.6. Simulated average dissolved oxygen levels for Mobile Bay during the months of simulated oyster larval releases. Average dissolved oxygen levels were not below larval tolerance of < 2.4 ppm during any of the simulated months.

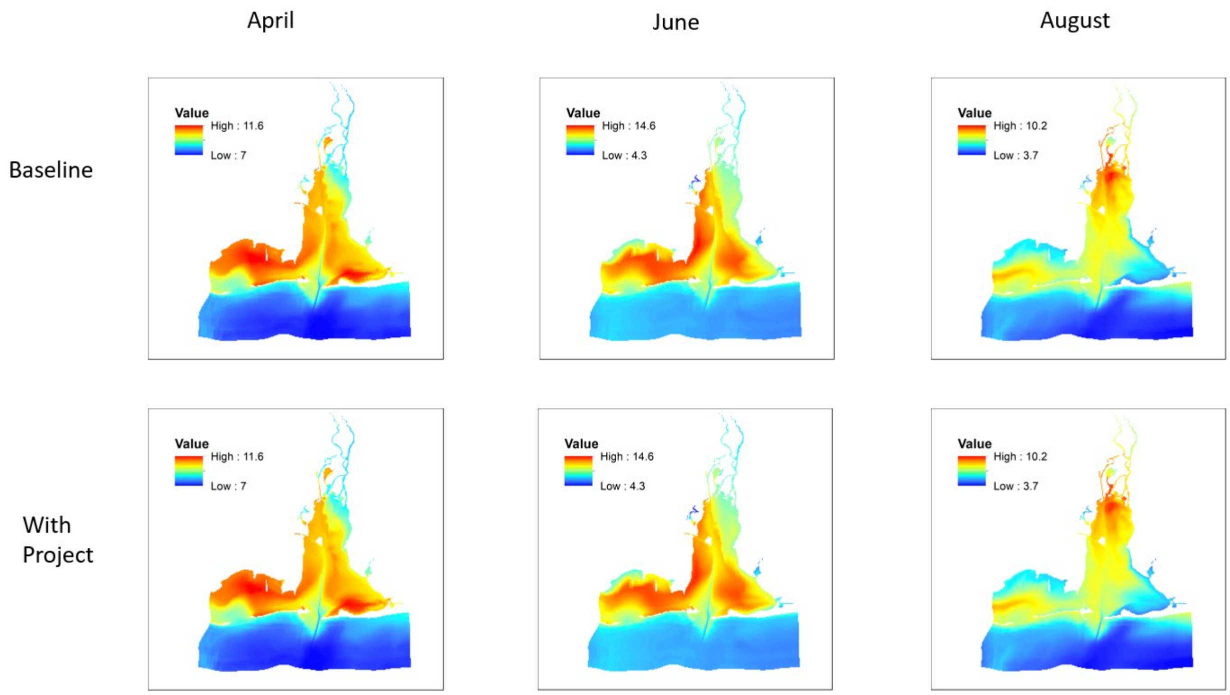


Figure 5.7. Simulated lowest 1% of dissolved oxygen levels for Mobile Bay during the months of simulated oyster larval releases. The lowest 1% of dissolved oxygen levels were not below larval tolerance of < 2.4 ppm during any of the simulated months.

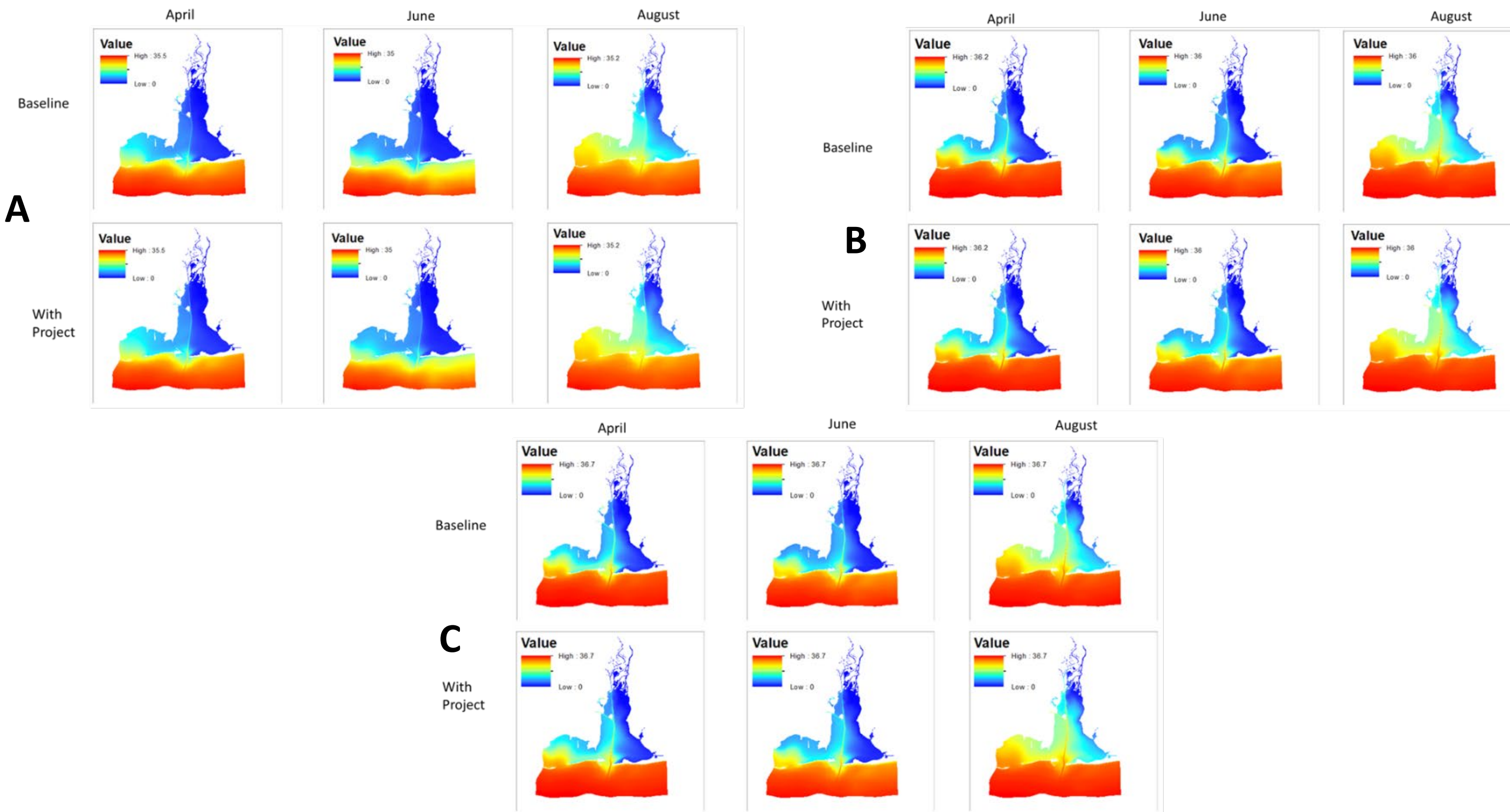


Figure 5.6. Simulated salinity values for the (A) top third of layers, (B) mean values, and (C) bottom third of layers from the water quality modeling for each of the three months simulated for oyster larvae dispersal. We considered mortality to occur if oyster larva were exposed to salinities < 6 ppt or > 37 ppt for longer than 10,000 s.

Chapter 6 - Fishery Assessment

Summary

An analysis of potential fishery-related impacts of deepening Mobile Harbor was conducted using data collected in 2016-17 by ERDC and Fisheries Assessment and Monitoring Program (FAMP) database (seine and trawl) collected by the Alabama Department of Marine Resources from 2000-2015. The principal objective was to develop statistical relationships between salinity and fish assemblage structure to establish baseline conditions and evaluate impacts of the project. A total of 2,097,836 individuals representing 162 species were recorded and used in the analysis. Mean abundance was calculated from the overall database for salinity tolerance guilds of the Mobile Bay fish community and included freshwater only, freshwater entering estuary, resident estuary, marine entering estuary, and marine only. Quantile regression was used to calculate statistical relationships between salinity and guild abundance to identify those guilds most susceptible to changes in salinity due to project effects. Two of the guilds showed a narrow range of salinity tolerance: Marine only between approximately 20-33 ppt and freshwater only less than 5 ppt. However, both of these guilds were rarely collected in the Mobile Bay. The three other guilds had a much wider range of salinity utilization suggesting that major changes in salinity were necessary to impact these groups of species. Modelled changes in salinity between baseline and post-project with and without sea level rise ranged from -1.0 to 6.0 ppt with an average of approximately 2.0 ppt. Small changes in salinity indicates that impacts to the Mobile Bay fishery are not expected. The freshwater entering estuary guild is likely the most susceptible to changes in salinity due to project construction, but the range they occupy suggests that salinity differences between baseline and post-project would not impact survival of the Mobile Bay fish community.

6.1 Introduction

Mobile Bay occurs in southwestern Alabama and extends 31 miles from the mouth of the freshwater Mobile-Tensaw River Delta south to its outlet into the Gulf of Mexico. It is one of the largest estuaries in the Gulf of Mexico, draining 70,267 square miles (Mullins et al. 2002). The width of the bay ranges from 8 miles near the mouth of the Mobile River to a maximum of 24 miles where it connects to the intercoastal waterway and Gulf of Mexico. Mobile Bay is relatively shallow with an average depth of approximately 10 feet with daily tide changes averaging 1.6 feet (Mullins et al 2002). The deepest areas of the Bay occur within the shipping channel maintained at 45 feet deep by USACE and can exceed 75 feet at some locations.

Mobile Bay ranks first in in the number of freshwater species in the Southeastern Atlantic and Gulf of Mexico drainages, with a total of 157 species recorded, 40 of which are endemic (Swift et al 1986). Long-term collections in Mobile Bay estuary by the Alabama Marine Resource Division, catalogued in the Fisheries Assessment and Monitoring Program (FAMP) database, list 140 species of estuarine fishes. Mobile Bay is also an important shrimp fishery in the Gulf of Mexico with average monthly harvests approaching 100,000 pounds from August to October (Loesch 1976). High biodiversity reflects the ecological importance of this drainage network, including inflows from the Black-Warrior, Tombigbee, and Alabama Rivers. Habitat complexity in the Bay, including seagrass beds, dunes and interdune wetland swales, saltwater marshes, freshwater wetlands, and bottomland hardwood forests, directly maintains this high biodiversity (Rashleigh et al. 2009).

An interesting phenomenon that occurs in Mobile Bay is referred to as a "jubilee." First reported by Loesch (1960) and later evaluated by May (1973), jubilees occur in the summer and fall when water becomes anoxic due to decaying plankton blooms and aquatic vegetation driving fish and shellfish towards the shore where oxygen is higher. Aquatic fauna become trapped between the shore and the anoxic water where they are easily harvested. Park et al (2007) further explained that Mobile Bay hypoxia is associated with a large oxygen demand during destratification events, can reoccur within hours to days depending on time of year, and has been identified a one of the priority areas of concern (Rabalais et al 1985). Other impairments to Mobile Bay include erosion, loss of emergent wetlands due to industrial, navigational, and urban development, dredging, and nonpoint source pollution (Roach et al.1987; Duke and Kruczynski 1992).

The ecological importance of Mobile Bay necessitates a complete evaluation of future water resource projects. The Water Resources Development Act of 1986 authorized USACE to deepen the Mobile Harbor as follows: deepening and widening of the entrance channel to 57 feet by 700 feet, and deepening and widening of the Mobile Bay channel from the mouth to south of Mobile River to 55 feet by 550 feet, for a total of 27 miles; deepening and widening an additional 4.2

miles of the Mobile Bay channel to 55 feet by 650 feet; and a 55-foot deep anchorage and turning basin in the vicinity of Little Sand Island. Portions of the authorized project have been constructed including deepening of the entrance channel to 47 feet by 600 feet and extending the upper channel by 4,600 feet to a depth of 45 feet. Changes in depth may alter salinity patterns in the surrounding estuarine ecosystem and impact fish and other faunal groups. The objectives of the fishery assessment was to establish baseline conditions in the project area including species distribution and abundance, and evaluate relationships between salinity and fish assemblage structure to predict potential environmental impacts on this resource.

