Evaluation of the Proposal for Widening and Deepening the Matagorda Ship Channel

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Abbreviations and Acronyms

2D	Two Dimensional
3D	Three Dimensional
ADCIRC	ADvanced CIRCulation model
ALCOA	The Aluminum Company of America
BI	Bird Island
BI-BM	Bird Island Channel Marker 19
BI-BM	Bird Island Channel Marker 19
CSAT	Corps Shoaling Analysis Tool
CHL	Coastal and Hydraulics Laboratory
CMS	USACE Coastal Modeling System
CR	Colorado River
CRNC	Colorado River Navigation Channel
DEM	Digital Elevation Model
EMAT2	East Matagorda Bay
EPA	Environmental Protection Agency
ERDC	U. S. Army Engineer Research and Development Centers
FPP	Frontal Passages Period
FR/EIS	Feasibility Report/Environmental Impact Statement
FWOP	Future Without Project
FWP	Future With Project
GCA	Gulf Coast Aquifer
Hg	Mercury
IEPR	Independent External Peer Review
LAVC	Lavaca
LB	Lavaca Bay
LB	Lavaca Bay
LCRA	Lower Colorado River Authority
MATA	Matagorda
MB	Matagorda Bay
MCR	Colorado River Mouth
MLLW	mean lower low water
MQTT2	Bob Hall Pier
MSC	Matagorda Ship Channel
MSL	mean sea level
NDBC	National Data Buoy Center
NOAA	National Oceanic and Atmospheric Administration
NTU	Nephelometric Turbidity Units
O&M	Operation & Maintenance
ODMDS	Ocean Dredged Material Disposal Site

PAH	Polycyclic Aromatic Hydrocarbons
PC	Pass Cavallo
PCB	Polychlorinated Biphenyl
PCNT2	Port O’Connor
PE	Polyethylene
PET	Thermoplastic Polyester
PP	Polypropylene
PS	Polystyrene
PUR	Polyurethane
PVC	Polyvinyl chloride
PWOP	Present Without Project
PWP	Present With Project
QP	Quiet Period
RMSE	Root Mean Square Error
RSLR	Relative Sea Level Rise
SAV	submerged aquatic vegetation
SEP	Southeasterly Period
SLAMM	Sea Level Affecting Marshes Model
SLR	Sea Level Rise
SWG	USACE Galveston District
TCRMP	Texas Coastal Resiliency Master Plan
TGLO	Texas General Land Office
TPAL	Tres Palacios
TPWD	Texas Parks and Wildlife Department
TSS	Total Suspended Solids
TWDB	Texas Water Development Board
USACE	U.S. Army Corps of Engineers
USFWS NWI	U.S. Fish and Wildlife Service, National Wetland Inventory
USGS	United States Geological Survey
UV	Ultraviolet
VCAT2	Lavaca Bay Bridge
VEC	Valued ecosystem components
WSE	Water Surface Elevation

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    - Groundwater AD
    - Habitats KT
    - Turbidity LR
    - Mercury JP
    - Plastics AD
    - Conclusions PM, JG, AD, JP, LR, MS, & KT
- Writing – review & editing led by PM & all authors contributed.

Abstract

There is a proposal for deepening and widening the Matagorda Ship Channel. However, there are concerns about specific environmental impacts to Matagorda Bay and Lavaca Bay such as: 1) Circulation and storm surge changes due to changing the bathymetry of Matagorda and Lavaca Bays due to deepening the channel and placement of dredge spoil parallel to the channel, which will funnel tidal movement along the axis of the bay and restrict mixing perpendicular to the channel. 2) Salinity changes due to the circulation changes and connecting with the puncturing of the groundwater lens beneath the bay. 3) Placement of dredge spoil could smother benthic habitats, such as oysters, seagrasses, or muddy bay bottoms and affect forage for desirable fish species. 4) Effects to primary producers or bottom dwelling organisms due to an increase in turbidity during the actual dredging. 5) Mobilization of mercury by the dredging because there are considerable amounts of mercury in sediments of the bay. 6) Plastic and microplastics interactions with sediments, turbidity, and other pollutants. The current study was undertaken to assess these possible concerns. The assessment has been done in two parts to identify potential physical and ecological effects. The assessments were based on a literature review of existing information. No sampling was conducted.

Introduction

The Calhoun Port Authority has initiated a study with the U.S. Army Corps of Engineers (USACE) to acquire a permit for improving the Matagorda Ship Channel (MSC), Port Lavaca, Texas (USACE 2019). The MSC is 26 miles long and extends from offshore in the Gulf of Mexico through Matagorda Bay and Lavaca Bay to the Port. The existing channel is used by vessels with drafts up to 38 feet deep at mean lower low water (MLLW). The channel bottom is 200 feet wide. The existing turning basin is 1000 by 1000 feet wide.

The current alternative plan A is to deepen the channel to 47 feet MLLW, widen it to 350 feet in the bay and 600 feet in the Gulf, and expand the dimensions of the turning basin to 1,200 by 1,200 feet. The study predicts this would provide $6,539,000 in total net benefits, with a benefit-to-cost ratio of 1.3 (USACE 2019).

The study also assesses the effects on the natural system and human environment, including the economic development effects of existing inefficiencies (USACE 2019). The study does identify environmental resources in Lavaca and Matagorda Bays in Section 2 and concludes there will be no changes to any natural resources due to the deepening and widening of the MSC as proposed in Section 3. In Section 5, it states "Cumulative impacts due to past, existing, and reasonably foreseeable future projects, along with the Recommended Plan, are not expected to have significant adverse effects to resources in the study area."

However, there are concerns specific to Matagorda Bay and Lavaca Bay that there could be the following potential environmental changes:

1) Physical changes
    a. Circulation and storm surge changes due to changing the bathymetry of Matagorda and Lavaca Bays due to deepening the channel and placement of dredge spoil parallel to the channel, which will funnel tidal movement along the axis of the bay and restrict mixing perpendicular to the channel.
    b. Salinity changes due to the circulation changes and connecting with the groundwater lens.
    c. Puncturing the groundwater lens beneath the bay.
2) Ecological changes
    a. Placement of dredge spoil could smother benthic habitats, such as oysters, seagrasses, or muddy bay bottoms. This would affect forage potential for desirable fish species.
    b. Effects to primary producers or bottom dwelling organisms due to increased turbidity during the actual dredging.
    c. Mobilization of mercury by the dredging. Lavaca Bay is a National Superfund Site with considerable amounts of mercury in sediments of the bay.
    d. Effects of plastics and microplastics because they can interact with pollutants such as mercury to enhance toxicity, and sediments to enhance turbidity effects.

Methods

The current study was undertaken to assess these possible issues of concern. The assessment was done in two parts to identify potential physical and ecological effects. The assessments of each issue were based on a literature review of past or similar projects performed in other areas, and examining existing data and information about Lavaca and Matagorda Bays. No new samples or statistical analyses were performed.

Results and Discussion

Physical Assessment

A hydrodynamic and salinity modeling study for Matagorda Ship Channel (MSC) channel improvement project was requested and funded by the USACE Galveston District (SWG). The numerical modeling study was performed by the U. S. Army Engineer Research and Development Centers' (ERDC) Coastal and Hydraulics Laboratory (CHL) Coastal Engineering Branch, Navigation Division. The modeling study investigated the impact to water levels, currents, and salinity in and around Matagorda Bay from the proposed ship channel widening/deepening project (Lin et al. 2018). An ERDC Letter Report (Lin et al. 2018) based on the study was prepared for the SWG in July 2018 and reviewed for the current report. Additionally, two conference proceedings (Lin et al. 2020a, Lin et al. 2020b) that are based on the modeling study have been reviewed.

Hydrodynamic and Salinity Model Settings

The USACE Coastal Modeling System (CMS) (Demirbilek and Rosati, 2011) was used to simulate hydrodynamics and salinity variation in Matagorda Bay with and without a proposed channel widening and deepening project of the MSC. Although one of the conference papers (Lin et al. 2020b) also mentioned using the ADvanced CIRCulation model (ADCIRC) as a regional hydrodynamic model to provide boundary conditions input to the CMS modeling, the ERDC Letter Report (Lin et al. 2018) does not mentioned the ADCIRC modeling.

The CMS calculates water levels, currents, waves, sediment transport, and salinity through coupling between a hydrodynamic model, CMS-Flow (Buttolph et al. 2006), and a wave spectral model, CMS-Wave (Lin et al. 2008). These two models can be run separately or coupled on a non-uniform Cartesian grid. In the coupling mode, the variables passed from CMS-Wave to CMS-Flow are the significant wave height, peak wave period, wave direction, wave breaking dissipation, and radiation stress gradients. CMS-Wave uses the updated bathymetry, water levels, and currents from CMS-Flow (Lin et al. 2020a). A coupled CMS model, in 2D version, was used to calculate the circulation and salinity change on a Cartesian grid (Lin et al. 2018). The physical processes being considered in the ERDC study (Lin et al. 2018) are water surface elevation (WSE), current and salinity, but the sediment transport and changes in morphology were not modeled.

The CMS model grid developed for the previous studies (Kraus et al. 2006, Maynord et al. 2011) was used for the ERDC study (Lin et al. 2018) with a small modification. The CMS grid covers the Matagorda Bay system with two major inlets, MSC and Pass Cavallo, and a smaller inlet along the lower reach of the Colorado River, which connects the mouth of Colorado River (MCR) through the Colorado River Navigation Channel (CRNC) (Lin et al. 2018, Figure 1). The model domain extends northward to include the lower reaches of Lavaca River and Garcitas Creek and southward to the 20 m (65.6 ft) depth in the Gulf of Mexico. The east boundary is along the tidal flat of East Matagorda Bay and the west boundary reaches the eastern part of Espiritu Santo Bay. The model cell resolution varies from 20 m (65.6 ft) near the MSC to 500 m (1,640 ft) in the offshore area. The model bathymetry data in the grid were updated with the latest bathymetry, digital elvation model (DEM), and channel survey data wherever available. Figure 1 shows the CMS grid and bathymetry used in the ERDC study (Lin et al. 2018).

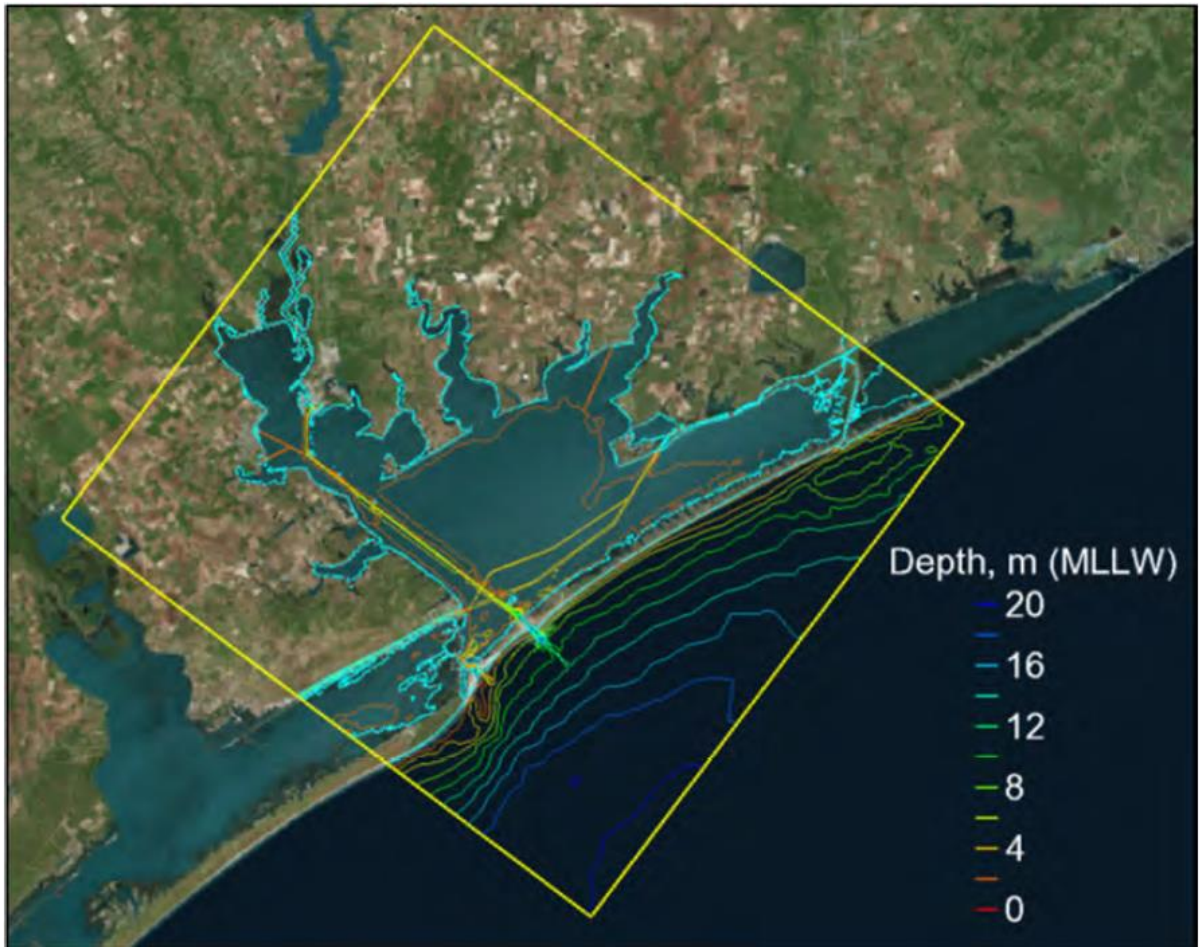

Figure 1. CMS Model domain and bathymetry (Lin et al. 2018).

The CMS model used in the study was applied in a 2D version for the salinity modeling. However, multiple modeling studies have shown that hydrodynamic transport in the estuaries can be well-described by 2D models (Achete et al. 2015, Mahn et al. 2014), whereas salinity dynamics and transport in the estuaries are governed by 3D processes such as estuarine circulation and is a bit unrealistic to simulate these processes in a 2D model (Martyr-Koller et al. 2017). Therefore, it is recommended to model salinity dynamics using a 3D model so that the gravitational circulation and temperature dynamics that govern salinity can be simulated (Martyr-Koller et al. 2017; Van den Heuvel 2010).

Model Calibration

Model calibration was conducted for a 3-week period in September 2005 (09/08/2005 – 09/30/2005). Table 1 summarizes the input data used by Lin et al. (2018) in the model calibration. The salinity data used for the model calibration was obtained from a 2005 field collection effort when salinity was measured at 10 stations in the Matagorda Bay and Lavaca Bay from September to December 2005 (Evans-Hamilton Inc. 2006). Figure 2 shows the locations of those salinity stations.

Table 1. Input data used in the model calibration simulation (Lin et al. 2018).