6.2 Methods

Fish were collected during September 2016 to evaluate recruitment and growth and May 2017 to evaluate the spawning period and young-of-year survival. The purpose of these collections were to establish baseline conditions and become familiar with the project area. ERDC conducted sampling in the freshwater, transition and upper bay zones for a total of 11 sites utilizing the same gear and protocol as with the Fisheries Assessment and Monitoring Program (FAMP) database (seine and trawl). The sampling efforts in the upper bay zone were conducted to provide complementary data in that zone and to also aide in calibrating efforts in the transition and freshwater zones with comparable efforts in the remaining zones. Data used for the fishery analysis encompassed 2000-2015, and ERDC data collected in 2016 and 2017.

A map depicting the sample station distribution (overall map with two insets) was created that illustrates the FAMP stations historically and currently sampled by Alabama Marine Resources Division (1981-present) as well as the location of the ERDC samples. The inclusion of all FAMP data provides a visual aide supporting the breadth of geographic coverage represented by the data. However, despite the broad geographic coverage represented by their database, only stations were located within the footprint of the model grid to be used as snapshots of modeled environmental parameters within the project area were included (Figure 6.1).

<u>Physical Model</u>

All sample stations (ERDC and FAMP) were plotted in ArcMap with the addition of a 500 m buffer to capture the variability in environmental conditions for any given sample. For the ERDC samples, the buffer around the entire length of each trawl sample was included to capture the habitat variability associated with each effort. The model grid layer were then added to the ArcMap project for each modeled environmental parameter: bottom and mean salinity (with and without sea level rise) and bottom and mean dissolved oxygen (without sea level rise). The intersecting cells from the respective model grid and the station buffer layer were extracted for evaluation of project impacts (Figure 6.2).

The initial model output provided for use for the fisheries assessment included modeled baseline conditions, with project conditions and the numerical difference (change) between baseline and project values. Basic summary statistics were generated (i.e., mean, minimum, maximum, standard deviation, percentile) for each modeled cell within the grid and for each respective condition. The MAX-DIFF value (maximum value of difference between baseline and project values per cell) was utilized to evaluate potential project impacts. This parameter was selected to illustrate a worst case scenario with regard to changes in salinity and dissolved oxygen due to the project.

Fish Model

Fish were collected by trawling and seining. A two-seam, 16-ft otter trawl was used to sample benthic fish over a range of water depths. A total of 2-5 trawl samples were taken at each site. The body of the trawl was made of 1⅜-inch webbing and the cod end liner was 3/16-inch mesh to retain smaller bodied individuals. Trawling occurred in water depths ranging from 5 to over 30 ft. The length of the tow lines were about three-times the water depth to ensure that the footrope of the trawl remained along the bottom. A tickler chain was attached to the footrope to disrupt the substrate and increase catch efficiency of benthic organisms. The net was deployed from the bow followed by the otter boards as the boat slowly backed up. Any twists or crossing of the ropes were corrected during deployment. A float line was tied to the cod end in case the trawl became entangled on underwater obstructions. If entangled, a trailer boat grabbed the float line and slowly backed up lifting the trawl from the obstruction; the sample was usually discarded. A

GPS recorded average speed and distance travelled during a 10-minute trawl sample, which was the duration used for the FAMP data. The trawl was retrieved after completion of the sample and contents of the cod end was emptied into a sorting container.

A 50 x 4 ft., 3/16-inch mesh knotless bag seine was used to sample shoreline fish and shellfish. One seine haul was taken per site, which was the same effort used for the FAMP data. Two people carried the seine out from the shoreline 60-ft, then moved parallel to the shore a short distance to avoid disrupting the sample area. The 60-ft distance was confirmed by a person with a range finder standing along the shoreline. The seine was unfurled and hauled towards the shoreline ensuring that the lead line was in full contact with the substrate. In structurally-complex areas (e.g., vegetation), a third person was located behind the mid-section of the seine in case the lead line became entangled on a snag. If entangled, the third person reached down and pulled back the lead line usually freeing the net from the snag. If the seine was readily freed, the sample was discarded and an adjacent site was sampled. Once the shoreline had been reached by the seiners, the wings of the seine was shaken down until all organisms are in the bag area where they were removed.

All organisms collected by trawl and seine were identified to species or the lowest practical taxon, enumerated, and measured. Large-bodied fish and shellfish were released at the point of capture after processing. Smaller bodied fish, shellfish, and other invertebrates were preserved in 10% formaldehyde and processed in the laboratory. A label was placed in each sample container including location, date, and sample number. Total length was measured for all fish. Carapace or disc width were measured for crabs, anemone, and other shellfish. Mantle length was measured for squids.

Physical and water quality habitat measurements were taken in conjunction with fishery collections at each site. A GPS location was recorded at each sampling site. Surface and bottom water quality were measured using a calibrated YSI multi-parameter meter and included temperature, pH, conductivity, salinity, and dissolved oxygen. Depth was recorded from boat-mounted transducers, and surface velocity was measured using a Marsh-McBirney flow meter. Substrate type (i.e., sand or mud/silt) was visually assessed from otter boards or using a stadia rod to probe the bottom.

All data, including FAMP from 2000-2005 and ERDC from 2016-17, were transferred to Excel spreadsheets, analyzed using the Statistical Analysis System 9.4, and all models were developed from this database. Salinity tolerance for project alternatives was the principal focus of the analysis. Salinity tolerance guilds of the fish community in Mobile Bay study areas were identified according to the Gulf Coastal Research Laboratory publication by Christmas (1973) following the recommendations by Elliott et al (2007). Guilds included: freshwater only, freshwater entering estuary, resident estuary, marine entering estuary, and marine only. Guilds representing species that are anadromous, catadromous, and freshwater introduced were not included. Mean abundance by guild was calculated prior to curve fitting techniques in SAS 9.4 (SAS 2013. Abundance was log transformed ($\log_{10}$ +1) to account for outliers and skewed data to approximate normality.

The physical water quality model developed by ERDC was used to predict changes in salinity gradients for baseline and alternatives, and the biological models developed from the FAMP and ERDC field data were compared to predicted changes in salinity. Biological relationships between salinity and guild abundance were evaluated using quantile regression using the sparsity method for confidence limits (SAS 2013). Species abundance-habitat relationships are typically skewed with zero-inflated count data, contains outliers, and does not meet the assumptions of normality required for linear regression (Terrell et al. 1996; Vaz et al 2008). Quantile regression is a non-parametric method of modeling response variables when assumptions of ordinary least squares regression are not met. It estimates multiple rates of change (slopes) from the minimum to maximum response, providing a more complete picture of the relationships between variables missed by other regression methods (Cade and Noon 2003). The 0.90 regression quantile was considered in model development, which represents the upper bounds of species–environment relationships and thus estimates how the environment is limiting the distribution of a species (Vaz et al. 2008). Diagnostic options in SAS 9.4 were utilized for the analysis.

6.3 Results and Discussion

Physical Model

Extracted cells from the model grid based on the intersect with the station buffer GIS layers ranged 132,216 – 159,801 cells per run depending on the chose environmental parameter (salinity, dissolved oxygen), parameter status (mean, bottom) and project condition (with/without sea level rise). The MAX-DIFF values for mean salinity without sea level rise ranged -2.0 to 5.8 with a mean value of 0.9 (95% CI: 0.003) and a median value of 1.0. Bottom salinity for the same condition had similar values (range: -1.6 to 5.8; mean: 0.6 (95% CI: 0.003); median: 0.6) although modeled mean salinity exhibited a greater range in values, the largest proportion were within the 0-2 MAX-DIFF range (Figure 6.3).

Figure 6.4 illustrates the seasonal variability in modeled output at each sample station for mean salinity without sea level rise. Some stations illustrate a wide range of salinity conditions through a typical water year; other vary less implying some underlying geographic pattern (e.g., transition, upper, middle or lower bay). However, the overwhelming majority of the values for mean salinity are below the 2 ppt difference between baseline and sea level rise suggesting little concern for impact. Those values exceeding 3 ppt were projected for January – May and were associated primarily with Little Sand Island adjacent to the current shipping channel. A similar pattern was exhibited for bottom salinity (without sea level rise) (Figure 6.5.) with few stations exceeding the 3 ppt salinity differential.

Salinity changes evaluated under the "with sea level rise" condition exhibited a narrower range in MAX-DIFF values for both mean (range: -1.7 to 6.4; mean: 0.9 (95% CI: 0.003); median: 0.9) and bottom salinity (range: -1.5 to 6.2; mean: 0.5 (95% CI: 0.003; median: 0.5) conditions (Figure 6.6). There was a slight reduction in central tendencies of the dataset for both mean (mean: 0.9 vs 0.9; median: 1.0 vs 0.9) and bottom salinity (mean: 0.6 vs 0.5; median: 0.6 vs 0.5) when considering comparisons to values generated under both project conditions (with/without sea level rise. However, the distribution of extracted model values from each condition were not significantly different (mean salinity KS test, $D = 0.18$, $p = 0.2$; bottom salinity KS test, $D = 0.09$, $p = 0.9$) (Figure 6.7, 8) indicating no appreciable differences in salinity values between current conditions and those projected under the sea level rise scenario.

Conditions for dissolved oxygen (without sea level rise) showed a smaller range in variability in the extracted values for both mean (range: -0.9 to 1.0; mean: -0.1 (95% CI: 0.001); median: -0.1) and bottom conditions (range: -0.7 to 2.4; mean: 0.4 (95% CI: 0.003; median: -0.01) compared to responses of salinity under similar conditions. The distribution of extracted values for dissolved oxygen were significantly different (KS test, $D = 0.54$, $p < 0.01$) between mean water column and bottom conditions (Figure 6.9). Bottom conditions experienced less variability with 98% of the MAX-DIFF values occurring between -0.5 and 0.5 indicating little projected change in dissolved oxygen levels for benthic oriented fishes. In contrast, 70% of the MAX-DIFF values for mean water conditions occurred between -0.5 and 0.5. Nearly 29% of the values exceeded the 0.05 mg/L MAX-DIFF condition with 1% exceeding the 2.0 mg/L MAX-DIFF condition. These results suggest overall changes in dissolved oxygen are likely to occur, but the extent of change will likely be minimal and expressed in reduced spatial and/or temporal basis.

<u>Fish Model</u>

Almost 1200 measurements of salinity and dissolved oxygen were taken during fish collections by both Alabama Marine Resources Division and ERDC (Table 6.1). A salinity gradient occurred among zones with the lower bay averaging 23 ppt, the middle bay at 12 ppt, upper bay at 8.9 ppt, transition zone at 3.7 ppt, and the freshwater sites at 0.1 ppt. Mean dissolved oxygen was approximately 7.0 mg/l at all zones. However, hypoxia (<3.0 mg/l) was measured at all zones except for the transition and freshwater zones. Higher dissolved oxygen in these two zones may have been due to the low sample size compared to Mobile Bay.

A total of 2,097,836 individuals representing 162 species were recorded and used in the analysis. Species were classified according to the salinity tolerance guilds (Table 6.2). The most speciose assemblage was represented in the marine entering freshwater guild, indicating the importance of the Mobile Bay to this group of fishes. This guild was dominated by three species comprising 79% of the total number of individuals: Spot, Gulf Menhaden, and Atlantic Croaker. The freshwater estuarine guild was next in number of species (21) with a total of 10,315 individuals. Three species comprised 75% of the total number of individuals: Sailfin Molly, Threadfin Shad, and Blue Catfish. The resident estuarine guild had 20 species comprised of 891,773 individuals, but the Bay Anchovy was overwhelming dominate making up 94% of the total. The freshwater only guild had 13 species dominated by Silverside shiner comprising 94% of the total. However, small sample size at these locations contributed to fewer number of species. The marine only guild had nine species, with Red Snapper comprising 91% of the total.

The relationship between guild abundance and salinity was portrayed as a box and whisker plot (Figure 6.10). To avoid a dominance biased analysis, the following species were not used in the evaluation of salinity: Bay anchovy, Spot, Gulf Menhaden, Atlantic Croaker, Pinfish, Spotfin Mojarra, and Inland Silverside. Two of the guilds showed a narrow range of salinity tolerance: Marine only between approximately 20-33 ppt and freshwater only less than 5 ppt. However, both of these guilds were rarely collected in the Mobile Bay. The three other guilds had a much wider range of salinity utilization suggesting that major changes in salinity were necessary to impact these groups of species.

Quantile regression models were developed seasonally for each guild further supporting the wide tolerance range of most species that occur in Mobile Bay (Figure 6.11). The mean abundance of freshwater entering estuary guild was negatively correlated to salinity, whereas the marine entering estuary and marine only were positively correlated. The resident estuarine model suggested little to no correlation with salinity indicating their overall tolerance and ability to osmoregulate as they move between salinity gradients. Given these relationships, and the physical model results presented previously, impacts to the Mobile Bay fishery are not expected. The freshwater entering estuary guild is likely the most susceptible to changes in salinity due to project construction. However, but the range they occupy suggests that differences in salinity between baseline and project alternative would have to much greater than the physical model suggests; including scenarios that incorporate sea level rise projections.

6.4 References

Christmas, J.Y., (editor). 1973. Cooperative Gulf of Mexico Estuarine Inventory and Study, Mississippi. Gulf Coast Research Laboratory, Mississippi Marine Conservation Commission.

SAS. 2013. Statistical Analysis Software, Version 9.4. SAS Institute Inc., Cary, NC.

Vas, S., Martin, C. S., Eastwood, P. D., Ernande, B., Carpentier, G. J. Meaden, and F. Coppin. 2008. Modelling species distributions using regression quantiles. *Journal of Applied Ecology* 45; 204–217.

Duke, T. W. and W. L. Kruczynski. 1992. Status and trends of emergent and submerged vegetated habitats of the Gul of Mexico. Gulf of Mexico Program, U.S. Environmental Protection Agency, Stennis Space Center, MS, 161 pp.

Elliott, M., A. K. Whitfield, I. C. Potter, S.J. M., Cyrus, D. P., F. G. Nordlie, and T. D. Harrison. 2007. The guild approach to categorizing estuarine fish assemblages: a global review. Fish and Fisheries 8: 241-268.

Loesch, H. C. 1976. Shrimp population densities within Mobile Bay. 1976. Gulf Research Reports (5) 2: 11-16. Retrieved from http://aquila.usm.edu/gcr/bol5/iss2/2

May, E. B. 1973. Excessive oxygen depletion in Mobile Bay, Alabama. Limnology and Oceanography 18: 353-366.

Mullins, M., H. Burch, M. Dardeau, and D. Strum (Eds.). 2002. The Mobile Bay National Estuary Program Comprehensive Conservation and management Plan Volumes 1-3. Mobile Bay National Estuary Program, Mobile, Al. (http://www.mobilebaynep.com/search/results/58761d7b8cc732b36ff51c8128c52830)

Park, K., C. Kim, and W.W. Schroeder. 2007. Temporal variability in summertime bottom hypoxia in shallow areas of Mobile Bay, Alabama. Estuaries and Coasts 30 (1): 54-65.

Rashleigh, B., M. Cyterski, L. M. Smith, and J. A. Nestlerode. 2009. Relation of fish and shellfish distributions to habitat and water quality in the Mobile Bay estuary, USA. Environmental Monitoring and Assessment 150: 181-192.

Roach, E.R., Watzin, M.C., Scurry, J.D., and Johnston, J.B., 1987, Wetland changes in coastal Alabama. Pages 92-101 *in* Lowery, T.A. A(ed). Proceedings of Symposium on the Natural Resources of the Mobile Bay Estuary, Mobile, Ala., February I 0-12, 1987: Mobile, Ala., Auburn University, Alabama Sea Grant Extension Service and Alabama Cooperative Extension Service, p. 92-101.

Swift, C. C., C. R. Gilbert, S. A. Bortone, G. H. Burgess, and R. W. Yerger. 1986. Zoogeography of the freshwater fishes of the southeastern United States: Savannah River to Lake Pontchartrain. Pages 212-266 in C. H. Hocutt and E. O. Wiley, editors. The zoogeography of North American freshwater fishes. John Wiley and Sons, New York.

Terrell, J. W., B. S. Cade, J. Carpenter, and J. M. Thompson. 1996. Modeling stream fish habitat limitations from wedged-shaped patterns of variation in standing stock. Transactions of the American Fisheries Society 125: 104-117.

Acknowledgements

We thank John Mareska with the Alabama Department of Marine Resources for providing the FAMP data and useful suggestions its application for this report. Field work was conducted by

ERDC Fish Ecology Team including Jay Collins, Steven George, Alan Katzenmeyer, and Bradley Lewis. Laruen Leonard entered the ERDC data and assisted in formatting the FAMP data used for this analysis.

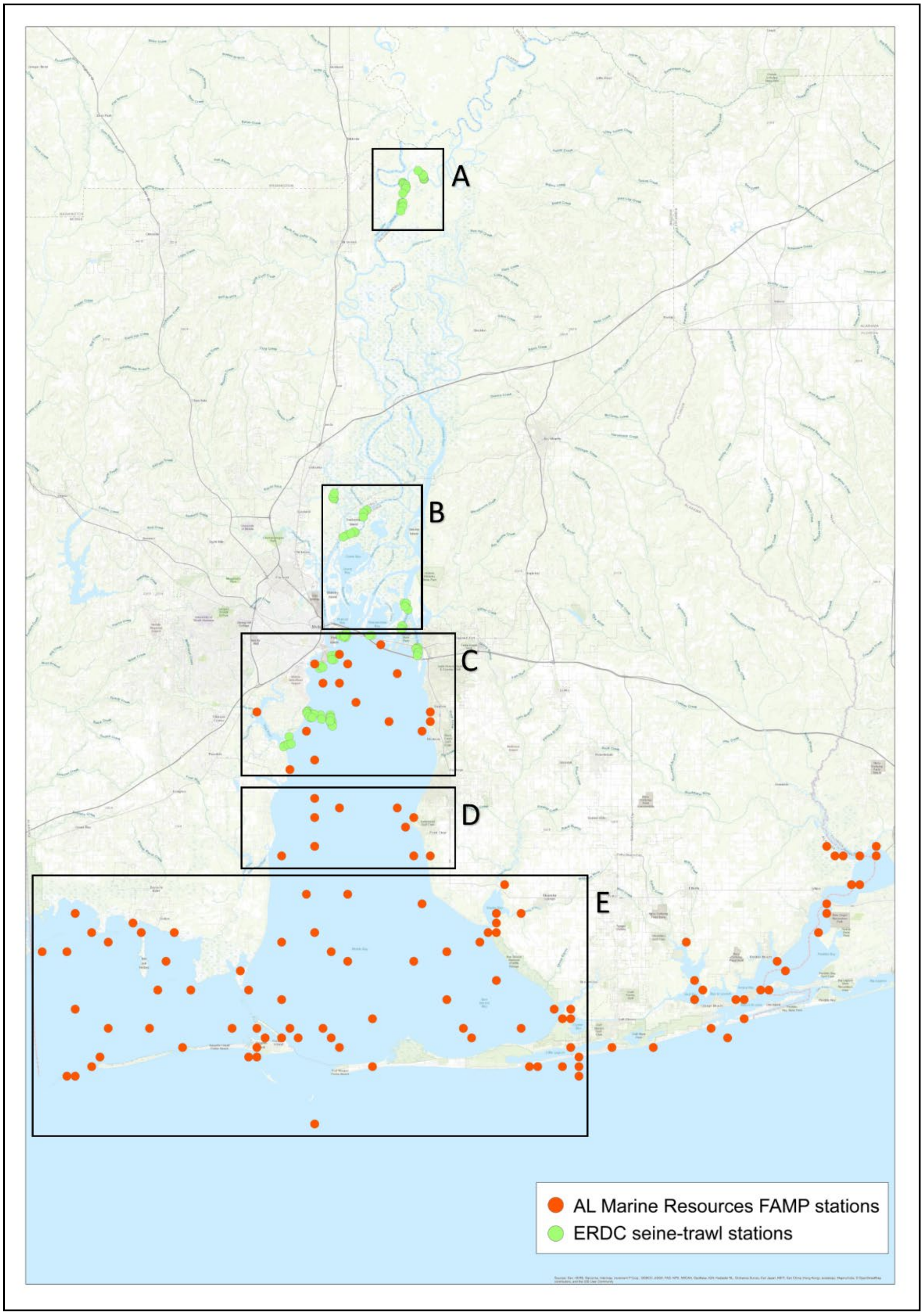


Figure 6.1. Distribution of ERDC sample stations (green) and Alabama Marine Resources FAMP stations (red) utilized for fisheries assessment. Zones within the project area are coded as freshwater (A), transition (B), estuarine-upper bay (C), middle bay (D) and lower bay (E).

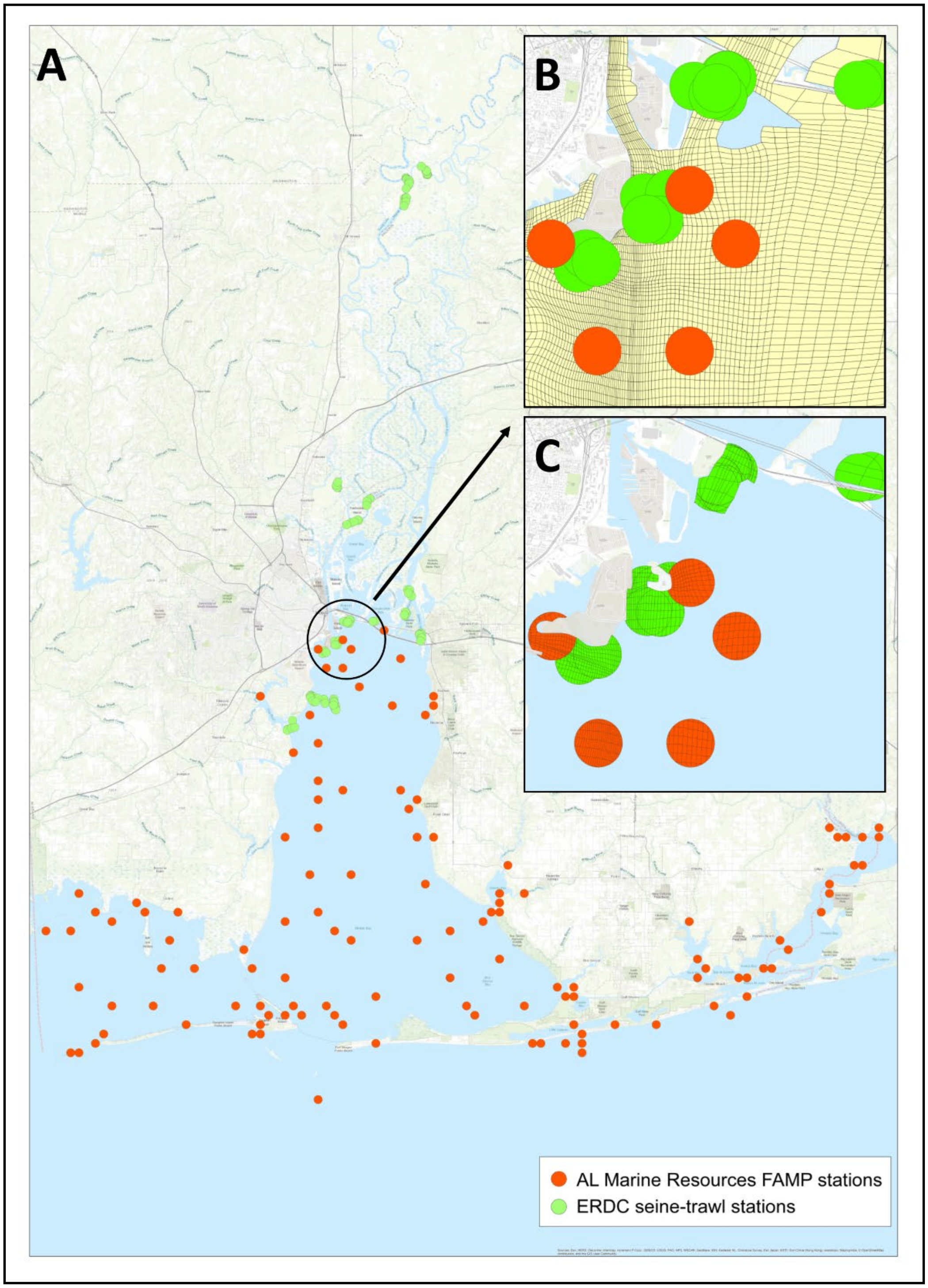


Figure 6.2. Distribution of ERDC sample stations (green) and Alabama Marine Resources FAMP stations (red) utilized for fisheries assessment (A). Panel B highlights a portion of the upper bay zone which depicts the station buffer layer and model grid. Panel C illustrates the extracted model grid cells for the corresponding sample stations.