Input Data	**Description**	**Station/Source**
Surface wind forcing		NOAA – Port O'Connor (PCNT2) and East Matagorda Bay (EMAT2)
Directional wave spectra		NDBC Station 42019 – located offshore at 70 miles south of Freeport, TX
Water level	Offshore boundary	NOAA – Bob Hall Pier (MQTT2)
	West boundary in Espiritu Santo Bay	MQTT2 (80% linear proportion to the fluctuation related to the mean of water levels at MQTT2)
	East boundary in the GIWW	n/a – non-flow condition
River discharge	Colorado River boundary	USGS – Colorado River near Bay City (8162500)
	Lavaca River boundary	USGS - Lavaca River near Edna (8164000)
	Placedo Creek boundary	USGS - Placedo Creek near Placedo (8164800)
Salinity	Offshore boundary	2005 field data collection effort - Pass Cavallo station (PC) and Bird Island Channel Marker 19 station (BI-BM)
	Colorado River boundary	2005 field data collection effort – Colorado River station (CR)
	Lavaca River boundary	2005 field data collection effort – Lavaca Bay station (LB)
	Garcitas Creek boundary	2005 field data collection effort – Lavaca Bay station (LB)
	Spatially varying initial salinity	n/a – prepared by assigning higher salinity in the Gulf of Mexico and lower value inside the bay system

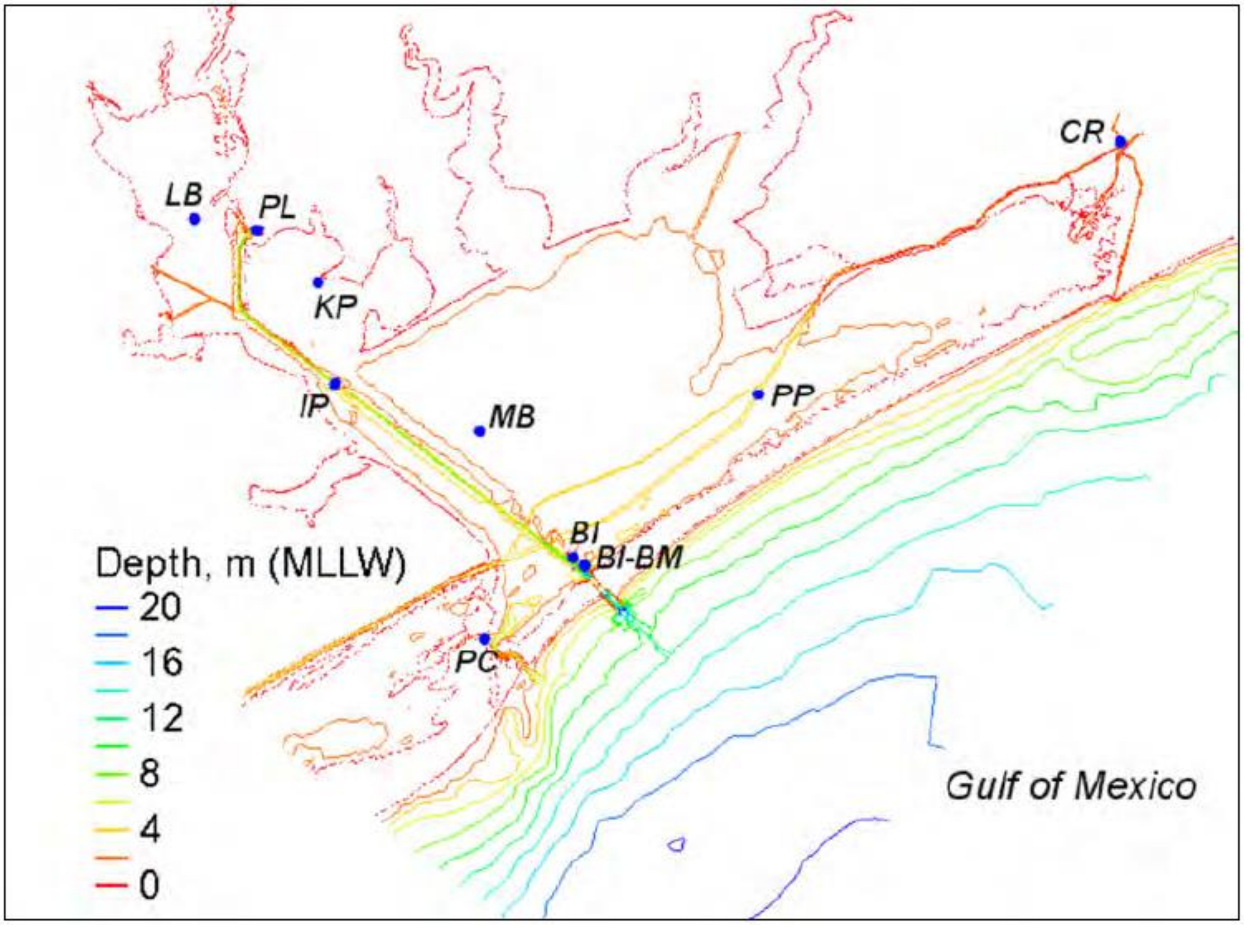

Figure 2 . Matagorda Bay system 2005 field data station locations (Lin et al. 2018, Evans-Hamilton Inc. 2006).

The bathymetry data in the model were updated with the latest bathymetry, DEM, and channel survey data. However, all other input data for model calibration was from 2005. Although the model grid was developed for the earlier studies, an optimized and timelier model could have been achieved if it was calibrated with the latest inputs. The model calibration results, in general, captures the trend of water level fluctuations but there is a detectable phase shift between the modeled and measured water levels at all the measuring stations that is attributed to the input water level data being used from the Bob Hall Pier for the offshore boundary (Lin et al. 2018). It is also mentioned that the difference of magnitude of modeled versus measured water levels is from using the 2017 MSC channel surveys and more recent Lidar data in the model grid (Lin et al. 2018). There is a tide gauge at Matagorda Bay Entrance Channel (NOAA station: 8773767) that has been operating since June 2016 and using the water level data from this station could have given better model performance. Similarly, there are active salinity stations operated by Lower Colorado River Authority (LCRA) in Matagorda Bay (Stations: 6984, 6996, 6990, 6985, NCM2) as well as retired stations operated by the Texas Water Development Board (TWDB) (Stations: MATA, LAVC, TPAL) that have recent salinity data available which could have been used for the salinity validation.

Table 2 shows a summary of statistical metrics for calibration model water levels results at 8 measuring stations, with station BI having the smallest RMSE of 0.108 m (0.36 ft) (Lin et al. 2018). The coefficients of correlation were relatively high and varied between 0.710 and 0.834, but the RMSE obtained for the water levels simulation was always higher than 0.1 m (0.33 ft) indicating the occurrence of slight differences between measured and modeled data. It should be noted that the mean tidal range in the bay is small, approximately 0.22 m (0.72 ft) at Port O'Connor and 0.25 m (0.82 ft) at Port Lavaca.

Table 2. Statistics of calibration model water level results at eight data collection stations (Lin et al. 2018).

Station	**Bias (m)**	**RMSE (m)**	**Correlation Coefficient**
CR (Colorado River)	0.041	0.129	0.818
MB (Matagorda Bay)	0.031	0.133	0.738
LB (Lavaca Bay)	0.011	0.132	0.717
VCAT2 (Lavaca Bay Bridge)	-0.003*	0.122*	0.532*
BI (Bird Island)	0.028	0.108	0.837
BI-BM (Bird Island Channel Marker 19)	0.014	0.109	0.710
PC (Pass Cavallo)	0.041	0.127	0.778
PCNT2 (Port O'Connor)	0.024	0.109	0.834

* Only 2.5-day data available in the statistics.

The model calibration results for salinity modeling show that the model has not captured the salinity variation accurately and this can also be seen in the statistical metrics table (Table 3). Although the bias obtained was quite low reflecting an overall low difference between measured and modeled results, the RMSE obtained was, in general, higher than 1.5 ppt. The negative bias at BI and PC stations that are close to the Gulf of Mexico indicates that the model underpredicts salinity in comparison to the measured values at these stations. Similarly, there was poor agreement between measured and modeled salinity at these stations as shown by quite low correlation coefficients.

Table 3. Statistics of calibration model salinity results at data collection stations (Lin et al. 2018).

Station	**Bias (ppt)**	**RMSE (ppt)**	**Correlation Coefficient**
CR (Colorado River)	0.10	3.82	0.78
LB (Lavaca Bay)	0.25	1.65	0.63
PL (Port Comfort)	0.81	1.71	0.61
MB (Matagorda Bay)	1.44	2.02	0.65
BI (Bird Island)	-0.59	1.08	0.27
PC (Pass Cavallo)	-0.08	1.53	0.17

Discharge data for the Colorado and Lavaca Rivers was used as model input (Figure 3). The black arrow in Figure 3 shows there was a large increase in discharge in the Colorado River between September 10 – 24, and a large drop afterward. Similarly, there was a slight increase in discharge on September 13 in the Lavaca River. The model output shows a little decrease in salinity due to these upticks in river discharge at a station near to the mouth of Colorado River (CR). However, the model might not have properly simulated the transport and mixing of freshwater in the bays, so the general variations expected in salinity with these increases in flow rates cannot be seen in the model output.

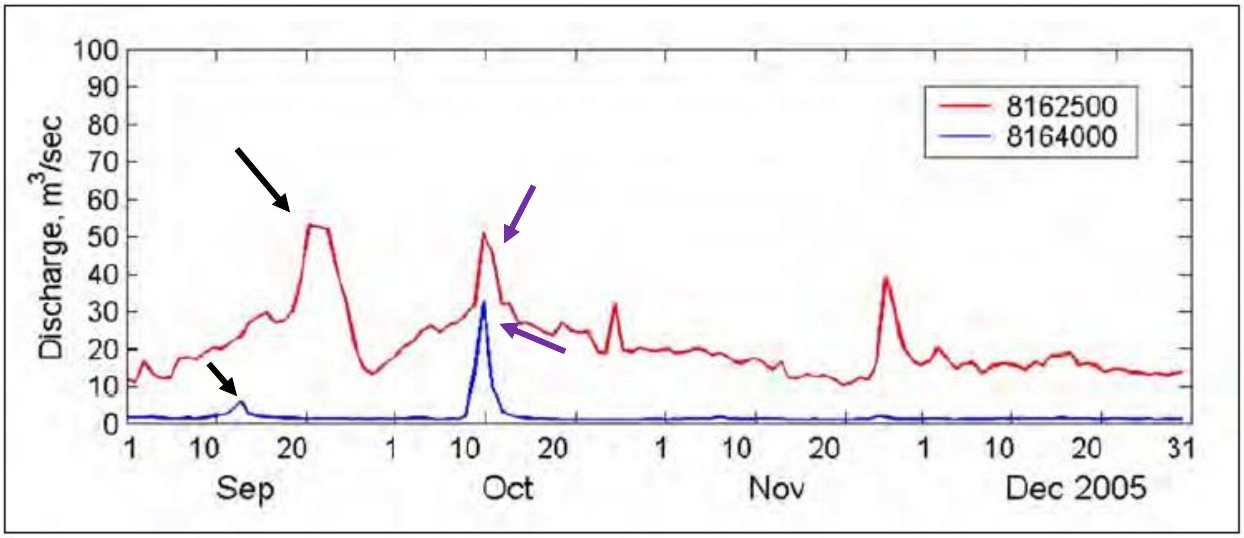

Figure 3 . River discharge input data at Colorado River (gage 8162500) and Lavaca River (gage 8164000) for September to December 2005. The black arrow in the graph is showing high river discharge during the model calibration period in CR and LR and purple arrow during the model validation period (Lin et al. 2018).

Model Validation

The model validation was setup the same as the model calibration. A two-week model validation was conducted for the period 10/01/2005 to 10/15/2005, following the model calibration period 9/10/2005 – 9/30/2005. A total of five-weeks in September-October 2005, therefore, was modeled for both calibration and validation simulations. However, it is recommended to run multiple validation simulations using data covering an alternative period and/or a different location without making any additional adjustment to the model parameters (Williams and Esteves 2017). Furthermore, wind forcing has a significant role in the water level in this area, and there are basically three wind regimes in this area shown in Figure 4 and as follows:

a) The period of November – February characterized by strong winds associated with frontal passages and dominant northerly flows, referred to as the frontal passages period (FPP).
b) The period of March – June characterized by weaker fronts at the beginning of the period and prevalent southeasterlies, referred to as the southeasterly period (SEP).

c) The period of July – October characterized by prevalent southeasterlies but lower in magnitude compared to the SEP, referred to as the quiet period (QP).

Incorporating multiple validation simulations by capturing multiple wind regimes would help finetune the model and potentially result in better model performance and predictions. Furthermore, the CMS model bathymetry was updated with recent lidar and MSC channel survey data, but all other input data were from 2005, which could have contributed to the difference of model results in the validation and calibration simulation. Therefore, multiple validation simulations covering alternative periods with variability of meteorological forcing and using all the latest input data could make the validation more robust with smaller residual errors.

The water level validation simulation has the same issue as in the calibration simulation – there is a detectable phase shift between the measured and modeled water levels at all the measuring stations. The statistical metric table (Table 4) shows similar model performance as in the calibration simulation (Table 2). Table 5 shows the model wave and current results of both the calibration and validation simulation. The table shows that the model wave and current results of the calibration simulation is a little better than the validation simulation. Typically, the required model performance at the calibration and validation stage for wave modeling is judged to be acceptable if the wave model outputs are biased to within ±15° of the mean observed direction (Williams and Esteves, 2017). The bias of validation wave direction is -25.7°.

Apparently, salinity validation was not performed as it is not presented in either the ERDC Letter Report (Lin et al. 2018) nor conference proceedings (Lin et al. 2020a, 2020b). There was a large increase in discharge in both the Colorado and Lavaca Rivers during the model validation period (shown by purple arrows in Figure 3). The issue of transport and mixing of freshwater in the bays as seen in the calibration simulation could be further analyzed if salinity validation results were generated.

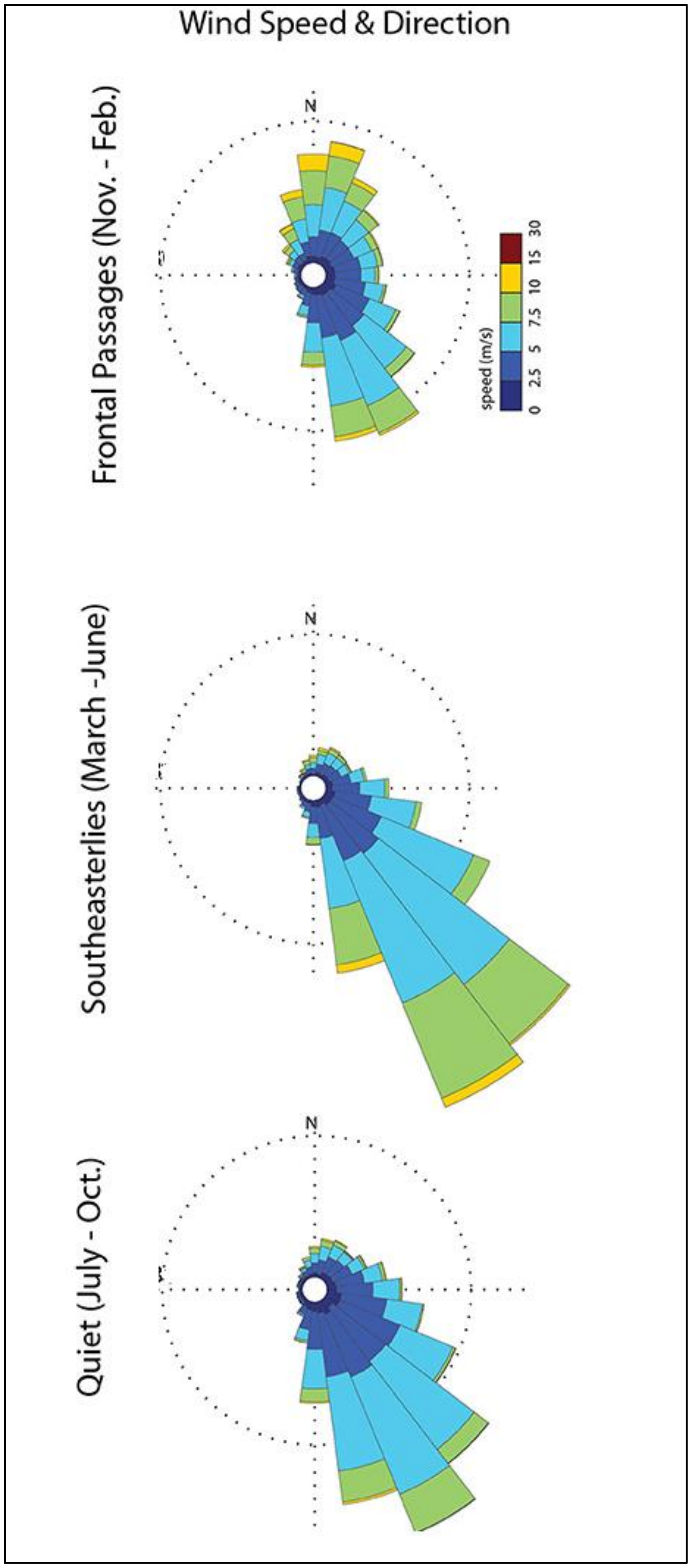

Figure 4 . Wind roses showing wind speed and direction at each wind regime for Corpus Christi Bay (adapted from Reisinger et al. 2017)

Table 4 . Statistics of validation model water level results at six data collection stations (Lin et al. 2018).