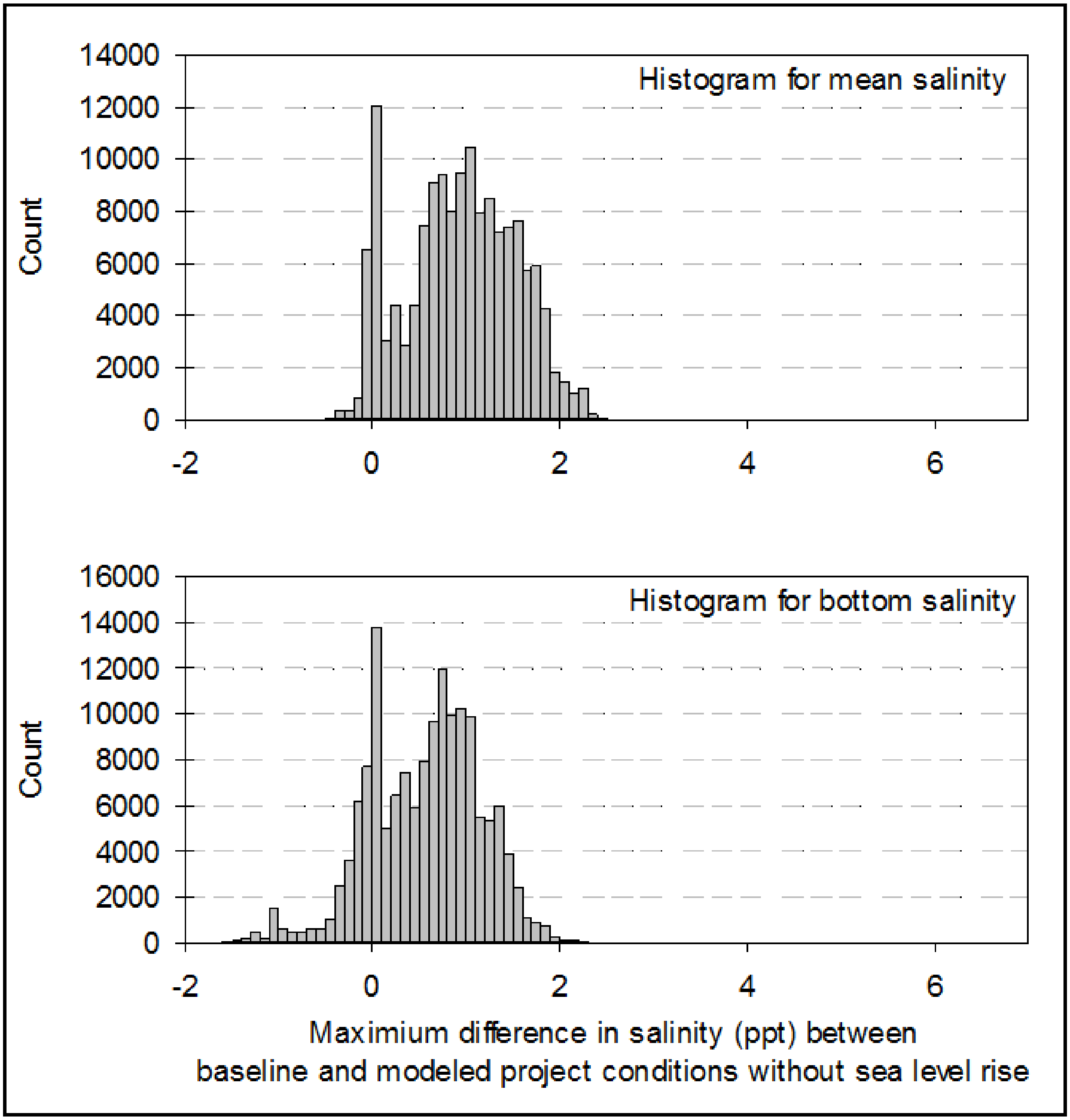


Figure 6.3. Maximum difference in model output between baseline and project conditions without sea level rise for mean and bottom salinity environmental parameters. Output values are based on intersect procedure between model grid and sample stations.

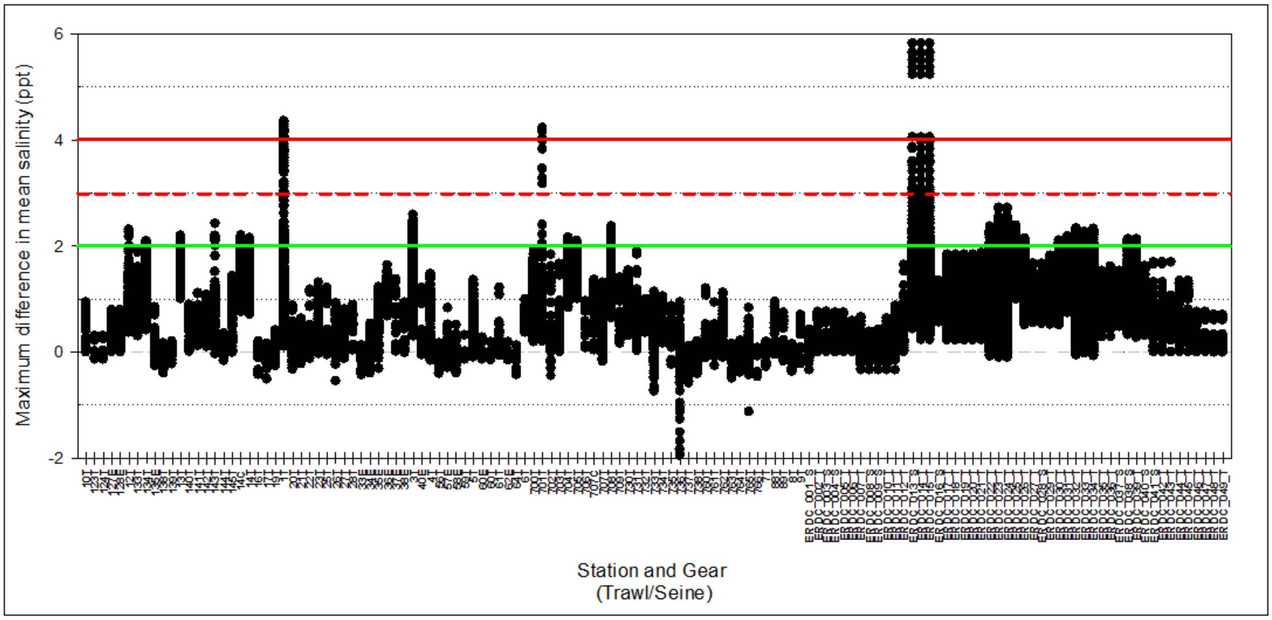


Figure 6.4. Model output for mean salinity (water column) with maximum difference in salinity (ppt) between baseline and modeled project conditions for all months at each designated AL Marine Resources and ERDC sample stations. Sampling station locations for codes in the x-axis are available upon request. For each station, the vertical row of dots represents all of the intersected cells from the model grid across all months. The stations are arranged alphabetically by station number and there is no geographic perspective (i.e., upper, middle or lower bay) portrayed by the order of the stations. Salinity differences with and without project are portrayed with reference lines at 2 (horizontal green line), 3 (horizontal dashed red line) and 4 ppt (solid horizontal red line).

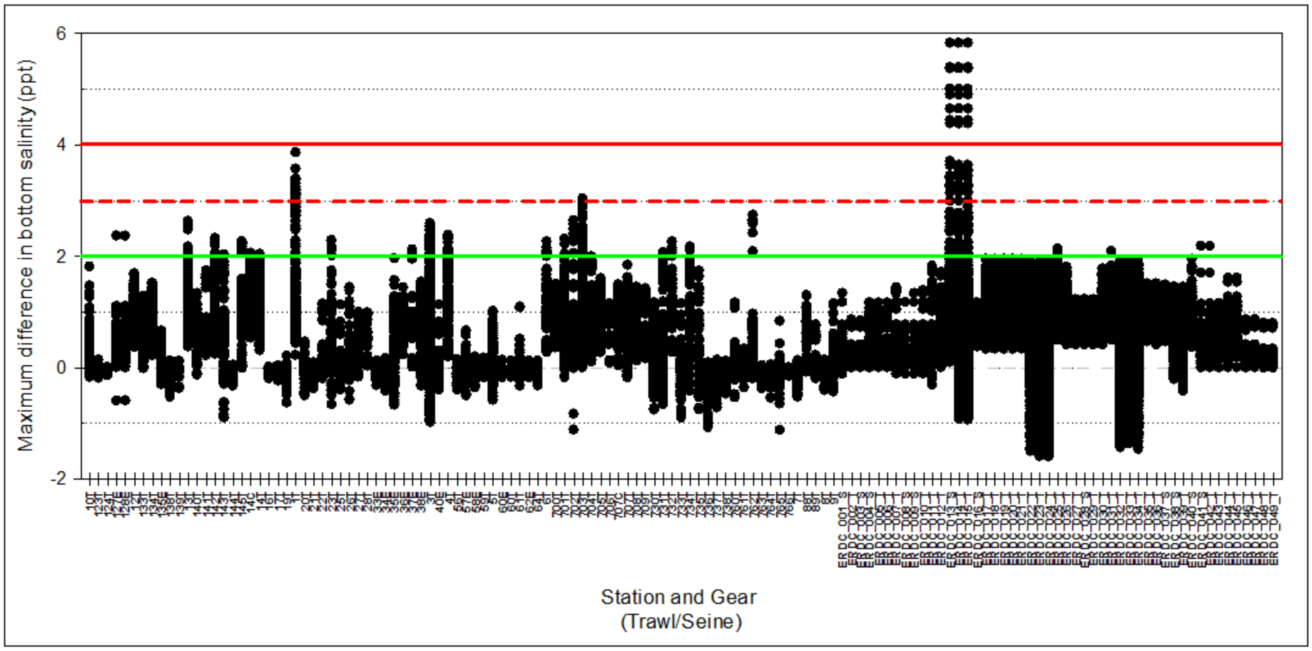


Figure 6.5. Model output for bottom salinity (lower third of water column) with maximum difference in salinity (ppt) between baseline and modeled project conditions for all months at each designated AL Marine Resources and ERDC sample stations. For each station, the vertical row of dots represents all of the intersected cells from the model grid across all months. The stations are arranged alphabetically by station number and there is no geographic perspective (i.e., upper, middle or lower bay) portrayed by the order of the stations. Salinity differences with and without project are portrayed with reference lines at 2 (horizontal green line), 3 (horizontal dashed red line) and 4 ppt (solid horizontal red line).