Station	**Bias (m)**	**RMSE (m)**	**Correlation Coefficient**
MB (Matagorda Bay)	-0.051	0.123	0.786
LB (Lavaca Bay)	-0.007	0.103	0.858
PCNT2 (Port O'Connor)	0.039*	0.092*	0.695*
BI (Bird Island)	-0.045	0.094	0.868
BI-BM (Bird Island Channel Marker 19)	0.036	0.089	0.879
PC (Pass Cavallo)	0.021	0.093	0.846

* Only 4.5-day data available in the statistics.

Table 5 . Statistics of calibration and validation model wave and current results at MB station.

Wave or Current	**Calibration Simulation (09/10/2005 – 09/30/2005)**			**Validation Simulation (10/01/2005 – 10/15/2005)**		
	Bias	**RMSE**	**Correlation Coefficient**	**Bias**	**RMSE**	**Correlation Coefficient**
Wave Height	-0.02 m	0.09 m	0.84	-0.05 m	0.12 m	0.68
Wave Period	0.08 sec	0.71 sec	0.59	-0.04 sec	0.52 sec	0.49
Wave Direction	13.5°			-25.7°		
Current Magnitude	-0.03 m/s	0.05 m/s	0.42	-0.04 m/s	0.05 m/s	0.32

Modeling Scenarios

The ERDC study (Lin et al. 2018) included three river flow conditions: high, medium, and low flows; and two historical hurricanes: Hurricane Rita (2005) and Hurricane Harvey (2017), with representation of the present and future conditions of ocean boundary water levels. The model representation of present water level was specified by the sea level projection for 2024, which was 2017 water level (MLLW) + 0.061 m (0.2 ft), and the future water level model condition was based on the projection of 2074, which was 2017 water level (MLLW) + 0.573 m (1.88 ft) for the projected sea level rise (SLR). The modeling scenarios included in the ERDC study (Lin et al. 2018) are the existing MSC configuration without deepening/widening project (w/o project) and the modification of the existing MSC with deepening/widening project (with project). Modeling scenarios were categorized into four groups considering present and future water level with and without projects as: PWOP, PWP, FWOP and FWP (Table 6). Therefore, a total of 4 x 5 = 20 modeling scenarios were modeled that are marked with X in Table 6.

Table 6. List of modeling scenarios (Lin et al. 2018).

River Inflow Condition	Present/Future Water Level and MSC Configuration			
	PWOP (present without project)	**PWP (present with project)**	**FWOP (future without project)**	**FWP (future with project)**
Hurricane Rita (2005)	X	X	X	X
Hurricane Harvey (2017)	X	X	X	X
High river flow	X	X	X	X
Medium river flow	X	X	X	X
Low river flow	X	X	X	X

Apart from the salinity inputs along the offshore boundary and river boundaries, all other inputs for the scenario modeling were from the same stations as in the calibration simulation. Table 7 shows the simulation period and five river flow input conditions for the scenario modeling.

Table 7 . Scenario model simulation period and river inflw conditions (Lin et al.,2018).

River Inflow Condition	**Simulation period**	**River inflow discharge***	**Wind and water level input ***
Hurricane Rita (2005)	8 - 30 Sep, 2005	field data	field data
Hurricane Harvey (2017)	10 Aug – 10 Sep, 2017	field data	field data
High river flow	10 Feb – 10 Mar, 2017	field data	field data
Medium river flow	1 July – 20 Aug, 2017	54.0 m3/sec	field data
Low river flow	1 July – 20 Aug, 2017	20.5 m3/sec	field data

Although future sea-level rise (SLR) was applied in the scenario modeling by incorporating future water levels based on the projection of SLR for 2074, the ERDC (Lin et al. 2018) study has not incorporated the potential coastal morphological changes due to SLR. Rising sea level can cause permanent changes in the landscape when it inundates low-laying land, and the study region is particularly vulnerable because of low elevations and eroding shorelines. Therefore, incorporating the landscape change due to the SLR in the future scenario modeling could have resulted in different estimates. For instance, Figure 5 shows a change in landscape due to 0.535 m (1.76 ft) of relative sea level rise (RSLR) in 2075 along the west side of Matagorda Bay in the Powderhorn Lake area modeled using Sea Level Affecting Marshes Model (SLAMM) for the Texas Coastal Resiliency Master Plan (TCRMP, Levitz 2018). A significant increase in salt marsh and open water can be seen in the future landscape map in comparison to the present landscape with a similar projected SLR in 2075 (Figure 5).

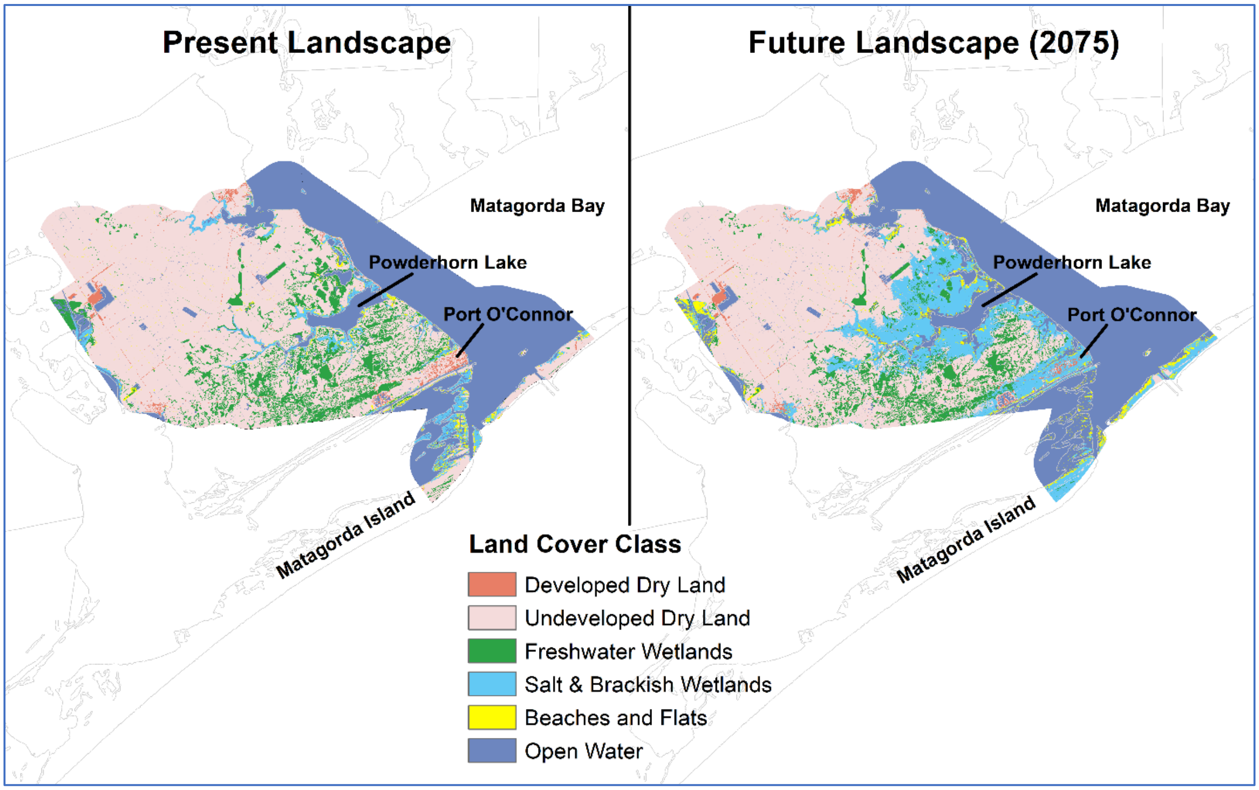

Figure 5 . Present landscape vs. future landscape in 2075 with 0.535 m of RSLR in the Powderhorn Lake area. The present landscape is based on the US FWS National Wetland Inventory (NWI) dataset and the future landscape is developed using SLAMM.

Circulation and Salinity Change Modeling Results

The ERDC study (Lin et al. 2018) analyzed modeling results by comparing differences between with and without channel improvement projects in present and future water level conditions, i.e., PWP – PWOP and FWP – FWOP (Table 6). The modeling results show that the difference of predicted water levels in Matagorda Bay with and without channel improvement projects in both present and future water level conditions is within ± 0.05 m (0.16 ft) for the Hurricane Rita, high, medium, and low flow conditions, and is within ± 0.5 m (1.64 ft) for the Hurricane Harvey simulation. Although it is mentioned in the Lin et al. (2018) report that the difference of ± 0.05 m (1.96 inch) is small, it is significantly higher than a similar study conducted in Corpus Christi Bay (Subedee and Gibeaut 2021). However, a 0.05 m (5 cm, 1.97 in) increase in the water level could shift intertidal marsh and tidal flats to open water environments.

The modeled water level difference in the ERDC study (Lin et al. 2018) is shown spatially with difference maps of PWP – PWOP and FWP – FWOP (Lin et al. 2018: Figures 4-2 to 4-10), and with bar graphs showing modeled water levels in PWP, PWOP, FWP and FWOP at some stations (Figures 4-11 to 4-13). Along with these maps and bar graphs, the water level hydrographs showing tides at different stations or points in the bay over the simulation period could have been presented, which would have shown how water level changes over time under different channel scenarios at these stations/points. These hydrographs would also have shown the changes in tide range that would have given more information about the changes in amplitude on daily high and low tides under different channel configurations.

The modeling results show that the difference of average current speeds between PWP and PWOP, and FWP and FWOP is within ±0.1 m/sec (0.33 ft/sec) (Lin et al. 2018). The maximum current can be 0.2 m/sec (0.66 ft/sec) stronger with the dredging project than without project for Hurricane Rita and three river inflow conditions, and 0.5 m/sec (1.64 ft/sec) for Hurricane Harvey. Although it is characterized as a small change in the Lin et al. (2018) Report, the current speed of ±0.1 m/sec (0.33 ft/sec) is enough to keep fine-grained sediment suspended and potentially increase turbidity, and ±0.5 m/s (1.64 ft/sec) can lead to enhanced shoreline and channel erosion (Biggs 1978, Sundborg 1967). Similarly, the currents along and near MSC are slightly stronger with the project than without the project scenarios.

Salinity modeling results show that the difference of model salinity fields for both PWP – PWOP and FWP – FWOP are less than 2 ppt excluding the placement areas (Lin et al. 2018). Although it is characterized as a small increase in the Report, the long-term average for salinity in the entire Lavaca-Colorado Estuary is 20 ppt (Montagna et al. 2018), so this increase in salinity is nearly a 10% change, i.e., 2 divided by 20 ppt. This is actually a large change that could alter water quality and secondary production. An ecological model was created to predict system-wide secondary production for benthic organisms in response to different freshwater inflow regimes that could result in diversions of freshwater from the Colorado River (Kim and Montagna 2009, 2012). A 10 % increase in long-term average salinity would have no effect on deposit feeders such as polychaete worms but would decrease secondary production of suspension feeders such as mollusks and crustaceans with effects being stronger in Lavaca Bay than Matagorda Bay.

For medium and low river inflow simulations, the salinity is generally higher in Matagorda Bay with channel improvement project than without project (Lin et al. 2018). Similarly, model results also show that higher salinity occurs more often in the bay with the future water level than the present water level. Therefore, if river discharge to the bay decreases in the future with climate change or increased diversions, the channel improvement project causes an increase in salinity, and a detrimental cumulative effect may occur. It is not mentioned in the Lin et al. (2018) Report if the salinity is for surface or bottom salinity, which is important because bottom salinity values are larger than or equal in magnitude to the surface salinity values due to the density differential of salt water.

Additional Modeling

It is mentioned in the Report (Lin et al. 2018) that more modeling studies are recommended to increase understanding of sediment transport and the potential increase of the channel shoaling rate due to the channel improvement projects. A total of four shoaling rate calculation methods were used and all these methods rely on bathymetric surveys available during 2011 - 2017. Among the four methods, the Corps Shoaling Analysis Tool (CSAT) seems to be the most comprehensive, and it was used to estimate annual shoaling rates along MSC. CSAT uses channel boundary information, hydrographic survey datasets and historical dredging records to compute shoaling rate used to predict future dredging volumes but does not include effects of the hydrodynamics and waves. Therefore, a numerical sediment transport model that incorporates hydrodynamics and waves could help not only to get accurate shoaling rates but also to understand the impacts on long-shore sediment transport due to the channel improvement projects.

The independent external peer review (IEPR) done by Battelle Memorial Institute (2018) has also pointed out the issue related to the calculation of shoaling rates in the study. Similarly, the IEPR report has raised concern about ship channel side slopes indicating that the side slopes in select reaches of the channel are too steep for the submerged granular soil to stand up. However, it appears that a slope stability analysis was performed for the final FR/EIS study (USACE 2019), which determined that the slopes are stable enough and the same slope was used in the modeling study (Lin et al. 2018).

Finally, the ERDC modeling effort by Lin et al. (2018) did not investigate the impacts of the channel improvement projects on storm surge water levels and inundation duration patterns in and around Matagorda Bay. It is important to assess how hurricane storm surge may change in the bay with modified channels. The FR/EIS (USACE 2019) has acknowledged likely increases in the frequency and intensity of tropical storms due to future climate change along the extensive Texas coast. Therefore, it is very important to conduct hydrodynamic storm surge modeling to evaluate coastal storm surge impacts in and around Matagorda Bay due to the proposed ship channel improvement projects.

Table 8 summarizes the issues in each component of Lin et al. (2018) modeling study and our recommendations to improve the predictions of water levels, currents, and salinity. It also summarizes additional modeling studies necessary for a holistic assessment of the impacts of the channel improvement projects in and around Matagorda Bay.

Table 8. Summary of the issues found in the ERDC modeling study (Lin et al. 2018) and recommendations for further studies.

Model Component	Critique/Comment	Recommendation
Model Development	• The CMS model used in the ERDC study was developed for previous studies with a small modification. • The CMS model used in the ERDC study was applied in 2D version for the salinity modeling. Multiple modeling studies have shown that hydrodynamic transport in the estuaries can be well-described by 2D models, whereas salinity dynamics and transport in the estuaries are governed by 3D processes such as estuarine circulation and temperature dynamics, which are unrealistic to simulate in a 2D model.	• A 3D model with a heat flux module for the calculation of temperature–salinity dynamics could have given better salinity variation.
Model Calibration / Validation	• The CMS model was updated with the latest bathymetry and channel surveys but all other inputs in model calibration/validation were from 2005. • A detectable phase shift between the modeled and measured water levels at all the measuring stations was observed due to the offshore water level input from Bob Hall Pier station. The salinity variation was not accurately captured in the salinity calibration simulation. The residual errors are high in some stations and errors seem systematic too. • Only a 3-week calibration and 2-week validation simulation in September/October 2005 was done so calibration/validation has not captured multiple meteorological variations. • The influence of significant increase in river discharge during the calibration/validation period cannot be seen in model output as expected. • Only salinity calibration results were presented but salinity validation results were not shown in the report.	• The model could be calibrated/validated using recent datasets available from multiple stations near to the MSC that are newly installed and active or have already retired but still have recent data. • Multiple validation simulations covering alternative periods with different wind regimes and incorporating recent input data could have made validation more robust and have improved the model performance.

Model Component	Critique/Comment	Recommendation
Modeling Scenarios	• Future sea level rise (SLR) was applied in the scenario modeling just by raising input water level along the offshore boundary, but potential coastal morphological and land cover changes due to the SLR have not been incorporated in the model. SLR can cause permanent changes in the landscape when it inundates low-laying land, and the region is particularly vulnerable because of low elevations and eroding shorelines.	• It is predicted that there will be a significant increase in salt marsh and open water in and around Matagorda Bay even with 0.573 m of projected SLR in 2074 used in this study. Therefore, the potential landscape changes due to the SLR in the future scenario modeling could have given different results.
Modeling Result	• The difference in water level with and without channel improvement projects is ±0.05 m which is significantly higher than similar study in Corpus Christi Bay. • The maximum current speeds with and without channel improvement projects is within ±0.1 m/s for all three river flow conditions, and ±0.5 m/s for Hurricane Harvey inflow condition. However, current speed of ±0.1 m/s is enough to keep fine-grained sediment suspended for longer leading to increased turbidity, and ±0.5 can lead to enhanced shoreline and channel erosion. • It is not mentioned if "salinity" in this study means bottom or surface salinity. However, the difference of model salinities with and without projects is less than 2 PPT that might not have significant consequence now but could be a serious problem in future as climate change can have compounding effects.	• It will be beneficial to see the impacts in tide range along different points in the bays with the water level hydrographs for further analysis.

Model Component	Critique/Comment	Recommendation
Additional Modeling	• It is mentioned in the ERDC study that more modeling studies are needed to understand sediment transport and potential increase of channel shoaling rate due to the channel improvement projects. The Corps Shoaling Analysis Tool (CSAT) was used to estimate annual shoaling rate along MSC. CSAT uses channel boundary information, historical channel surveys and dredging records to compute shoaling rate and predict future dredging volumes, but it does not include effects hydrodynamics and waves. • A very important missing component in this modeling study is an investigation of impacts of the channel improvement projects on storm surge water levels and inundation duration patterns in and around Matagorda Bay. The USACE (2019) study has acknowledged increase in the frequency and intensity of tropical storms due to future climate change along the extensive Texas coast, but a hydrodynamic storm surge modeling effort was not conducted to find any additional inundation due to the penetration of storm surge by increased channel depth and width.	• A numerical sediment transport modeling that incorporates hydrodynamics and waves could help not only to get accurate shoaling rates but also to understand the impacts on long-shore sediment transport due to the channel improvement projects. • A few representative storms of different characteristics can be simulated with and without channel improvement projects that will assess how hurricane storm surge may change in the bay with modified channels. This hydrodynamic storm surge modeling can help evaluate storm surge impacts (e.g., additional inundation area, inundation duration pattern etc.) in and around Matagorda Bay.

Groundwater and Hydrogeology

The Gulf Coast Aquifer is a shallow lens of freshwater beneath the bay. Deepening the Matagorda Bay ship channel may potentially impact local hydrology and groundwater. Dredging bay sediments risks breaching underlying confining layers which may disturb natural pressure and flow fields in the shallow subsurface or allow increased communication between previously confined groundwater and bay surface waters (Duncan 1972, Lautier 1998, Moore 1999, Teatini et al. 2017). Additionally, removal of contaminated bay sediments may place other resources (i.e., groundwater) and habitats (i.e., wetlands or bird islands) around the dredge spoil placement area at risk. Where the aquifer is in direct contact with the bay, water exchange will occur. This groundwater-surface water interaction may result in salinization of the aquifer (i.e., saltwater intrusion) or focus fresh groundwater discharge at the breach (George et al. 2020), depending on hydraulic heads (i.e., water table or potentiometric surface > sea level = groundwater discharge to the bay, water table or potentiometric surface < sea level = saltwater intrusion) and density differences (i.e., saltwater is more dense than freshwater and will sink into the aquifer) (see Santos et al. 2012 for more detailed discussion of mechanisms of groundwater-surface water interaction). Saltwater intrusion into coastal aquifers may render groundwater non-potable as well as facilitate chemical reactions (e.g., ion exchange and oxidation-reduction reactions) which chemically alter the intruding sea water and cause enrichment in radium, barium, and nutrients (Moore 1999). Furthermore, interruption of the hydraulic connection, and the associated nutrient inputs, to the seafloor farther offshore could negatively impact benthic communities at the base of the food chain (Pinkney 2018).

Groundwater Case Studies

The potential impacts of ship channels and dredging projects on groundwater-surface water interactions and coastal hydrology may be inferred from other studies.

- Duncan (1972) documented breaches in the confining unit above the Floridan aquifer that correlated with dredged channels and turning basins in Port Royal Sound using seismic surveys. Where the principal artesian aquifer is exposed to the saltwater of the estuary, continued heavy groundwater withdrawals from the aquifer will lead to a gradual acceleration of salt-water intrusion into the upper permeable zone of the principal artesian aquifer.
- The Houston Ship Channel was identified as the source of saltwater that was detected in shallow wells between Baytown and Houston, Texas, with analysis of inorganic constituents, dissolved organic carbon, and tritium in groundwater and surface waters (Jorgensen 1976). Estimates of the vertical leakage indicated that the saltwater intrusion from the surface water to the Chicot aquifer, was not sufficient to substantially alter the inorganic chemistry of the groundwater.
- In coastal Louisiana, simulation modeling of channels and canals found that under similar conditions (e.g., freshwater inflow, wind speed and direction, tidal range), saltwater intrudes farther inland in large and deep channels than in smaller and shallower channels (Wang 1988). The 5-ppt isohalines were found to be 20 to 75 km (12.4 to 46.6 mi) from their respective channel entrances. Computer simulations showed that deepening and widening the channels increases saltwater intrusion and changes patterns of salinity distribution in channels with more pronounced effects in large and deep channels than in smaller and shallower ones. Furthermore, this saltwater intrusion may cause marsh loss/retreat.
- A modeling study tested the significance of removal of confining beds by dredging and resultant increase in surface area of aquifer materials in conjunction with present (1993-1994) and projected future (2020) pumping rates from nearby municipal and industrial well systems in Wilmington Harbor (Lautier 1998). The slight decrease in groundwater level in the future simulation was deemed to be due to increased pumping and not by channel deepening.
- In southern Brazil, Santos et al. (2008) found that dredging the upper layer of soils in rice irrigation canals removed the surface confining layer, thus, exposing the deeper, more permeable sandy layers. The removal of the surface confining layer increased the

groundwater fluxes to the lagoon by two orders of magnitude compared to the beaches along the lagoon shoreline.

- In coastal Baldwin County, Alabama, Murgulet and Tick (2008) identified direct infiltration of surface water from the Intracoastal Waterway and saltwater ponds located along the Intracoastal Waterway as a source of saltwater intrusion and contamination of the shallow aquifer.
- The USGS assessed the potential effects of navigation channel deepening in the St. Johns River on salinity in the adjacent surficial aquifer system (Bellino and Spechler 2013). Simulated results modeled a range of plausible hypothetical conditions and indicated dredging would have little to no effect on salinity variations in the high use areas upstream of the proposed dredging activities. However, given the models did not necessarily simulate actual conditions, long-term monitoring of groundwater levels and water quality were recommended to determine if any salinity changes occur within the aquifer after deepening the channel.
- An assessment of the hydrogeological effects of dredging navigable canals through shallow lagoons in Venice found that dredging can perturb the natural pressure and flow fields in the shallow subsurface area around the channel through tidal effects and ship-wake effects (Teatini et al. 2017).
- Preliminary results from a study that combined geochemical and geophysical surveys in the urbanized Wismar Bay, southern Baltic Sea, shows that the bottom sediments in the central bay have been impacted locally by excavation and shallow geologic units have been changed (von-Ahn et al. 2021). The excavation reduced the shallow geologic thickness of the aquitard promoting freshwater discharge to surface marine sediments observed as a freshened porewater with a water isotope composition close to the local meteoric water line of the southern part of the German Baltic Sea. This suggests discharge of relatively modern freshwaters. These early results indicate that artificial infrastructures, such as sediment channeling, may ease hydrological connections between coastal aquifers and coastal bottom water increasing submarine groundwater discharge.

Hydrogeologic Units and Geologic Setting

The Lavaca-Colorado Estuary overlies the Gulf Coast Aquifer (GCA). The GCA is a major aquifer paralleling Texas' Gulf of Mexico coastline. The aquifer is used for municipal, industrial, and irrigation purposes (George et al. 2011). It consists of several confined and unconfined aquifers. A large portion of the water contained within the GCA is used for irrigation and drinking water purposes by the rapidly growing coastal communities (Chowdury and Turco 2006). From the land surface downward, the GCA is comprised of the following hydrogeologic units: the Chicot aquifer, the Evangeline Aquifer, the Burkeville confining unit, the Jasper aquifer, and the Catahoula confining unit (Baker 1979, Kasmarek 2013). The Chicot aquifer comprises (youngest to oldest) the alluvium, Beaumont Formation, Montgomery Formation, Bentley Formation, and Willis Formation. All of the sedimentary units thicken toward the Gulf of Mexico. Thus, while there is little to no risk from dredging of puncturing the major confining units of the GCA, the Lavaca-Colorado Estuary is in contact with the Chicot aquifer's alluvium and Beaumont Formations and local confining layers may be at risk. In the upper 30.5 m (100 ft) of the Beaumont Formation near Lavaca-Matagorda estuary, three primary saturated sand and silt zones have been identified with intervening clay layers (Scanlon et al. 2005). In the coastal counties, the water table is typically 4.3 - 6.1 m (14 - 20 ft) below the surface and water-table conditions (i.e., unconfined aquifer exposed to atmospheric pressure only) occur principally at depths less than 30.5 m (100 feet) (Hammond 1969, Scanlon et al. 2005).

The freshwater part of the aquifer extends beneath the Gulf of Mexico to the southeast (Bruun et al. 2016). Freshwater saturated thickness in the GCA averages about 305 meters (1,000 ft). The maximum total sand thickness ranges from 396 m (1,300 ft) in the northeast to 213 m (700 ft) in the south. Lavaca-Matagorda estuary marks a transition point for groundwater salinity (Bruun et al. 2016, Young et al. 2010). Northeast of the estuary groundwater is predominantly fresh whereas southwest of the estuary groundwater becomes more brackish to saline. The hydraulic conductivity (the ability of a fluid to pass through the pore spaces and fractured rock) of the aquifer also decreases from 2.13 m/d (7 ft/d) in the northeast to 0.3 m/d (1 ft/d) in the south. The transmissivity (the ability of the aquifer to transmit groundwater throughout its entire saturated thickness) of the aquifer ranges from over 1,301 m^2/d (14,000 ft^2/d) in the northeast to less than 93 m^2/d (1,000 ft^2/d) in the south (George et al. 2011). The GCA discharges the most groundwater to surface water of all Texas aquifers, with an estimated 3.8 million acre-feet per year (Bruun et al. 2016).

Groundwater seepage is often patchy, diffuse, temporally variable, and may be associated with a complicated assortment of confined, semi-confined, and unconfined aquifers (Santos et al. 2012). The geology of the GCA is complex due to cyclic deposition of sedimentary facies (Chowdury and Turco 2006). The recurring sedimentation facies of the GCA were created by changes in land-surface subsidence of the depositional basin and sea-level transgressions (i.e., rise) and regressions (i.e., decline) (Kasmarek and Robinson 2004). Repeated sea-level changes and natural basin subsidence created discontinuous beds of sand, silt, clay, and gravel. Fluvial-deltaic processes deposited continental sediments during periods of sea-level decline whereas continental sediments were altered and marine sediments were deposited during periods of sea-level rise. Due to this complex depositional process, the aquifer facies alternate between the predominantly continental sediments that compose the aquifers and the predominantly marine sediments that compose the confining units and clay layers within aquifers. Thus, the GCA system has a high degree of heterogeneity in both lateral and vertical extents and groundwater may be locally confined by interbedded silt/clay lenses less than 1.8 m (6 ft) thick. Dredging bay sediments risks puncturing one of these interbedded confining lenses within the surficial aquifer creating contact between bay surface water and previously confined groundwater. The potential impacts of dredging on the surficial aquifer will be site specific due to the interlayered, discontinuous sequences of clay, silt, sand, and gravel formed by these depositional processes.

At the ALCOA site, three groundwater-bearing zones were identified that occur at relatively consistent depths: Zone A at an elevation of 1.5 to -0.3 m mean sea level (5 to -1 ft msl), Zone B at -6.1 to -9.1 m msl (-20 to -30 ft msl), and Zone C -14.9 to -20 m msl (-49 to -66 ft msl) (EPA

2001, Scanlon et al. 2005). Zone A discharges to the bay system at the shoreline where it crops out. Zone B discharges to the bay system in the existing ship channel and turning basin where the deep channel intercepts Zone B. Zone C does not currently outcrop in the bay system. Assuming the depth to Zone C remains consistent under the bay, the proposed deepening of the ship channel to ~13 m (~44 ft) with an allowable over-depth of 0.61 m (2 ft) leaves ~1 m (~3.3 ft) of confining sediment in place and, thus, should not intersect this groundwater-bearing layer. However, thinning the confining clay layer may enhance groundwater discharge to the bay (von-Ahn et al. 2021). Groundwater discharges upwards through strata offshore from both Zone B and Zone C. Similar layering of shallow groundwater-bearing zones may occur to the south of the ship channel. The location of potential breaches and the direction of flow at these breaches and degree of interaction requires further assessment for the Lavaca-Matagorda Estuary.

The Lavaca-Matagorda estuary surficial sediments are predominantly fine-grained mud (White et al. 1989). These middle bay muds are ~10 m (~33 ft) to ~15 m (~49 ft) thick (Anderson et al. 2016). This low hydraulic conductivity mud "cap rock" within the estuary likely acts as a confining layer that may limit groundwater discharge to the bay or redirect groundwater flow along the paleovalley margins (Spalt et al. 2018). The existing ship channel is ~11 m (~36 ft) deep, which means it may have already completely removed the mud cap in places. The proposed deepening of the ship channel would increase the depth of the channel to ~13 m (~44 ft), potentially exposing a larger area of the underlying aquifer. The low relief/elevation of the coastal plain creates small hydraulic heads. Thus, hypothetically, in dry years/drought conditions, when the water table lowers saltwater intrusion into the aquifer could potentially be exacerbated by the increased connections between the surface water and groundwater, like the findings of Duncan (1972). Alternatively, further exposure of the underlying aquifer could greatly increase the volume of groundwater discharging into the bay, like the findings of Santos et al. (2008).

Finally, concerns for the mobilization of contaminants from the dredge spoils into the groundwater raised by Battelle (2018) in the final IEPR report should be addressed. Mercury has an affinity for clay, iron oxides, and organic matter in sediments (Różański et al. 2016) and the highest concentrations of mercury have been observed in Lavaca Bay at the end of the Matagorda Ship Channel near the ALCOA site where clayey mud is the dominant sediment type (see *Mercury Mobilization* section below). While transport of mercury by leaching to the subsurface and groundwater is often thought to be insignificant, the influence of mercury speciation on transportation rates within soils is not well constrained and varies greatly between sites and within sites (O'Connor et al. 2019 and references therein). Factors that influence mercury transport in soils and porewaters include: the availability and solubility of different mercury species (e.g., organo-Hg compounds have low solubility whereas methylmercury hydroxide is highly soluble), the properties of the soil solution (e.g. presence of organic matter or excess sulfide ions, Eh and pH conditions), composition of organic matter (e.g., HgS may be soluble in waters containing fulvic acids or thiol ligands whereas humic substances may inhibit this process), and mineral composition of the sediments (e.g., high clay content of soil may reduce mercury transport due to mercury sorption to clay mineral surfaces or reduced permeability, mineral kaolinite particles in porewater may resorb mercury from sand through competitive sorption and act as a carrier in soil pores). Though it is likely only the shallowest unconfined unit of the Chicot aquifer would be at risk, a site-specific assessment of the mercury speciation and sediment and groundwater/porewater properties would be necessary to address the concerns posed by the dredge spoil placement.

Ecological Assessment

The location of the project area is taken from Slide 8/Page 8 entitled "Placement Plan" from a public presentation by Corragio Maglio (Figure 6, Maglio 2019). This "plan" is used in the habitat overlays below.