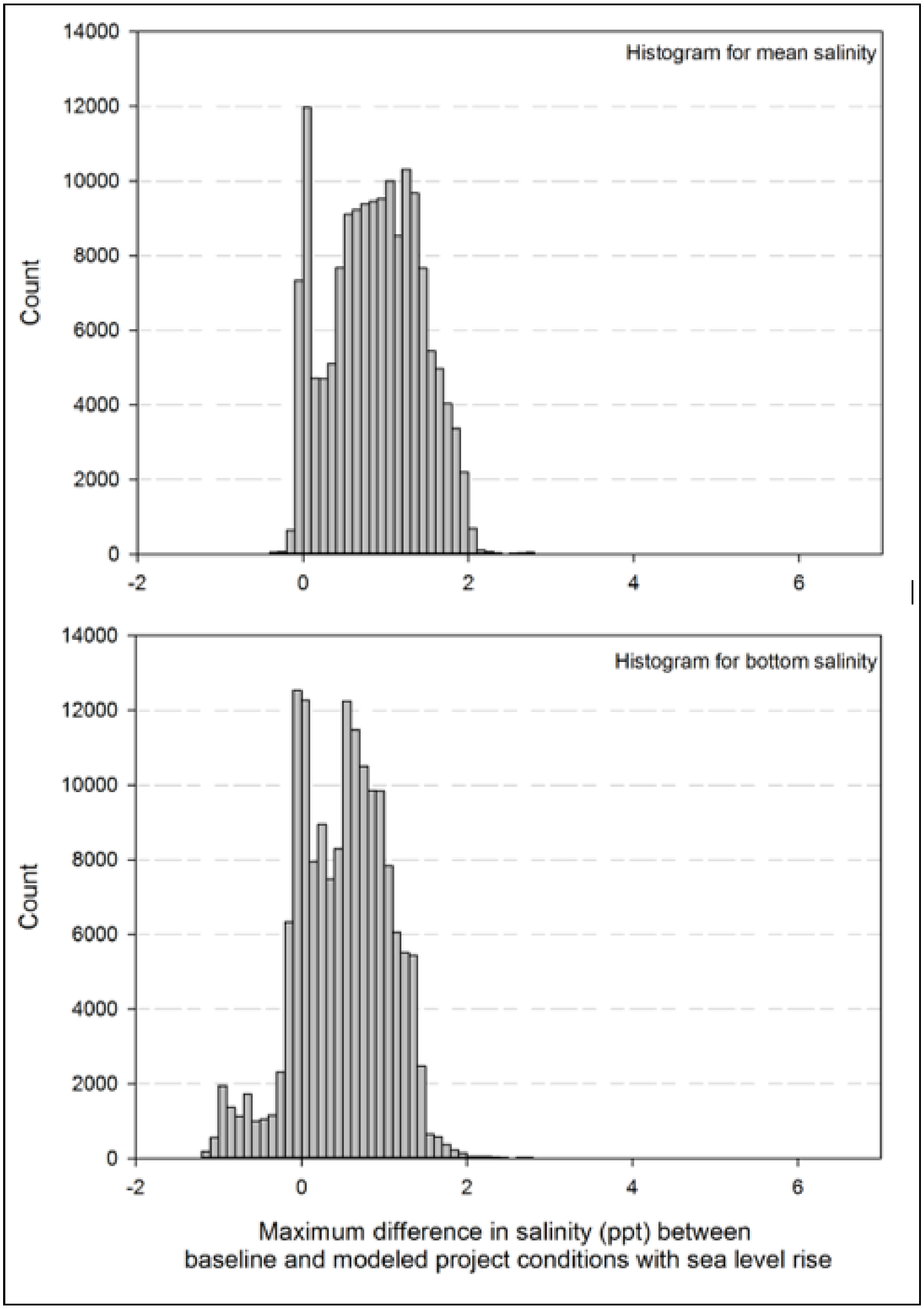


Figure 6.6. Maximum difference in model output between baseline and project conditions with sea level rise for mean and bottom salinity environmental parameters. Output values are based on intersect procedure between model grid and sample stations.

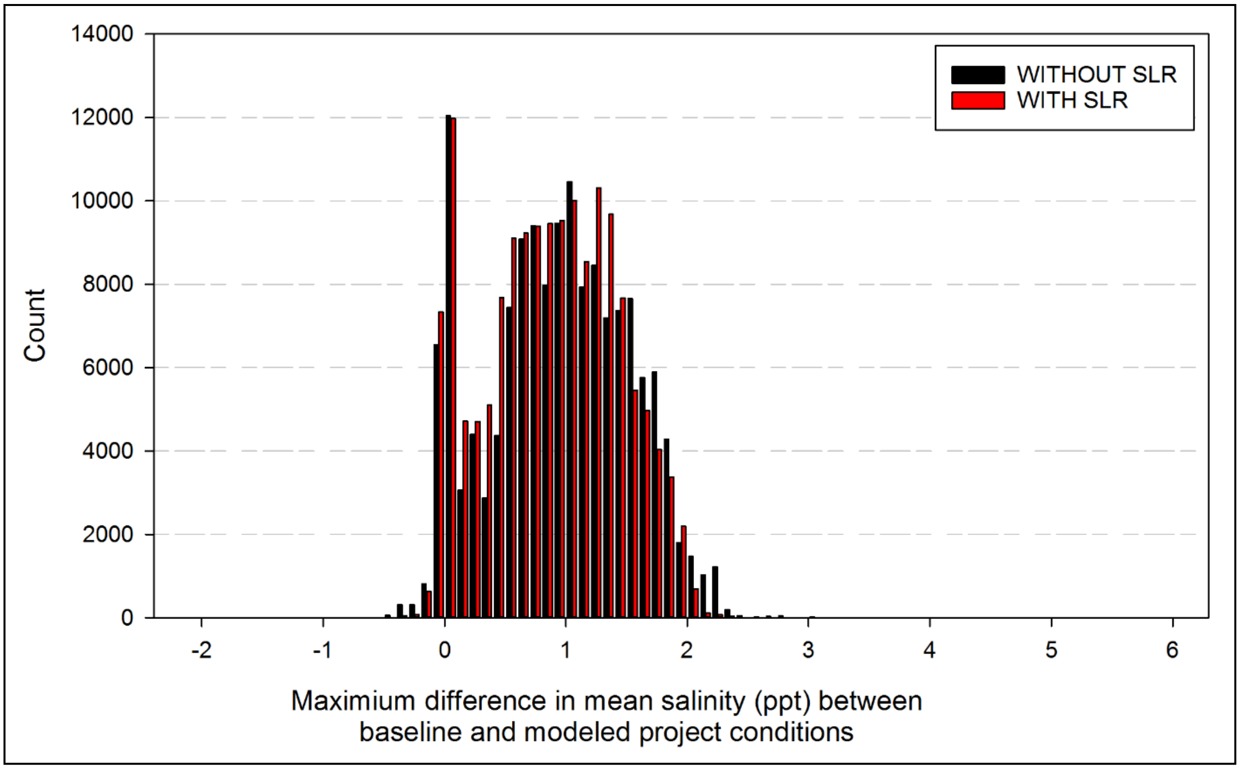


Figure 6.7. Comparative distribution for without and with sea level model projections regarding maximum differences in computed mean salinity values (ppt) between baseline and modeled project conditions.

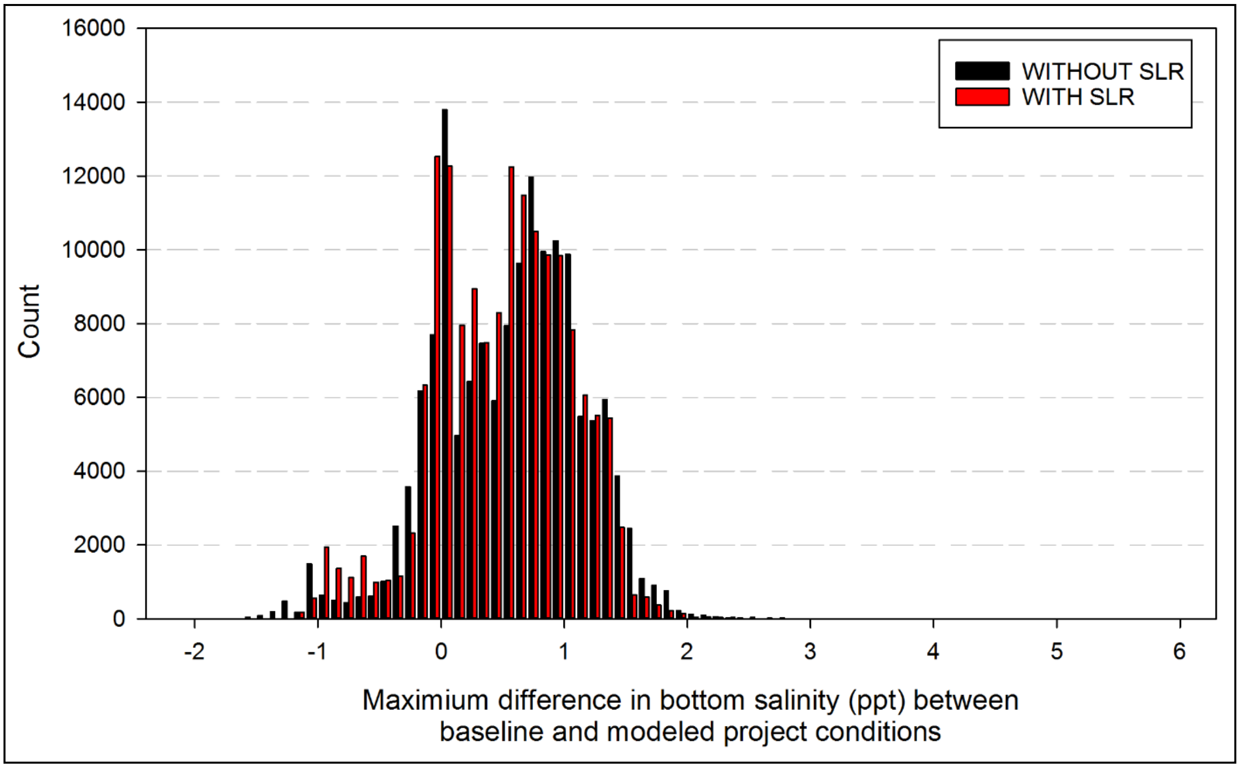


Figure 6.8. Comparative distribution for without and with sea level model projections regarding maximum differences in computed bottom salinity values (ppt) between baseline and modeled project conditions.