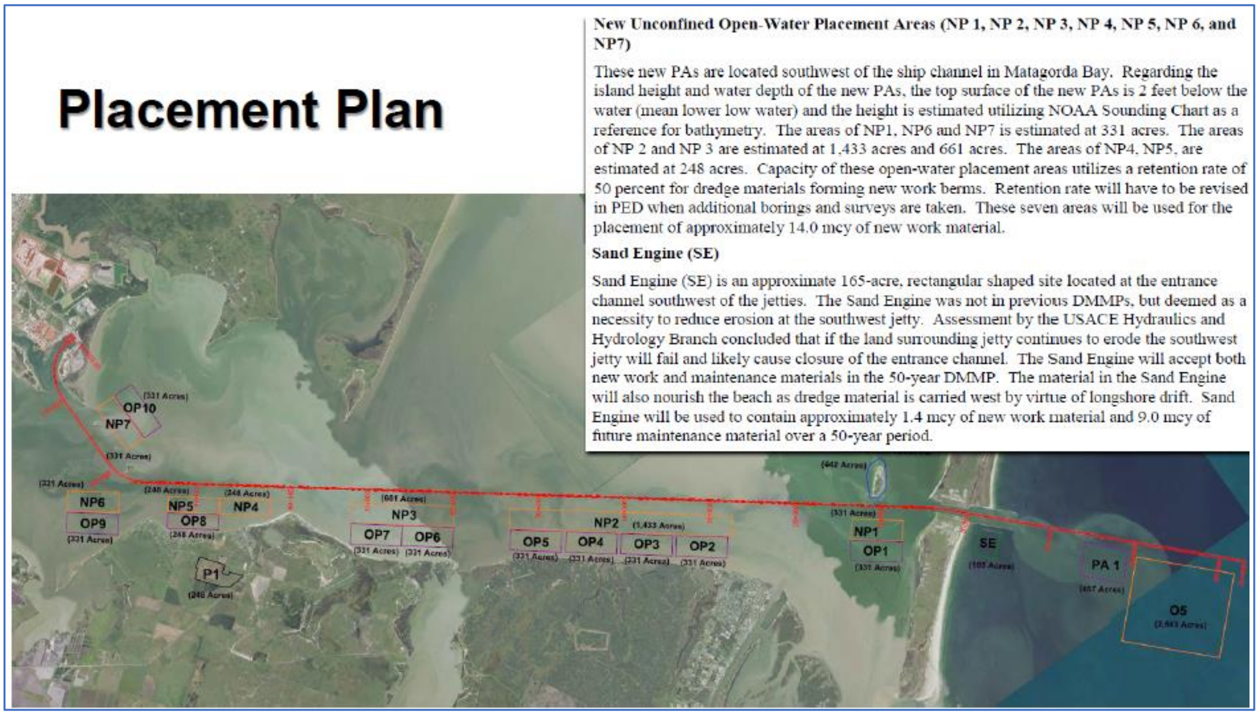

Figure 6. Proposed channel, basin, and placement areas (Maglio 2021) used in all habitat comparisons.

The Mitigation Plan is also presented in the presentation (Maglio 2021) and the study (USACE 2019, Appendix B, Section 5).

Table 9. Habitat mitigation plan (Maglio 2021,USACE 2019.

Habitat Type	Acreage Impacted	Acreage Created	Responsible Action
Oyster Reef	129.2		Dredging/Placement
Bay Bottom	3927		Placement
Offshore Bottom	2053		Placement
Oyster Reef		130	Mitigation

Habitats

The proposed plan to deepen and widen the Matagorda Bay ship channel may impact important habitats including oyster reefs, seagrass beds, and the open bay bottom. The physical removal of the substrate and the deposition of material are the most likely direct effects of dredging projects (Erftemeijer et al. 2012). Dredging activities may affect habitats indirectly as a consequence of increased sedimentation, which can potentially bury/smother benthic communities (Maurer et al. 1981, Jones and Candy 1981, Wilber and Clarke 2001, Fraser et al. 2017). When considering a dredging operation, it is important to think about the resources that may be impacted. Valued ecosystem components (VECs) are resources that have been identified as having aesthetic, ecological, economic, or social value (Montagna et al. 2013). These resources include species that create habitat for other organisms (foundation species) and are iconic to an area such as oyster reefs and seagrass beds. Some resources may be more sensitive to increases in sedimentation than others (Erftemeijer et al. 2012). Ecosystem services are the benefits provided by the environment to human health and well-being (Costanza et al. 1997). For example, seagrass beds provide services such as coastal protection, erosion control, water purification, tourism, education, recreation, and research (Barbier et al. 2011).

Oyster reefs

Oyster reefs are valued for the ecosystem services they provide (Grabowski and Peterson 2007). Oyster reefs supply habitat for marine species (Grabowski et al. 2005), serve as a foraging ground for juvenile fish species (Coen et al. 1999), reduce erosion by acting as natural breakwaters (Meyer et al. 1997, Scyphers et al. 2011), and act as a biological filter to clean the water and process estuarine substances (Dame et al. 1984). Oysters are also recognized for their economic importance as both a commercial and recreationally harvested species (Pollack et al. 2013). The concern for oyster habitats being impacted by dredging activities stems from the catastrophic loss of oyster reefs to non-dredging related sedimentation events (Wilber and Clarke 2001). Sedimentary particles can obstruct the oysters' filtering apparatus, which would require a significant amount of metabolic energy to remove (Volety and Encomio 2006). This can result in death by smothering or hypoxia or hypoxia/anoxia, decreased fitness because of diminished reproductive energy allocation, or poor health due to immunologic stress. (Volety and Encomio 2006).

Sedimentation has been measured on intertidal oyster reefs in South Carolina (Hadley et al. 2010). Five sites of multiple-aged reefs were sampled using oyster population parameters. Sediment coverage was negatively correlated with abundance and overhead photographs demonstrated the reduction of oyster habitat that occurs from sedimentation.

Surveys were conducted on the oyster bottom in Chesapeake Bay to document the sedimented oyster habitat (Smith et al. 2005). Researchers performed 16 surveys that covered a total of 39 km^2 (15 mi^2) of bottom that was classified in 1911 as supporting oyster populations. Over 90% of this area has degraded from productive oyster bottom to mud, sand, or heavily sedimented oyster shell (Smith et al. 2005). Larval oysters prefer to settle on hard substrates and adult oyster shell which results in the formation of an oyster reef. However, the presence of sediment on an oyster reef may inhibit larval recruitment (Mackenzie 1983). Severe sedimentation in Chesapeake Bay rendered the area unsuitable for oyster habitat and recruitment.

Field experiments were conducted in the lower Chesapeake Bay to examine eastern oyster populations and the relationship between reef height and sedimentation (Colden et al. 2017). Reefs higher than 0.3 m (1 ft) supported greater oyster density and survival. There was also less sediment deposition. Reefs lower than 0.3 m (1 ft) experienced heavy sediment deposition and were ultimately buried (Colden et al. 2017)

The oyster habitat located in Lavaca Bay is in the path of the proposed dredging site (Figure 7). TPWD (2004) used sidescan sonar data to determine where hard bottom surface structures were in Lavaca Bay. While oyster reefs were the most common highly acoustically reflective surface, oyster covered pilings, oyster covered pipelines, and hard surface sand deposits also displayed a high reflectance signature. Sub-bottom profiler data was consulted in hard bottom areas and signature verification cruises were conducted to verify oyster reef signatures. This study mapped a total of 3,249 acres of oyster reefs in Lavaca Bay. Some of the major reefs mapped were Lap Reef, Gallinipper Reef, San Point Reef, Keller Reef, and Middle Ground Reef. There were also numerous small, mainly linear patch reefs that had developed, particularly along the submerged spoil banks adjacent to the dredged ship channels. TGLO (2004) used acoustic equipment and a poling technique in shallow areas to determine the location of the oyster reefs.

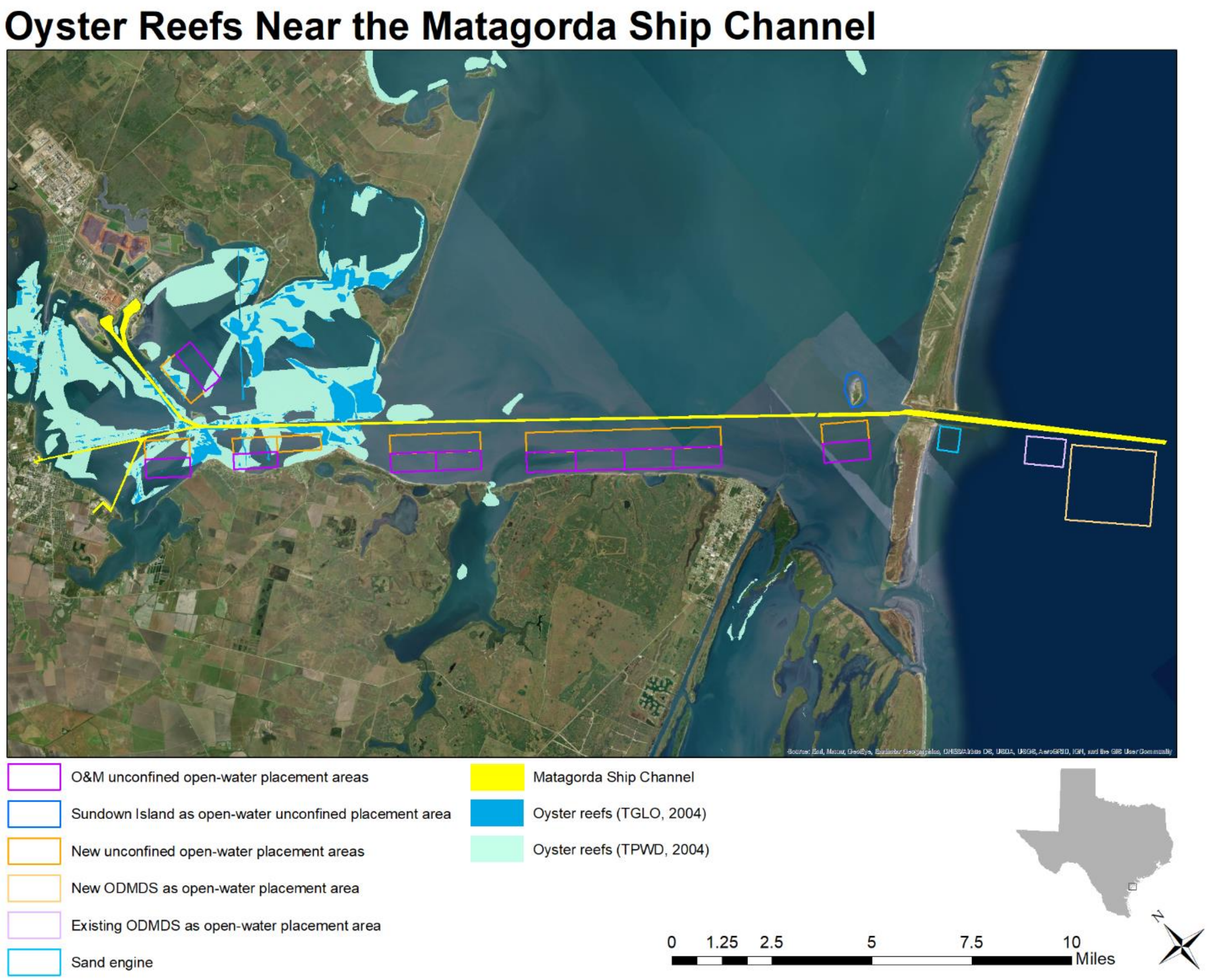

Figure 7. Oyster reef habitat overlaid with dredging and placement plans (Maglio2021).

The oyster reefs may be directly impacted by construction or indirectly by high sediment loads (Figure 8). A total of 838.63 acres of oyster reef habitat overlaps with the outline of the new channel and new and existing spoil placement areas.

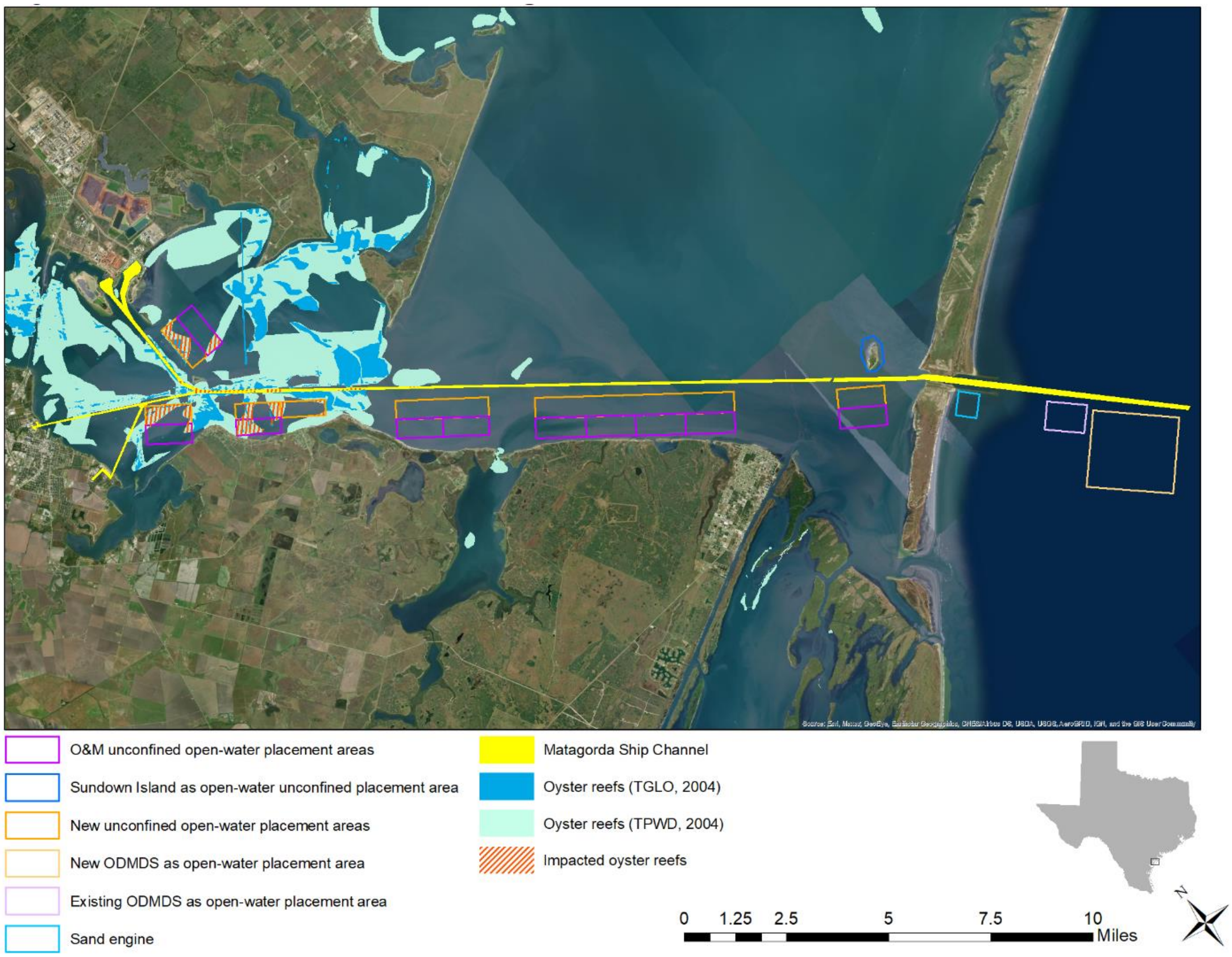

Figure 8. Oyster reef habitat within the project area.

Seagrass beds

Seagrass provides important ecosystem services. Benefits include stabilizing the sediment, providing food and shelter for marine life, and oxygenating the water column. Seagrasses are highly sensitive to changes in water quality, sediment loading, and other inputs that accumulate due to the alteration of coastal water bodies (Dennison et al. 1993). Therefore, seagrasses can be useful for identifying changes in environmental thresholds that are triggered by dredging operations. Seagrasses can be directly affected at the dredge and disposal sites, when they are often physically removed or buried, or indirectly by changes in water quality or bathymetric changes, which may sometimes occur because of dredging activities (Erftemeijer and Lewis 2006).

There are five species of seagrass that exist along the Texas coast. Shoalgrass (*Halodule wrightii*) is the most common seagrass along the coastline and is usually found in water is about 0.3 – 1 m (1 - 3 ft) deep. Star Grass (*Halophilla engelmannii*) is a small seagrass that is found in mixed beds with other types of seagrasses. Both Manatee Grass (*Cymodocea filiformis*) and

Turtle Grass (*Thalassia testudinum*) are found in shallow waters ranging from 0.6 -1.5 m (2 - 5 ft). Lastly, Widgeon Grass (*Ruppia maritima)* is mostly found is freshwater however, it has a high tolerance for salinities and can therefore be found in some Texas estuaries. Seagrasses' ability to resist and recover from dredging disturbances varies by species and is related to life history characteristics (Fraser et al. 2017).

A study by Kilminster et al. (2015) summarized seagrass vulnerability to disturbance by grouping species of seagrass into three categories based on their life history characteristics. Persistent species are those with long turnover times and are more resistant to disturbance. A typical persistent species is Turtle Grass. Colonizing species have limited resistance to disturbance but they recover quickly. Typical colonizing seagrasses include Shoal Grass, Widgeon Grass, and some species of Star Grass. Opportunistic species share traits with both classifications. Typical opportunistic seagrasses include Manatee Grass.

Onuf (1994) conducted an analysis of seagrass cover in response to dredging activities between 1965 - 1988 in the Laguna Madre. Light reduction caused by turbidity plumes from maintenance dredging was the primary cause of seagrass losses. It was found that adverse effects were strongest close to disposal areas; however, they were still detectable over 1 km (0.62 mi) from the nearest dredge disposal area. Effects were seen up to 10 months after dredging in the seagrass meadow and transition zone. It was suspected that dispersion events caused by wind-generated waves were responsible for the prolonged effects.

Another study by Sheridan (2004) analyzed long term alterations in seagrass habitats at dredged material placement sites in Laguna Madre, Texas. Maintenance dredging occurred between 1994 - 1995 and seagrasses were buried at disposal sites. Seagrass characteristics were measured and compared to nearby and distant seagrasses over three years. Signs of steady seagrass re-colonization occurred in the latter half of the study period. Mean seagrass coverage of deposits had reached 48% approximately 3 years after dredging occurred. *Halodule wrightii* significantly recovered within 3 years.

Seagrasses most at risk in Lavaca and Matagorda Bay are *Halodule wrightii* and *Thalassia testudinum.* Seagrass beds exist predominately along the western shores of Matagorda Bay, near the proposed dredging site (Figure 9). The dataset from TPWD is comprised of a merged compilation of seagrass datasets from various surveys conducted by TPWD, U.S Geological Survey, and the University of Texas Bureau of Economic Geology. Aerial photo interpretation and digitization or ground transects was used to provide data. The TPWD dataset represents seagrass distributions from 2001 or 2007. TPWD will be releasing an updated Matagorda Bay seagrass dataset in 2021. The dataset from NOAA represents a comprehensive shapefile that includes all submerged aquatic vegetation (SAV). NOAA Coastal Services Center worked cooperatively with TPWD and Texas A&M University Center for Coastal Studies to develop SAV data for the coastal bays.

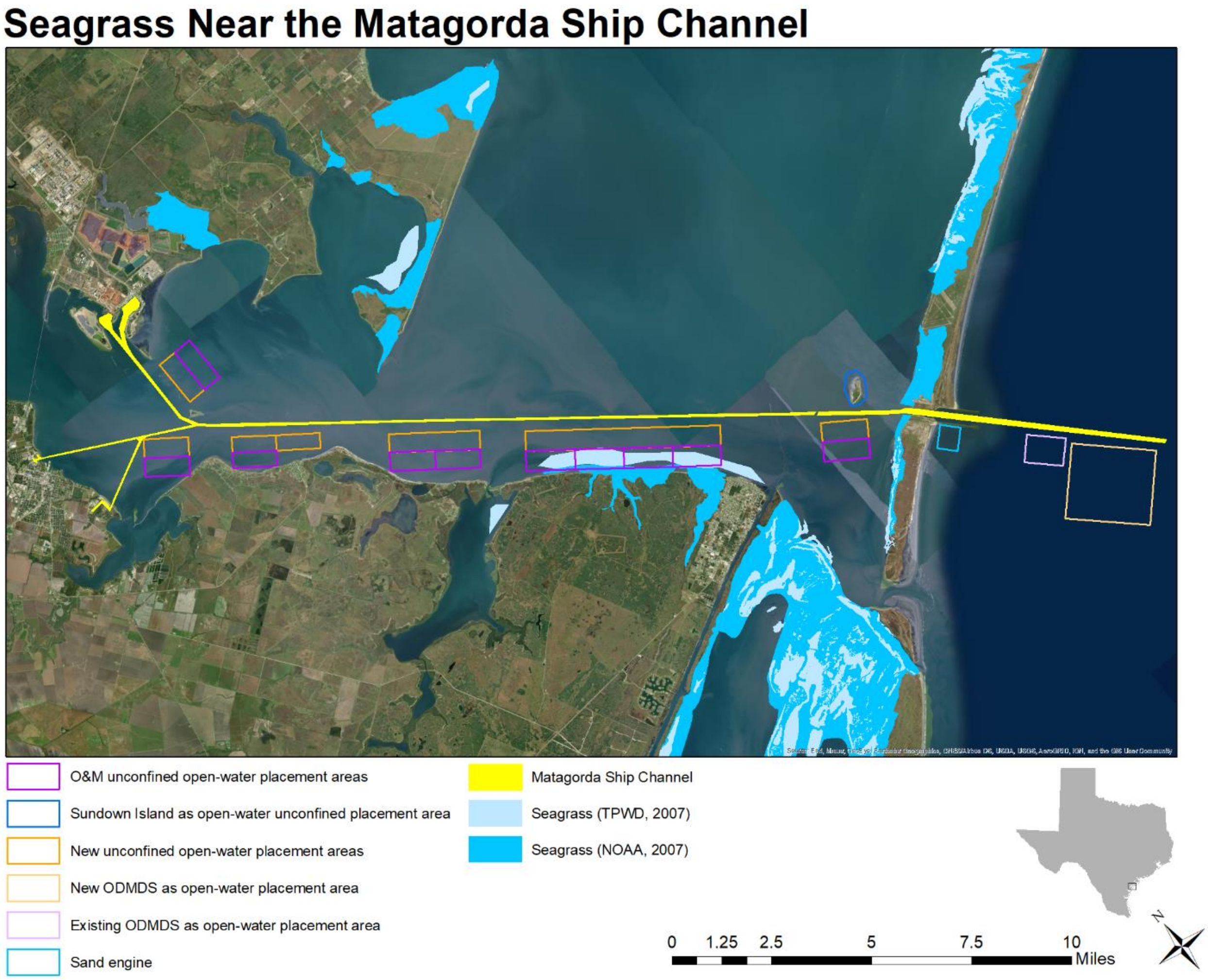

Figure 9. Seagrass habitat areas (TPWD 2007, NOAA 2007) and the project location areas (Maglio 2021).

The intersection between the project area and seagrass habitat is shown in orange hatching in Figure 10. The new unconfined open-water placement area NP 2 is located along the seagrass habitat. A total of 1017.39 acres of seagrass is predicted to be affected by dredging operations spoil placement. It is likely that additional seagrass loss will occur due to light reduction caused by turbidity plumes. Based on existing literature, the seagrass beds in Matagorda Bay may take up to 3 years to fully recover. Seagrass habitat is not listed in the mitigation plan (Maglio 2021,USACE 2019, Table 9).

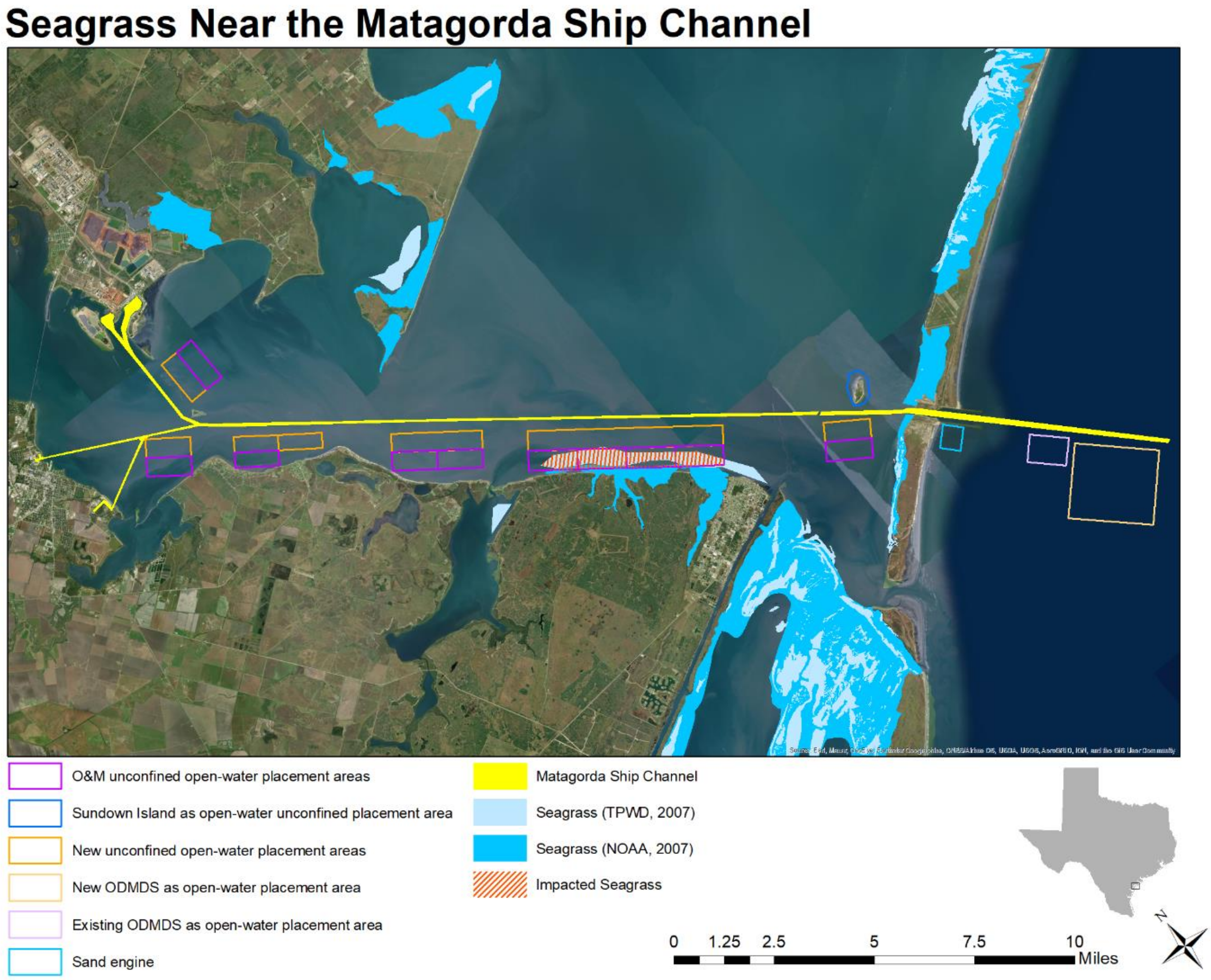

Figure 10. Area of seagrass habitat that could be effected by the prject.

Bay Bottoms

Benthos are essential to the functioning of an ecosystem. They play an important role in the food web and serve as an indicator of environmental conditions. Benthic-pelagic coupling is a concept that involves the exchange of energy, nutrients, and organisms from the benthos to pelagic or demersal nekton (Nunnally, 2019). Nutrients such as nitrogen silicate and phosphorus are recycled by processes in the benthos. Additionally, the decomposers and detritus feeders that live on the bay bottom break down dead plants, animals, and waste products. Benthic organisms are separated into two groups based on their habitat: infauna and epifauna. Infauna burrow themselves into the sediment and epifauna live either directly on the surface of the bay bottom or attached to hard surfaces. Benthic organisms are often used as bioindicators of disturbance because they are relatively long lived, typically immobile and susceptible to changes in their environment (Montagna et al. 2013). Dredging results in the removal of benthic habitat as well as the epifaunal and infaunal organisms. The organisms can be impacted directly during the physical removal of the substrate or indirectly by smothering during spoil placement. The type of sediment and the thickness of the deposited sediment layer are highly relevant for the survival of the benthic community (Powilleit et al. 2009). The limit for the different species to survive burial by burrowing to the sediment surface is called "fatal depth" (Essink, 1999). However, benthic organisms can still be affected even before fatal depth is reached. The potential impacts of dredging on the benthic community depend on biological process such as feeding mechanisms, life history characteristics, mobility, stage of development, and environmental conditions (Fraser et al. 2017). The effect of sedimentation on sessile organisms depends on the duration of exposure, proximity to the dredging operations, and morphology. Mobile benthic invertebrates are generally less vulnerable to sedimentation because they can move to less disturbed areas. Because certain critical life stages are still susceptible to many indirect impacts of sedimentation, mobility alone does not suggest that these organisms are resistant to dredging (Fraser et al. 2017). The effects of dredging and sediment disposal on benthic fauna has been investigated.

Rice et al. (1981) conducted a study to evaluate the effects of dredged sediments on benthic communities from Bayboro Harbor in Florida. It was concluded that reduced densities were detected, and the benthic environment suffered from severe degradative effects at the dredged disposal site. However, it was acknowledged that some buried habitats may be uncovered quickly while others may remain affected for years.

In a study by Saloman et al. (1982), benthic data was analyzed from Panama City Beach, Florida to study the effects of offshore dredging. Results showed that dredging caused an immediate decline in the benthic community followed by a rapid recovery in both number of individuals and number of species after 1 year. The same area was analyzed 3 to 4 years after dredging occurred and there were no long-term adverse effects detected (Cutler and Mahadevan 1982).

Following a dredging project near Fort Pierce Inlet, Florida, changes in benthic fauna were monitored (Johnson and Nelson 1985). Researchers observed a 72% reduction in the abundance of benthic communities. Conversely, a study by Blake et al. (1996) sampled benthic faunal community conditions at four locations off the coast of Florida pre-and post-dredging activities, and no change was found in density or infaunal species richness after dredging operations.

A study conducted outside of the United States yielded similar results. Powilleit et al. (2009) measured responses of six different macrobenthic invertebrates from a brackish water habitat in Mecklenburg Bay (western Baltic Sea) to test the organisms' escape reaction to dredged material. Three bivalves (*Arctica islandica, Macoma, Balthica, and Mya arenaria*) and a polychaete (*<u>Nephtys</u> hombergii)* showed high escaping potentials as they successfully escaped 32 - 41 cm (1 – 1.3 ft) of deposited sediment. One polychaete (*Bylgides sarsi*) escaped from 16 cm (0.52 ft) of sediment and the other (*Lagis koreni*) did not migrate.

Based on the literature, dredging operations may have immediate adverse effects on Lavaca and Matagorda Bay's open bay bottom habitats. It is likely that the maintenance dredging will result in sedimentation/burying of the benthos. Based on existing literature, recovery of the benthic communities is expected to occur in the range of 1 - 3 years.

Turbidity Effects

Sediment resuspension can lead to turbidity, which is often a temporary effect of dredging. Turbidity is a visual measurement of the clarity of a liquid and serves a major indicator of water quality. Turbidity is composed of the particles suspended or dissolved within water producing a cloudiness or muddiness (International Association of Dredging Companies 2021). Naturally occurring turbidity takes place in all water bodies during storms, river inflow, tides, algal blooms, and seasonal wind events. As turbidity levels increase there is a decrease of optical quality within the water, and a decrease in the distance that light can reach within the water. Turbidity is not classified as a pollutant but indicates how much inorganic and organic matter or sediment is in the water. Total suspended solids (TSS) are a measurement of particles within the water, the greater level of TSS, the cloudier the water leading to higher levels of turbidity. Another common unit to measure turbidity is Nephelometric Turbidity Units (NTU), and NTU measures the average volume of particles by incident light scattered at right angles from the sample.

Photosynthesis / Phytoplankton / Primary Production

Because increased turbidity reduces light, there is concern about the effects on photosynthesis, which requires light. Turbidity is a natural occurrence in water, and many marine species tolerate and survive in low light conditions depending on exposure time. Natural occurring turbidity can reduce sunlight for a short period of time, but an increase in turbidity can result in the disruption of flora and fauna within the environment. Photosynthesis can be greatly disturbed when sunlight is blocked from penetrating water by turbidity. Particulate matter or suspended solids modify light penetration, reducing sunlight, leading to the reduction in productivity of benthic alga and primary producers within an area (Birklund 2005). Light regime effects on primary production by phytoplankton and macrophytes are dependent on variables such as temporal and spatial scales of the exposure for available light, nutrient concentrations, and the requirement of light for specific species (Birklund 2005).