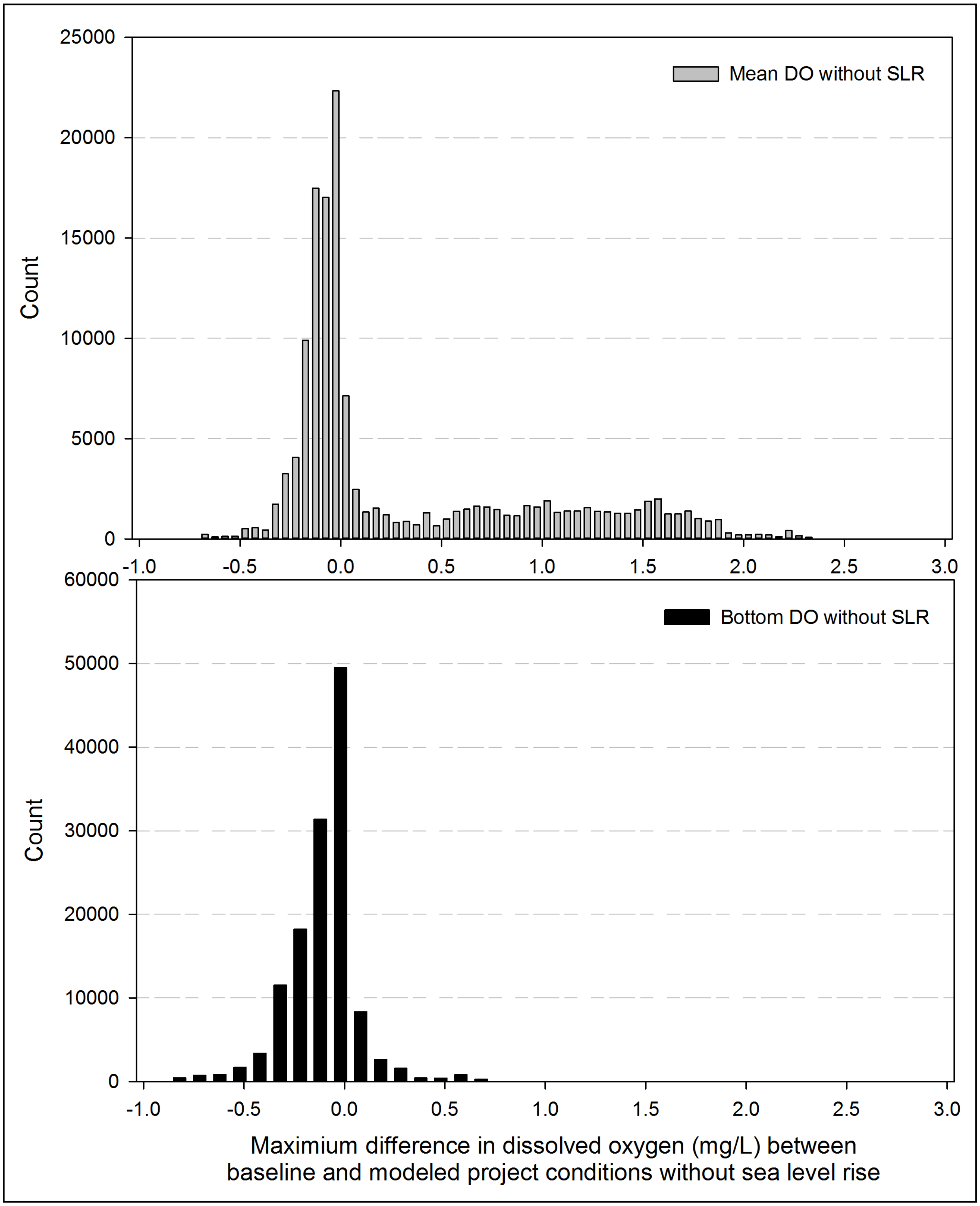


Figure 6.9. Maximum difference in model output between baseline and project conditions without sea level rise for mean and bottom dissolved oxygen environmental parameters. Output values are based on intersect procedure between model grid and sample stations.

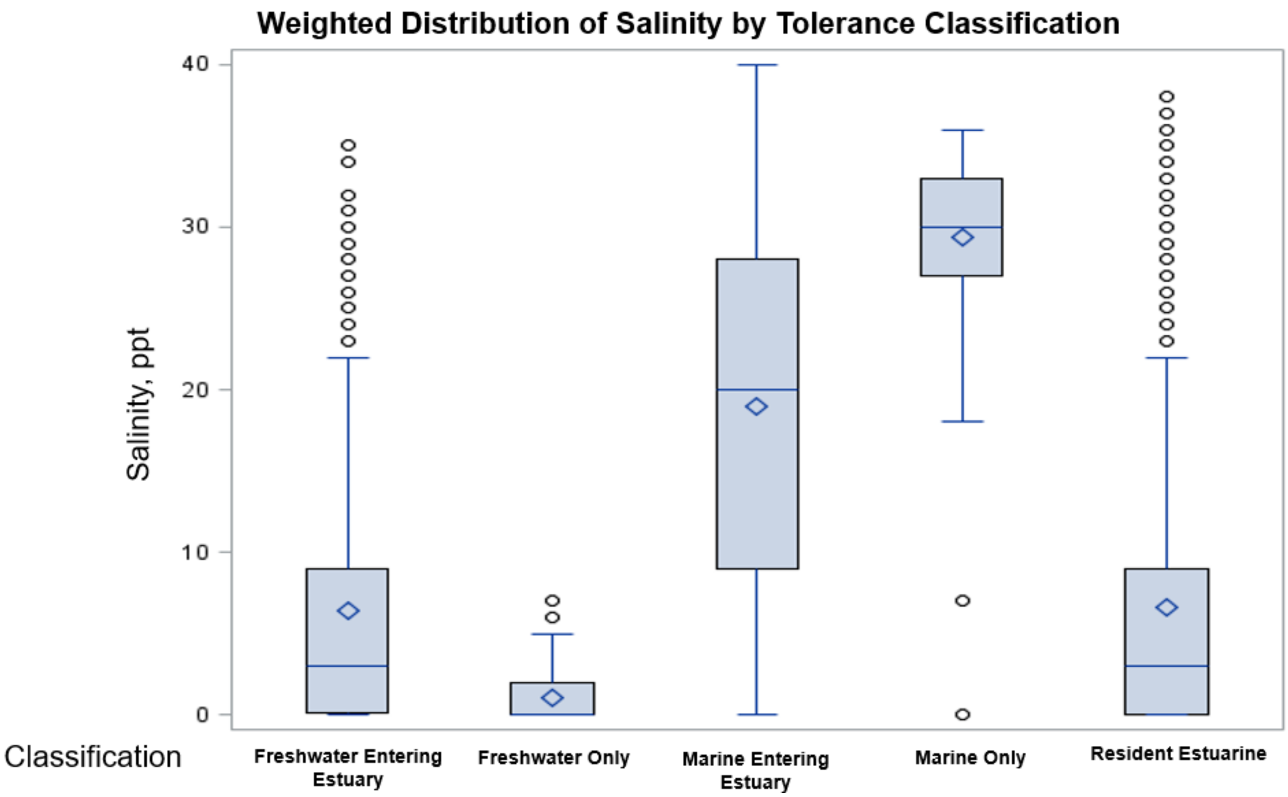
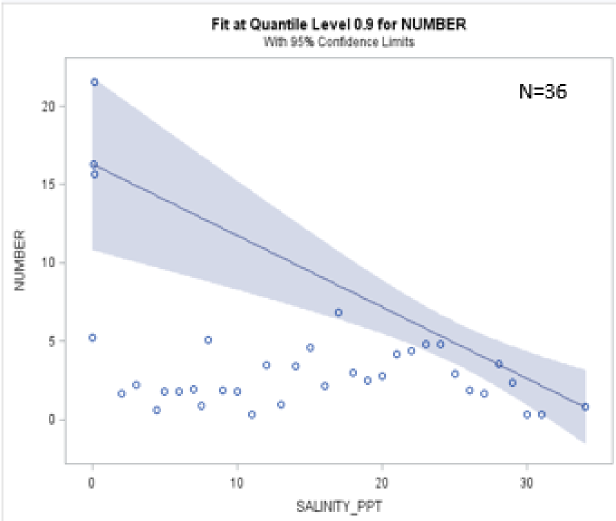


Figure 6.10. Box and whiskers plot of the weighted distribution of fish and shellfish by salinity tolerance classification in the Mobile Bay project area based on FAMP and ERDC collections from 2000-2017. Statistical properties are weighted according to the number of individuals as: $x_w = \sum_i w_i x_i / \sum_i w_i$ where $w_i =$ number of individuals and x_i = salinity. Each box includes mean weighted abundance (diamond), median (horizontal line inside box), first and third quartile (lower and upper edge of box, respectively) and minimum and maximum values (endpoint of lower and upper whisker, respectively). Circles represent extreme values outside of the normal distribution.

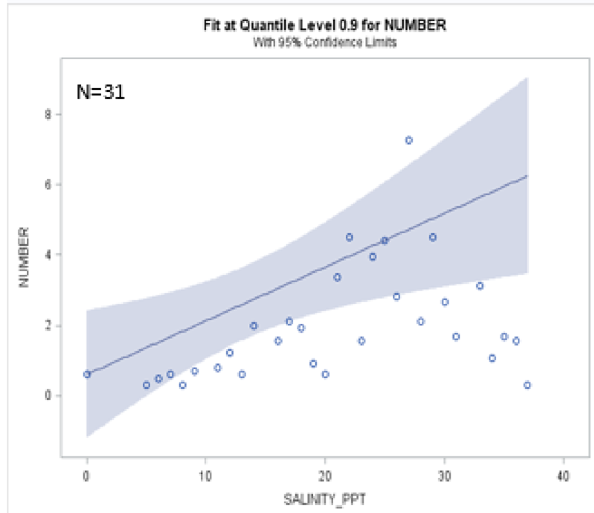
SEASON=Fall CLASSIFICATION=Freshwater entering estuary

Parameter Estimates							
Parameter	DF	Estimate	Standard Error	95% Confidence Limits		t Value	Pr > |t|
Intercept	1	16.3051	2.7515	10.7134	21.8969	5.93	<.0001
SALINITY_PPT	1	-0.4567	0.1068	-0.6738	-0.2395	-4.27	0.0001



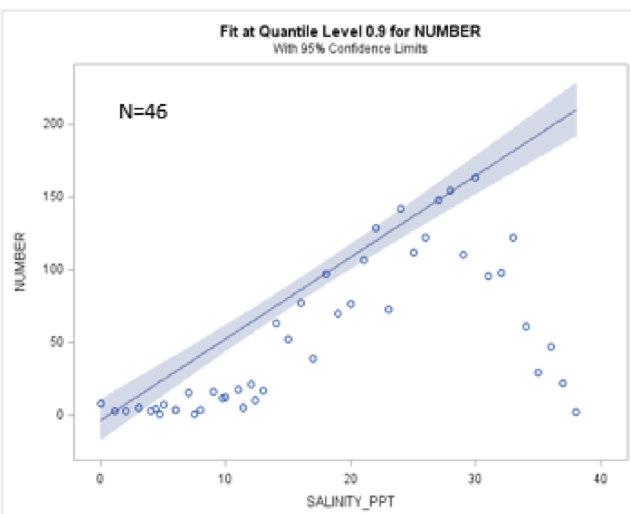
SEASON=Fall CLASSIFICATION=Resident estuarine

Parameter Estimates							
Parameter	DF	Estimate	Standard Error	95% Confidence Limits		t Value	Pr > |t|
Intercept	1	0.6021	0.8995	-1.2377	2.4418	0.67	0.5086
SALINITY_PPT	1	0.1528	0.0548	0.0408	0.2648	2.79	0.0092



SEASON=Fall CLASSIFICATION=Marine entering estuary

Parameter Estimates							
Parameter	DF	Estimate	Standard Error	95% Confidence Limits		t Value	Pr > |t|
Intercept	1	-3.7033	7.1645	-18.1423	10.7357	-0.52	0.6078
SALINITY_PPT	1	5.6069	0.3731	4.8550	6.3588	15.03	<.0001



SEASON=Fall CLASSIFICATION=Marine only

Parameter Estimates							
Parameter	DF	Estimate	Standard Error	95% Confidence Limits		t Value	Pr > |t|
Intercept	1	-5.7755	0.4258	-6.7127	-4.8384	-13.56	<.0001
SALINITY_PPT	1	0.3218	0.0164	0.2857	0.3579	19.62	<.0001

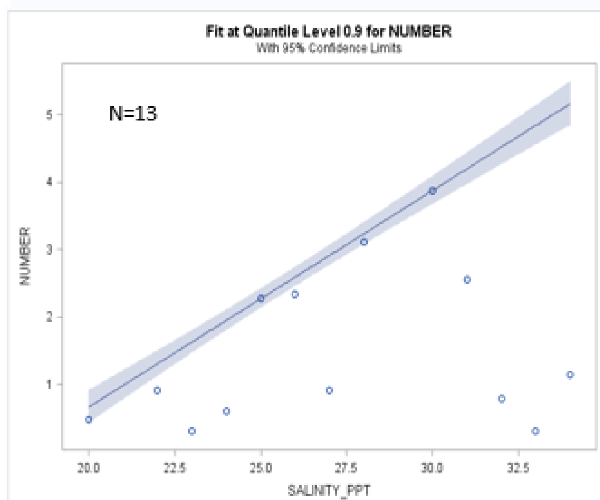
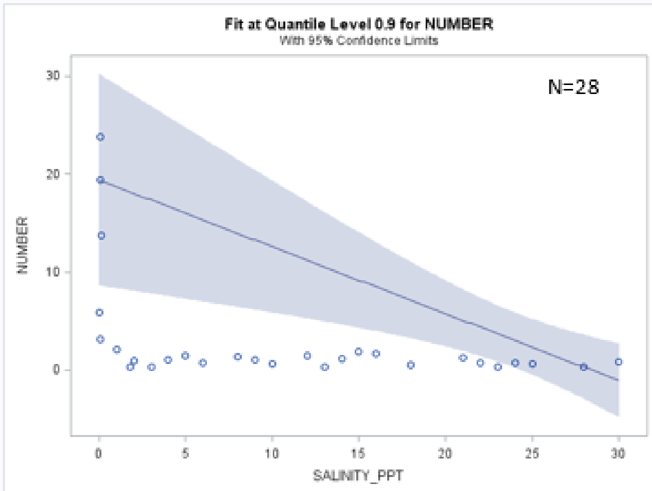


Figure 6.11. Quantile regression between numbers of fish classified according to salinity tolerance and salinity in ppt. The line indicates the 90% quantile and the shaded portion is the 95% confidence interval around the regression line. Parameter estimates are provided along with the probability of significance. Figures are shown by season.