Bioindicators commonly used to analyze water quality and ecological conditions from a change in turbidity are phytoplankton. Phytoplankton are sensitive, respond quickly to water quality changes, and remain in one general area. A case study in the Tagus estuary (Portugal) where sediments were dredged for 5 months to extend a navigable waterway measured phytoplankton biomass using chlorophyll *a* concentration and a species richness index (Cabrita 2014). Analyses compared data before dredging and during dredging. The biomass of phytoplankton (as indicated by chlorophyll *a*) during dredging was similar to the seasonal patterns found within the Tagus estuary before dredging, leading to a conclusion that dredging had minimal impact on the variability of phytoplankton biomass. Thus, it was determined that short term dredging did not result in light limitations or reduction of phytoplankton biomass (Cabrita 2014).

Seagrass can also be affected by dredging because they require light to reach the bottom. An environmental monitoring program occurred on the Danish-Swedish border in Öresund to determine the effects of a dredging project of approximately 7 million m^3 of material from the seabed (Valeur 2001). Eel grass, *Zostera marina*, covered a large portion of this marine ecosystem. During this dredging event, eelgrass was reduced near the dredging site due to the reduction of transparency in the water, but the long-term evaluations determined that there were no permanent regional changes in turbidity and sedimentation from this construction. The population of eelgrass was last surveyed with expectation of full recovery. However, if a permanent increase of turbidity occurs in Matagorda Bay, then a loss of seagrass habitat is likely.

Filter Feeding

Particulate matter resuspended in the water column in high concentrations can affect filter feeders. Turbidity may impact filter feeders by clogging the feeding appendages with inorganic material with low nutritional value, or diluting suspended organic food (Birklund 2005, Zarillo et al. 2009). The suspension of inorganic particles can affect the quality of filtered material, leaving filter feeders exerting more energy in the process of food selection. Increased turbidity and increased sediment concentration may lead to harmful impacts on benthic organisms because these organisms are at risk for slowed growth and tissue abrasion. Indirect effects can also occur. The release of nutrients to the water column from harmed organism tissue includes organic matter such as carbohydrates, fats, and lipids (Zarilla et al. 2009). The release of these nutrients can stimulate bacterial metabolism and blooms that may in turn lead to hypoxia or anoxia formation as a result from increased oxygen consumption locally.

Filter-feeding bivalves are common in the proposed dredged site. Bivalves are adapted to tolerate natural concentrations of suspended matter in the water column and have a protection mechanism for sediment overload with the ability of closing their valves (Widdows et al. 1979). The feeding response of unionid mussels were compared in clear water and turbid water, with the expectation that turbid water would impair mussel feeding (Tuttle-Raycraft and Ackerman 2019). The particle clearance rates peaked at the lowest TSS concentration with the highest velocity of water movement. In contrast, the mussels experienced a low clearance rate at higher TSS concentrations with low velocity rates. An increase in velocity of water movement led to a higher clearance rate and the mussel’s ability to feed despite the level of TSS within the water. It was determined that the feeding rate of these mussels was impacted by the concentrations of organic and inorganic material as well. But the clearance rate of these bivalves was greatly influenced by the TSS and the velocity of water. With an increase in water velocity and high levels of TSS, the mussels experienced a 40 - 60% increase in clearance rate (Tuttle-Raycraft and Ackerman 2019). If turbidity is increased in the proposed dredging site, then it is likely there will be effects on filter feeding mollusks.

Fish effects

Feeding and migration of fish may be affected by turbidity levels. Some predators and prey rely on the detection of each other by sight, which can be affected by turbid water. The risks that are a result of this impairment include changes in trophic interactions within the food web and poor nursery habitat conditions (Pekcan-Hekin 2007). Predation can be greatly affected by low visibility of water and lack of light levels. Turbidity reduced the efficiency of hunting prey for planktivorous fish, and loss of benthic prey due to the clogging of feeding structures (Hecht and Van der Lingen 1992, Benfield and Minello 1996).

Fish communities often reside at the edge of a plume, between the turbid and clear water. Prey can hide from predators such as birds or larger fish. An analysis was performed on the behavior of herring and cod in waters with differing thresholds of sediment plumes. It was discovered that both species were similar in sensitivity and the avoidance threshold was around 3 mg/L in clay plumes (Westerberg et al. 1996, Johnston and Wildish 1981). This threshold was determined to be the avoidance limit for both herring and cod for TSS. The cod were placed in nighttime conditions with low light visibility and the avoidance threshold levels were consistent with daytime predation rates (Westerberg et al., 1996). Although this laboratory study identified the avoidance level, in situ field conditions can be different and predicting effects can be difficult. When species of fish experience a sediment gradient along an area, tolerance and avoidance levels in this area may change.

Many commercial fish have pelagic eggs. Species of fish found in the proposed dredged site, such as black drum (*Pogonias cromis*), red drum (*Sciaenops ocellatus*), and spotted seatrout (*Cynoscion nebulosus*) have pelagic eggs. Water columns with high concentrations of total suspended solids are expected to alter the sinking rates of eggs dispersed throughout the water column. An analysis of cod revealed a positive linear relationship between the concentration of suspended silt sediments and the sinking rate of cod eggs (Birklund 2005, Westerberg et al. 1996). The eggs stick to the suspended clay or mud sediment leading to a faster sinking rate. An increase in sinking rate of pelagic eggs can increase egg mortality if the egg reaches the bottom before development is complete.

Mercury Mobilization

The Aluminum Company of America (ALCOA) released wastewater, containing mercury (Hg), into Lavaca Bay from 1966 to 1979 (EPA 2001). Human exposure to different forms of Hg can affect areas of the brain, kidneys, gut lining, and can be distributed to other organs throughout the entire body (Bernhoft 2012). In the estuarine and marine environment, Hg can be biomagnified in food webs making fish dangerous to eat, and human exposure is often due to consumption of contaminated fish. Mercury is volatile, so other exposures come from out-gassing, and Hg can be absorbed into the human body through two routes: inhalation and ingestion. In Lavaca Bay, most of the Hg in sediments is in the area surrounding the dredge spoil island (Bissett 2008). Past dredging events are known to have mobilized the Hg in the sediments (Bloom and Lasora 1999). This Hg contaminated area is within the proposed dredging project area, thus there are four areas of concern: 1) short-term effects of the mobilization of Hg during dredging, 2) long-term effects of the mobilization of the Hg, 3) bioaccumulation and biomagnification of Hg, and 4) a potential change in Hg dynamics due to nutrient loading.

The historic ALCOA Hg discharge site was in the area where road FM 1593 ends (Bloom 2004). Hg concentration assessments in bay sediments have not been conducted recently, and the last sampling collection was conducted in 2002. A new Hg concentration assessment should be conducted to accurately assess the current location and concentration of the Hg in Lavaca Bay, Texas. A review, summary, and model of the distribution of Hg in sediments demonstrates the highest concentrations around the dredge spoil island, within the area that is now closed (Bissett 2008, Figure 11a). Bisset (2008) also predicted Hg concentrations in oyster tissues, and they are high, i.e., > 1 µg/g, throughout Lavaca Bay (Figure 11b). The concentration of Hg in sediments is highest at a sediment depth between 10 – 30 cm (.003 -01 ft) (Bloom et al. 1999).

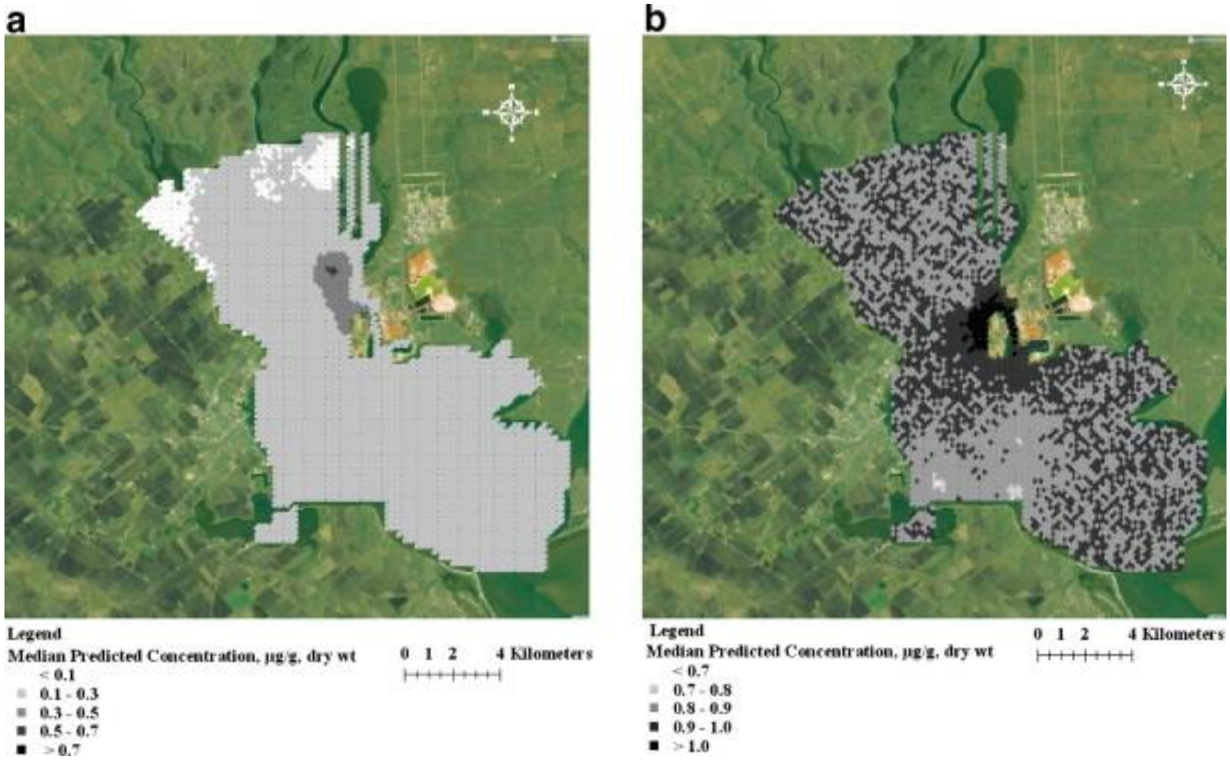

Figure 11. Predicted mercury concentrations in Lavaca Bay (a) sediments and (b)oysters (Bissett et al. 2008).

The map of Hg distributions from Bisset (2008, Figure 11) was overlayed with the project design provided in the USACE public presentation (Maglio 2021, Figure 12). The main area of concern can be seen at the terminus of the MSC project where the turning basin will be located, and the highest concentrations of Hg in the sediment are found (Figure 13a).

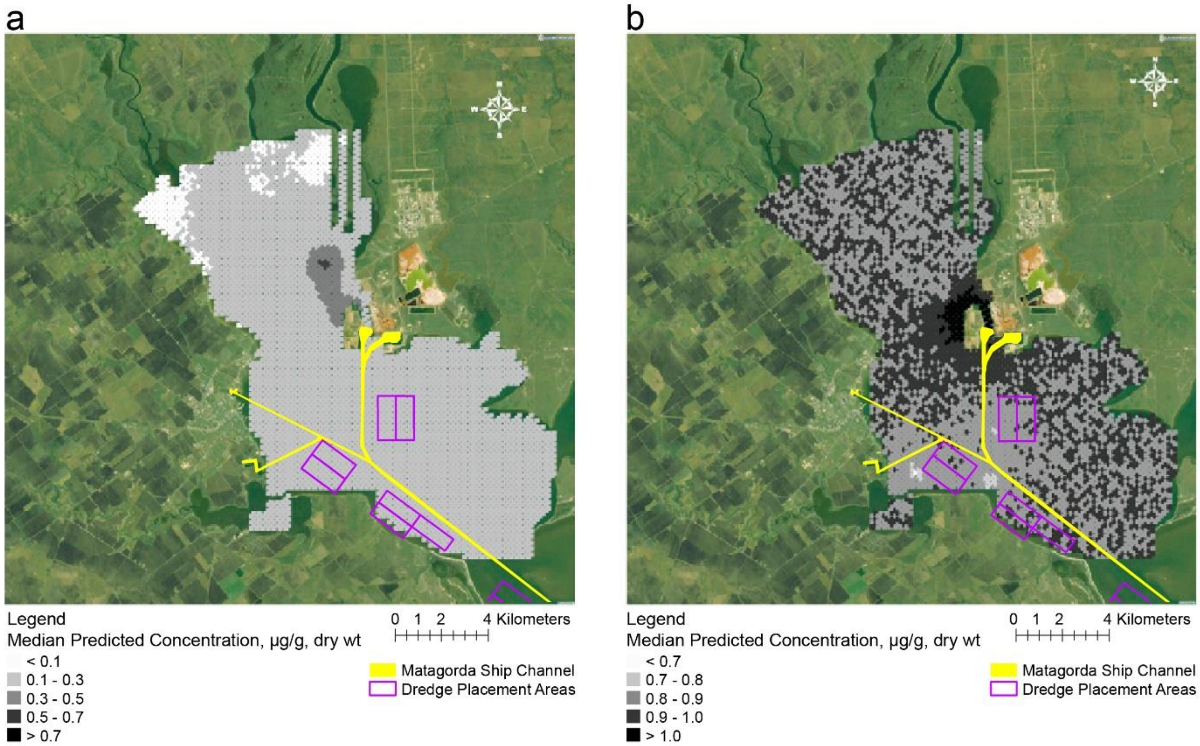

Figure 12. Mercury distribution in sediments (a) and oyster tissue (b) provided by Bisset et al. (2008) and project design provided by Maglio (2021). The yellow lines indicate the position of the Matagorda Ship Channel, and the purple outline indicate proposed location for dredge spoil placement.

The dependence of Hg toxicity is driven by dynamics of Hg and methylation because methyl-Hg is the organic, toxic form. Methylation of Hg occurs in reducing sediments rich in sulfate (King et al. 2001). These kinds of sediments are typical of estuarine muds. Once mercury is introduced into the environment it can attach to sediments or TSS in the water column, settle to the bottom, and begin to methylate. Nutrients found in the water column, mainly Nitrogen, will also methylate in the sediment. The chemical competition between Hg and other elements can result in Hg remaining in the water column for longer periods of time (Driscoll 2012). The dynamics of the short-term effects of the mobilization of mercury are mainly affected by the methylation of Hg. High concentrations of Hg can be observed in correspondence to fine sediment grain size, such as clay and silt (Covelli 2021). Methylation of Hg is both a biotic and abiotic process (Celo et al. 2006).

Given the importance of fine sediments in Hg methylation, the distribution of fine sediments (White et al. 1989) was overlayed on Figure 12 to display the location of muds (i.e., the clay + silt fractions) in Lavaca Bay, Texas (Figure 13). The location of the muddy sediments corresponds to the location where the highest Hg concentrations were recorded. Lavaca Bay, Texas is known to have “high sedimentation rates and low remixing rates” (Bloom et al. 2004). When Lavaca Bay sediment in sea water is shaken in the laboratory, between 1% and 5% of the Hg attached to the sediment is desorbed and released into the water column (Bloom 1999). Considering that in Lavaca Bay there are high sedimentation rates, low remixing, and high methylation after dredging, it is likely that mobilization of the Hg will be high (Bloom et al. 2004).