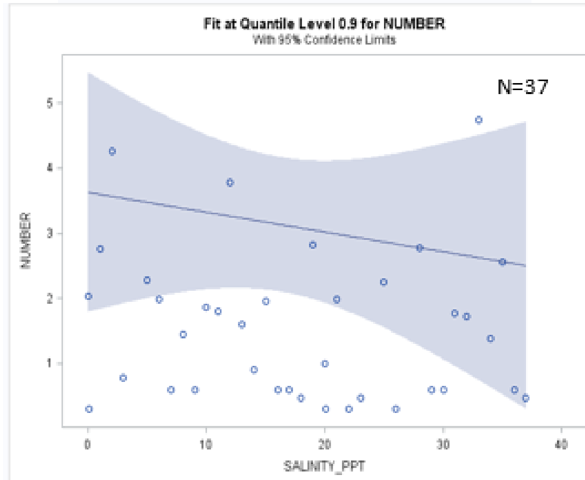
SEASON=Spring CLASSIFICATION=Freshwater entering estuary

Parameter Estimates							
Parameter	DF	Estimate	Standard Error	95% Confidence Limits		t Value	Pr > |t|
Intercept	1	19.4019	5.3415	8.4224	30.3814	3.63	0.0012
SALINITY_PPT	1	-0.6822	0.2128	-1.1195	-0.2448	-3.21	0.0035



SEASON=Spring CLASSIFICATION=Resident estuarine

Parameter Estimates							
Parameter	DF	Estimate	Standard Error	95% Confidence Limits		t Value	Pr > |t|
Intercept	1	3.6274	0.9176	1.7646	5.4902	3.95	0.0004
SALINITY_PPT	1	-0.0303	0.0468	-0.1252	0.0646	-0.65	0.5208



SEASON=Spring CLASSIFICATION=Marine entering estuary

Parameter Estimates							
Parameter	DF	Estimate	Standard Error	95% Confidence Limits		t Value	Pr > |t|
Intercept	1	26.3135	3.4457	19.3646	33.2623	7.64	<.0001
SALINITY_PPT	1	1.0889	0.2297	0.6256	1.5522	4.74	<.0001

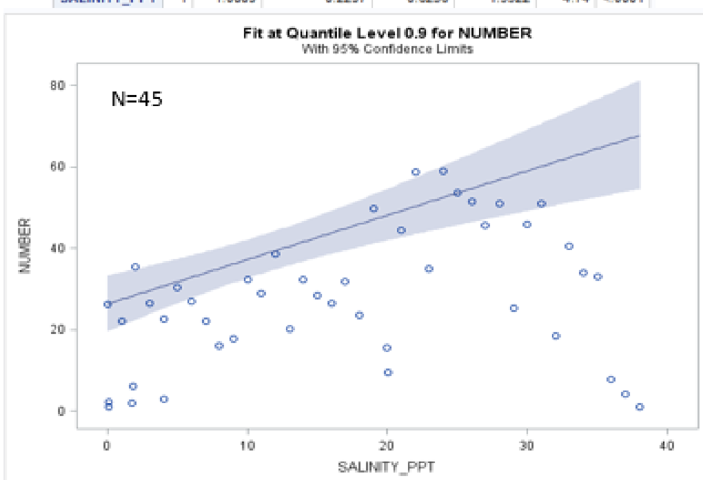
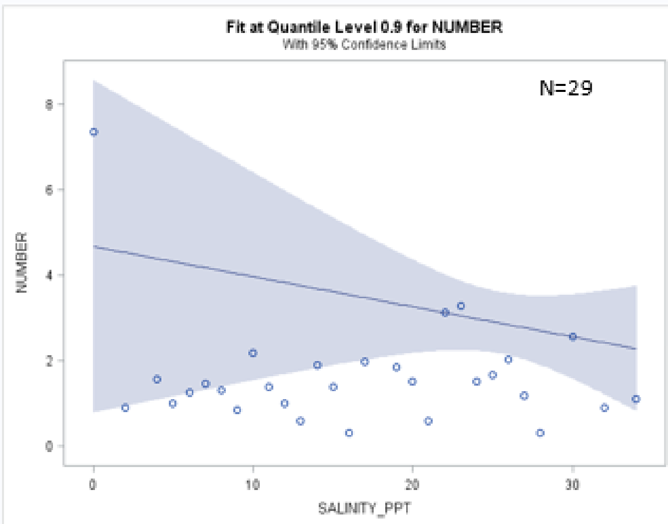


Figure 6.11. (Continued)

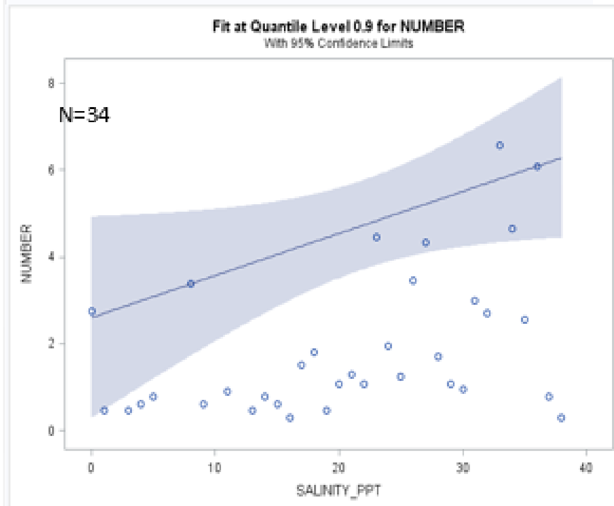
SEASON=Summer CLASSIFICATION=Freshwater entering estuary

Parameter Estimates							
Parameter	DF	Estimate	Standard Error	95% Confidence Limits		t Value	Pr > |t|
Intercept	1	4.6754	1.9289	0.7176	8.6332	2.42	0.0223
SALINITY_PPT	1	-0.0703	0.0743	-0.2227	0.0821	-0.95	0.3521



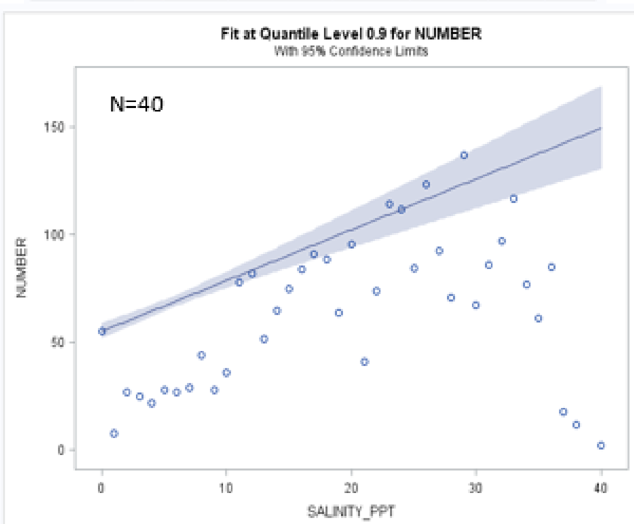
SEASON=Summer CLASSIFICATION=Resident estuarine

Parameter Estimates							
Parameter	DF	Estimate	Standard Error	95% Confidence Limits		t Value	Pr > |t|
Intercept	1	2.6055	1.1533	0.2562	4.9547	2.26	0.0308
SALINITY_PPT	1	0.0968	0.0473	0.0006	0.1931	2.05	0.0487



SEASON=Summer CLASSIFICATION=Marine entering

Parameter Estimates							
Parameter	DF	Estimate	Standard Error	95% Confidence Limits		t Value	Pr > |t|
Intercept	1	55.1491	1.8615	51.3806	58.9176	29.63	<.0001
SALINITY_PPT	1	2.3624	0.2720	1.8118	2.9130	8.69	<.0001



SEASON=Summer CLASSIFICATION=Marine only

Parameter Estimates							
Parameter	DF	Estimate	Standard Error	95% Confidence Limits		t Value	Pr > |t|
Intercept	1	-0.2755	1.9123	-4.4421	3.8910	-0.14	0.8878
SALINITY_PPT	1	0.1075	0.0699	-0.0448	0.2599	1.54	0.1501

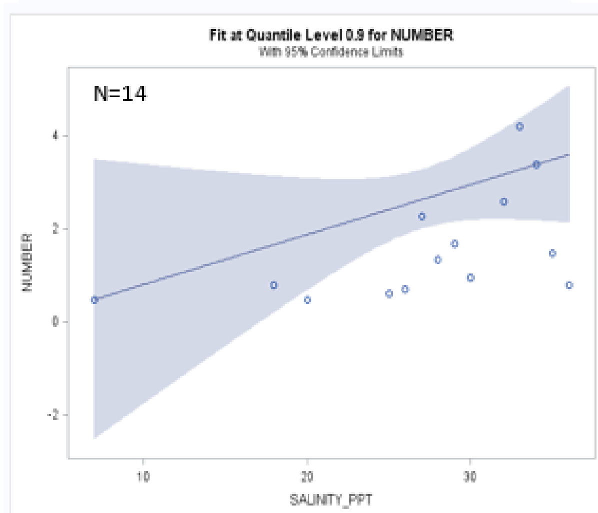
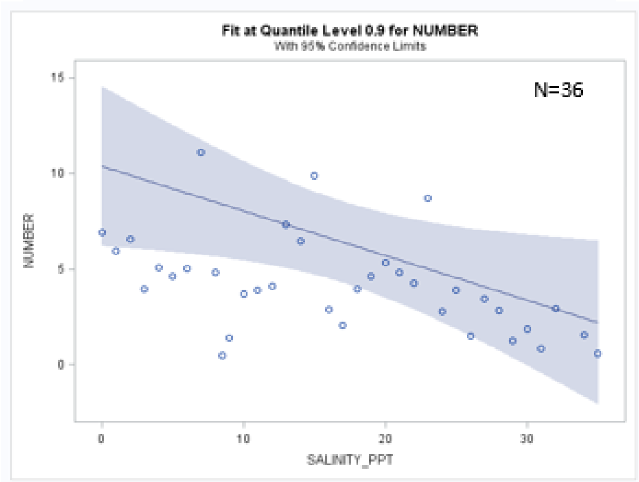


Figure 6.11. (Continued)

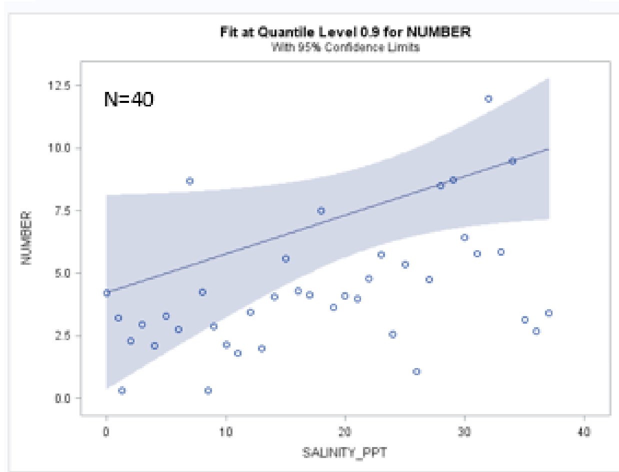
SEASON=Winter CLASSIFICATION=Freshwater entering estuary

Parameter Estimates							
Parameter	DF	Estimate	Standard Error	95% Confidence Limits		t Value	Pr > |t|
Intercept	1	10.3734	2.0819	6.1425	14.6043	4.98	<.0001
SALINITY_PPT	1	-0.2332	0.1042	-0.4451	-0.0214	-2.24	0.0319



SEASON=Winter CLASSIFICATION=Resident estuarine

Parameter Estimates							
Parameter	DF	Estimate	Standard Error	95% Confidence Limits		t Value	Pr > |t|
Intercept	1	4.2278	1.9430	0.2944	8.1612	2.18	0.0358
SALINITY_PPT	1	0.1551	0.0781	-0.0031	0.3132	1.99	0.0544



SEASON=Winter CLASSIFICATION=Marine entering estuary

Parameter Estimates							
Parameter	DF	Estimate	Standard Error	95% Confidence Limits		t Value	Pr > |t|
Intercept	1	17.8073	0.7788	16.2320	19.3825	22.87	<.0001
SALINITY_PPT	1	2.4613	0.3030	1.8485	3.0742	8.12	<.0001

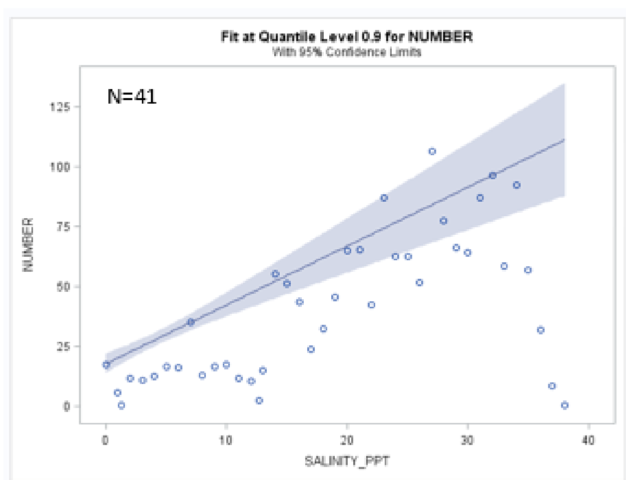


Figure 6.11. (Continued)

Table 6.1. Mean values of Salinity (ppt) and Dissolved Oxygen (mg/l) by zone in Mobile Bay project area.						
ZONE	Variable	N	Mean	Std Dev	Minimum	Maximum
Lower Bay	Salinity	864	23.1	8.4	0.5	37.3
	Dissolved Oxygen	863	6.6	1.7	0.4	12.2
Middle Bay	Salinity	272	12.0	7.3	0.5	30.5
	Dissolved Oxygen	272	6.8	2.0	0.5	12.0
Upper Bay	Salinity	199	8.9	6.3	0.3	24.5
	Dissolved oxygen	198	6.5	2.1	1.7	13.0
Transition	Salinity	12	3.7	3.7	0.1	9.7
	Dissolved Oxygen	12	7.0	1.3	5.0	8.8
Freshwater	Salinity	4	0.1	0.0	0.1	0.2
	Dissolved Oxygen	4	7.4	0.6	6.7	8.0

Table 6.2. Species abundance in the Mobile Bay project area by salinity classification. Species are arranged in order of numerical abundance.				
CLASSIFICATION=Freshwater entering estuary				
Common Name	Frequency	Percent	Cumulative Frequency	Cumulative Percent
Sailfin molly	3141	29.53	3141	29.53
Threadfin shad	2910	27.36	6051	56.9
Blue catfish	1932	18.17	7983	75.06
Largemouth bass	740	6.96	8723	82.02
Redear sunfish	460	4.33	9183	86.35
Redspotted sunfish	369	3.47	9552	89.82
Western mosquitofish	319	3	9871	92.82
Channel catfish	301	2.83	10172	95.65
Bluegill	143	1.34	10315	96.99
Black crappie	133	1.25	10448	98.24
Gizzard shad	79	0.74	10527	98.98
Smallmouth buffalo	19	0.18	10546	99.16
Longear sunfish	18	0.17	10564	99.33
Skipjack herring	18	0.17	10582	99.5
Spotted gar	16	0.15	10598	99.65
Saltmarsh topminnow	14	0.13	10612	99.78
Longnose gar	11	0.1	10623	99.89
Least killifish	6	0.06	10629	99.94
River carpsucker	2	0.02	10631	99.96
Alligator gar	1	0.01	10632	99.97
Coastal shiner	1	0.01	10633	99.98
Golden topminnow	1	0.01	10634	99.99
White crappie	1	0.01	10635	100
CLASSIFICATION=Freshwater only				
Common Name	Frequency	Percent	Cumulative Frequency	Cumulative Percent
Silverside shiner	2060	94.71	2060	94.71
Freshwater drum	40	1.84	2100	96.55
Emerald shiner	24	1.1	2124	97.66
Silver chub	17	0.78	2141	98.44
Fluvial shiner	9	0.41	2150	98.85
Mississippi silvery minnow	8	0.37	2158	99.22
Golden shiner	6	0.28	2164	99.49
Green sunfish	4	0.18	2168	99.68
Crystal darter	2	0.09	2170	99.77
Starhead topminnow	2	0.09	2172	99.86

Banded pygmy sunfish	1	0.05	2173	99.91
Flathead catfish	1	0.05	2174	99.95
Taillight shiner	1	0.05	2175	100
CLASSIFICATION=Marine entering estuary				
Common Name	Frequency	Percent	Cumulative Frequency	Cumulative Percent
Spot	531328	44.54	531328	44.54
Gulf menhaden	238228	19.97	769556	64.51
Atlantic croaker	172572	14.47	942128	78.98
Pinfish	46220	3.87	988348	82.85
Spotfin mojarra	38045	3.19	1026393	86.04
Sand seatrout	28855	2.42	1055248	88.46
Striped mullet	28126	2.36	1083374	90.82
Hardhead catfish	14575	1.22	1097949	92.04
Dusky anchovy	12567	1.05	1110516	93.09
Star drum	11950	1	1122466	94.09
Striped anchovy	8795	0.74	1131261	94.83
Atlantic bumper	7215	0.6	1138476	95.43
Rough silverside	6076	0.51	1144552	95.94
Blackcheek tonguefish	5753	0.48	1150305	96.43
Silver perch	5174	0.43	1155479	96.86
Bay whiff	4358	0.37	1159836	97.23
Gafftopsail catfish	2868	0.24	1162704	97.47
Gulf butterfish	2852	0.24	1165556	97.7
White mullet	2281	0.19	1167837	97.9
Least puffer	2184	0.18	1170021	98.08
Inshore lizardfish	1934	0.16	1171955	98.24
Fringed flounder	1921	0.16	1173876	98.4
Banded drum	1774	0.15	1175650	98.55
Bighead searobin	1628	0.14	1177278	98.69
Southern kingfish	1484	0.12	1178762	98.81
Silver seatrout	1160	0.1	1179922	98.91
Southern hake	1113	0.09	1181035	99
Scaled sardine	1022	0.09	1182057	99.09
Pigfish	994	0.08	1183051	99.17
Atlantic cutlassfish	757	0.06	1183808	99.23
Atlantic stingray	755	0.06	1184563	99.3
Spotted hake	754	0.06	1185317	99.36
Silver jenny	689	0.06	1186006	99.42
Marsh killifish	647	0.05	1186653	99.47
Atlantic moonfish	579	0.05	1187232	99.52
Southern flounder	444	0.04	1187676	99.56

Harvestfish	436	0.04	1188112	99.6
Spadefish	399	0.03	1188511	99.63
Gulf pipefish	389	0.03	1188900	99.66
Atlantic needlefish	381	0.03	1189281	99.69
Lane snapper	341	0.03	1189622	99.72
Red drum	288	0.02	1189910	99.75
Lookdown	270	0.02	1190180	99.77
Chain pipefish	252	0.02	1190432	99.79
Rock sea bass	250	0.02	1190682	99.81
Crevalle jack	204	0.02	1190886	99.83
Leatherjacket	194	0.02	1191080	99.84
Crested cusk-eel	187	0.02	1191267	99.86
Ladyfish	149	0.01	1191416	99.87
Dwarf sand perch	142	0.01	1191558	99.88
Leopard searobin	133	0.01	1191691	99.9
Gray snapper	130	0.01	1191821	99.91
Sheepshead	127	0.01	1191948	99.92
Bluntnose jack	109	0.01	1192057	99.93
Gulf flounder	93	0.01	1192150	99.93
Guaguanche	71	0.01	1192221	99.94
Atlantic midshipman	69	0.01	1192290	99.95
Longspine porgy	67	0.01	1192357	99.95
Atlantic thread herring	64	0.01	1192421	99.96
Spotted whiff	62	0.01	1192483	99.96
Spanish mackerel	47	0	1192530	99.97
Smooth butterfly ray	44	0	1192574	99.97
Southern stargazer	40	0	1192614	99.97
Blackwing searobin	39	0	1192653	99.98
Skilletfish	38	0	1192691	99.98
Florida pompano	31	0	1192722	99.98
Fat sleeper	23	0	1192745	99.98
Lined seahorse	23	0	1192768	99.99
Bluefish	19	0	1192787	99.99
Northern kingfish	19	0	1192806	99.99
Round scad	11	0	1192817	99.99
Crested blenny	10	0	1192827	99.99
Emerald sleeper	10	0	1192837	99.99
Lined sole	10	0	1192847	99.99
Singlespot frogfish	10	0	1192857	99.99
Gulf kingfish	9	0	1192866	99.99
Northern sennet	8	0	1192874	99.99
Yellowfin menhaden	7	0	1192881	100
Clearnose skate	6	0	1192887	100

Cobia	6	0	1192893	100
Southern puffer	6	0	1192899	100
Southern stingray	6	0	1192905	100
Pygmy sea bass	5	0	1192910	100
Sharksucker	4	0	1192914	100
Bluespotted searobin	3	0	1192917	100
Scrawled cowfish	3	0	1192920	100
Smooth puffer	3	0	1192923	100
Bandtail puffer	2	0	1192925	100
Blue runner	2	0	1192927	100
Lyre goby	2	0	1192929	100
Tripletail	2	0	1192931	100
Atlantic threadfin	1	0	1192932	100
Cownose ray	1	0	1192933	100
Florida blenny	1	0	1192934	100
Frillfin goby	1	0	1192935	100
Great barracuda	1	0	1192936	100
Roundel skate	1	0	1192937	100
Shortnose batfish	1	0	1192938	100
CLASSIFICATION=Marine only				
Common Name	Frequency	Percent	Cumulative Frequency	Cumulative Percent
Red snapper	288	91.43	288	91.43
Broad flounder	9	2.86	297	94.29
Blackedge cusk-eel	8	2.54	305	96.83
Rough scad	3	0.95	308	97.78
Dusky flounder	2	0.63	310	98.41
Spotted batfish	2	0.63	312	99.05
Mexican searobin	1	0.32	313	99.37
Round herring	1	0.32	314	99.68
Smoothhead scorpionfish	1	0.32	315	100
CLASSIFICATION=Resident estuarine				
Common Name	Frequency	Percent	Cumulative Frequency	Cumulative Percent
Bay anchovy	840659	94.27	840659	94.27
Inland silverside	30448	3.41	871107	97.68
Rainwater killifish	12137	1.36	883244	99.04
Sheepshead minnow	2551	0.29	885795	99.33
Speckled worm eel	1256	0.14	887051	99.47
Spotted seatrout	1024	0.11	888075	99.59
Clown goby	954	0.11	889029	99.69

Striped killifish	852	0.1	889881	99.79
Gulf killifish	540	0.06	890421	99.85
Highfin goby	511	0.06	890932	99.91
Naked goby	324	0.04	891256	99.94
Diamond killifish	257	0.03	891513	99.97
Green goby	145	0.02	891658	99.99
Gulf toadfish	56	0.01	891714	99.99
Black drum	40	0	891754	100
Freckled blenny	9	0	891763	100
Code goby	5	0	891768	100
Twoscale goby	3	0	891771	100
Feather blenny	1	0	891772	100
Striped blenny	1	0	891773	100