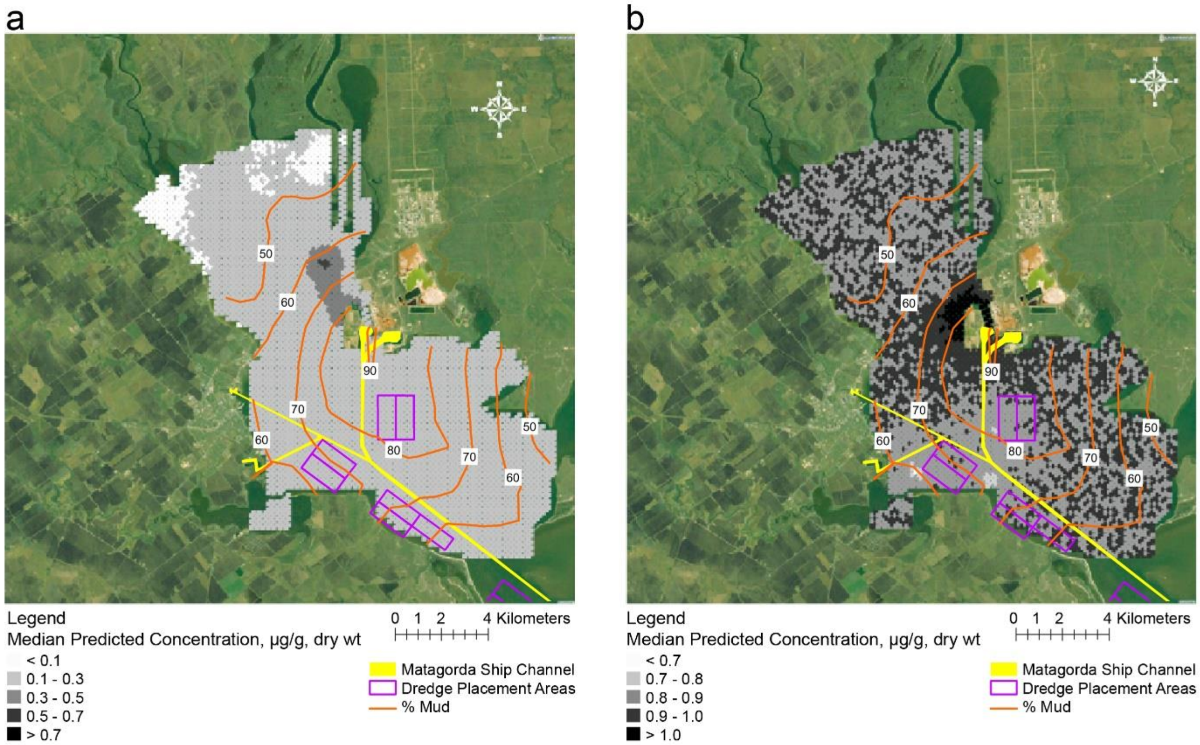

Figure 13. Mercury concentrations in Lavaca Bay (a) sediments, and oyster tissue (b) (Bissett et al. 2008), project location and footprint (Maglio 2021), and sediment grain size as percent mud in orange isopleths (White et al. 1989).

In contrast, in the sediments of the Gulf of Trieste (northern Adriatic Sea, Italy), only 1.3% of Hg compounds are soluble, so that a resuspension event there caused by dredging over a limited period of time might have a negligible impact to the water column (Covelli 2021).

The dynamics of the long-term effects of the mobilization of mercury is affected by continuous resuspension events which could occur due to depression waves (i.e., the wake) created by ships as they move through a channel (Teatini 2017). The increased surface area of the MSC, after dredging is completed, allows for a greater disturbance following each depression wave. These resuspension events could be responsible for maintaining a slightly higher level of Hg concentration in the water column for a longer period of time after the concentration spike from the dredging event itself.

Lavaca Bay is a Superfund site with an area closed to fishing (Figure 14). The elevated Hg concentrations in the water column and sediments can cause Hg and other heavy metals to be ingested or absorbed via bioaccumulation by smaller organisms and biomagnified up the trophic levels when they are fed upon by larger fish.

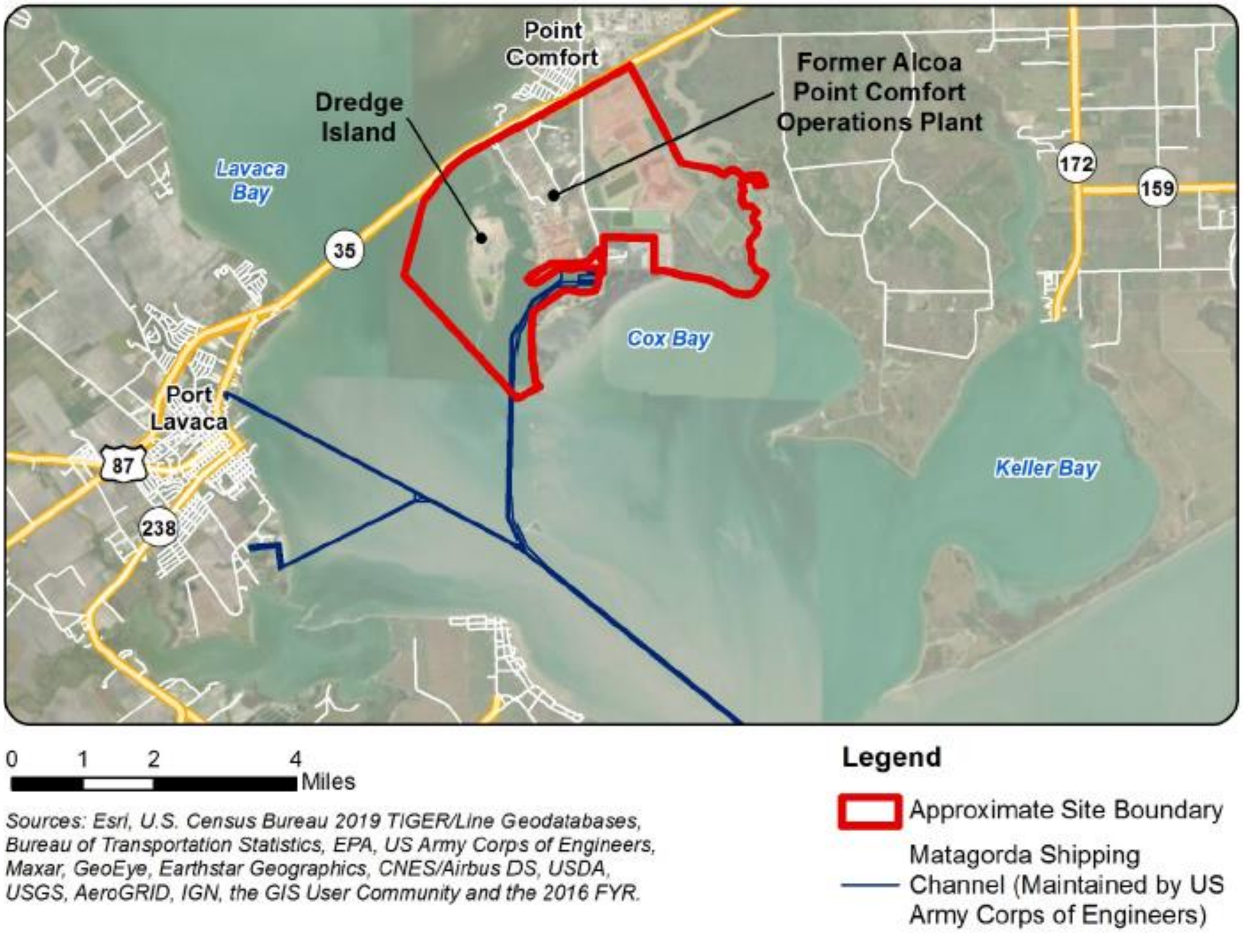

Figure 14. Alcoa, Point Comfort, Lavaca Bay Superfund Site (EPA 2021). Blue lines are ship channels. Red lines is boundary of former Alcoa plant and area of water closed to fishing.

Biomagnification of Hg is concentrated up the trophic levels with the mass consumed by each level (Capuzzo 1987). One particular fish of human interest is red drum (*Sciaenops ocellatus*), a popular sport fish, that have been found to contain around twice the level of Hg that is safe for human consumption (EPA 2019). Mercury concentrations in red drum tissues have been slowly declining since 1997 (EPA 2012, Figure 15). The most recent sampling in 2020 has measured the lowest concentrations of Hg in fish tissue to date. If dredging of the MSC mobilizes a significant amount of Hg, then this downward trend could be halted or reversed resulting in red drum containing an unsafe level of Hg for an extended period of time.

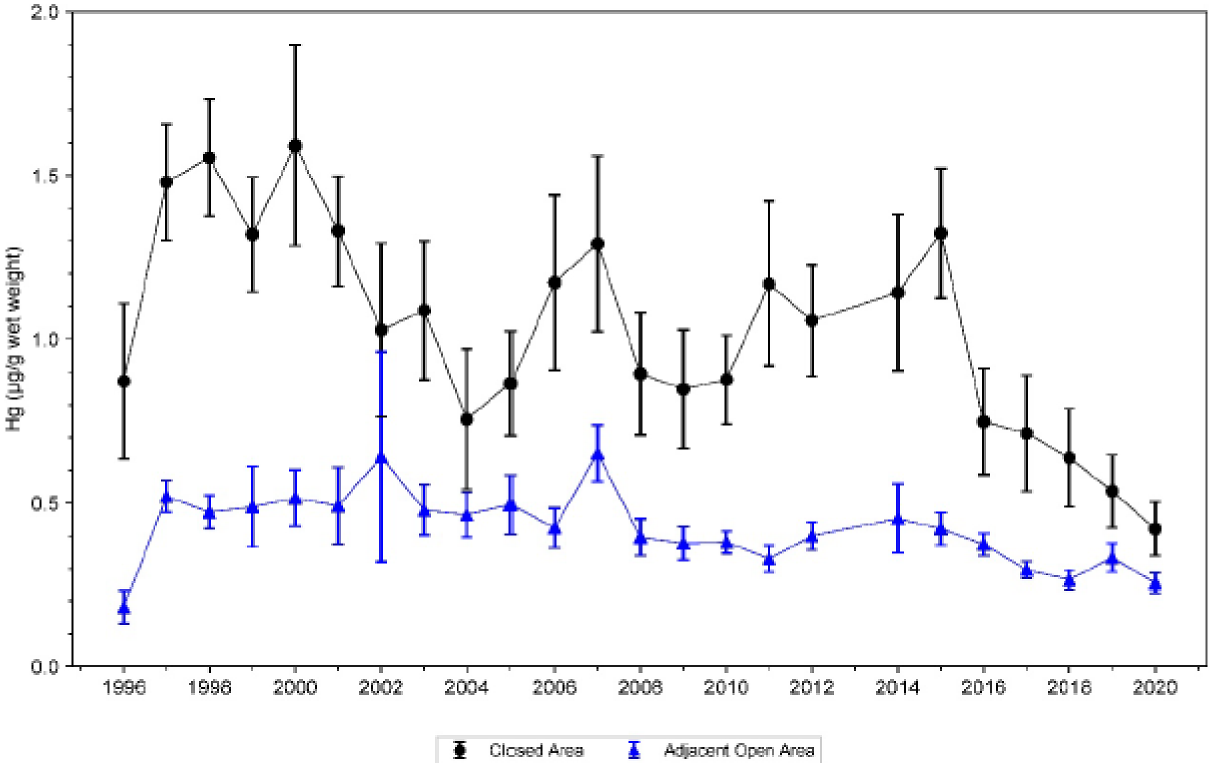

Figure 15. Average mercury concentration in red drum tissue found in Lavaca Bay (EPA 2021). Red drum collected from the open area of Lavaca Bay (blue) and closed area (green).

Plastic and Microplastic

Plastic and microplastic contamination may also play a significant role in the Lavaca-Matagorda Bay system in relation to contaminant transport (i.e., mercury and PAHs) and food webs and it is possible that their presence could alter food web dynamics due to dredging. Plastics can negatively impact organisms of different trophic levels, from plankton through bivalves, fishes, birds, and up to large mammals and turtles, through contact or ingestion (Santos-Echeandía et al. 2020 and references therein). However, the effects of microplastics on organisms vary widely because there are many variables to consider and there is still much to learn about transport, uptake, and stress mechanisms. Several studies have demonstrated that pollutants, such as heavy metals (Fernández et al. 2020, Holmes et al. 2014, Nakashima et al. 2012, Rochman et al. 2014) and hydrophobic organic matter (Lee et al. 2014), adsorb on plastic surfaces in marine ecosystems and could act as vectors for the entry of metals and organic matter into marine food webs. Metals have been shown to adsorb onto microplastics at concentrations that are several orders of magnitude higher than in the surrounding water, thus increasing the potential exposure to aquatic organisms (Holmes et al. 2014, Nakashima et al. 2012, Naqash et al. 2020). Additionally, microplastics have been found to adhere to the surfaces of macroalgae (i.e., seagrasses) where the contaminated microplastics may also enter the marine food chain through the consumption of such primary producers (Gutow et al. 2015, Priscilla et al. 2019).

Some plastic polymers, such as polyethylene (PE), polypropylene (PP), polyurethane (PUR), and foamed polystyrene (foamed-PS), are less dense that seawater and float (Santos-Echeandía et al. 2020). These lighter polymers often wash ashore and are found as plastic litter on beaches and coastlines. However, more dense polymers, such as polyvinyl chloride (PVC), polystyrene (PS), and thermoplastic polyester (PET), have greater densities than seawater and will sink in the aquatic environment. These heavier polymers may accumulate or be buried in the sediments. Microplastics may be divided into primary and secondary types (Naqash et al. 2020). Primary microplastics are manufactured deliberately such as pellets and mircobeads found in cleaning agents, paints, and cosmetic products. Secondary microplastics are created by size reduction of plastic debris by physical and chemical processes, such as weathering, exposure to ultraviolet radiation, and mechanical forces. Commercial plastics commonly contain various chemical additives, which include plasticizers, UV inhibitors and flame retardants that protect the plastic from breakdown caused by irradiation and weathering, while also increasing consumer uses (Lee et al. 2020). While the additives help improve the product quality, they can also leach from the plastic when it is in the environment.

In areas with metal or organic matter contamination of the sediments, such as Lavaca Bay (mercury and PAHs), higher contaminant concentrations may sorb onto the plastics, particularly polystyrene, due to prolonged contact and greater weathering of the plastic. While in the sediments, the plastics may pose a risk to benthic sediment feeders or detritus feeders (Karlsson et al. 2017, Panti et al. 2015, Priscilla et al. 2019). If a resuspension event were to occur, such as a tropical storm or dredging project, plastics in the sediment may be resuspended and transported elsewhere. Thus, resuspension events potentially reintroduce contaminated plastics into the water column where they may once again come into contact or be ingested by estuarine organisms or disperse/transport the plastic and associated contaminants to somewhere else in the environment. Given the known mercury and PAH contamination in Lavaca Bay and their genotoxic (Santos-Echeandía et al. 2020), carcinogenic, mutagenic, and toxic effects (Guven et al. 2018), this raises concerns about microplastics in the sediments and the potential for them to act as contaminant transport vectors.

The role of plastics and microplastics in Lavaca-Matagorda Bay as a vector for contaminant transport is not well constrained. In addition to the plastic litter typical of most bay ecosystems, Lavaca-Matagorda Bay also has abnormally high amounts of pre-production plastics and powder/flakes in its waters and along the shoreline due to releases by Formosa Plastics. Furthermore, while it is assumed that sediments in the vicinity of Formosa Plastics will have more microplastic contamination as the Nurdle Patrol finds more plastic nurdles washed ashore in Cox Creek and near the plant (Tunnell 2021), the actual concentrations and distributions of plastics in the sediments of Lavaca-Matagorda Bay are currently not well understood. As part of a recent legal settlement, Formosa Plastics Corporation must pay outside contractors to remediate the pellets and powder accumulated in Cox Creek and Lavaca-Matagorda Bay.

On-going MBMT plastics research

The Matagorda Bay Mitigation Trust has funded research to resolve the prevalence of plastics and measure mercury concentrations, to map the concentration and distribution of microplastic particles in water and sediment, and to assess the toxicity and bioaccumulation of persistent pollutants that adsorb to microplastic particles.

- Jeremy Conkle, Texas A&M University-Corpus Christi – Mercury and Plastic in Commercial and Recreational Fisheries in Lavaca, Matagorda, and San Antonio Bays: Risk Assessment and Interaction between the Two Contaminants
    - This study will investigate the prevalence of plastic, measure Hg concentrations, and calculate the selenium:mercury molar ratios in commercial and recreational fisheries (e.g., red drum, black drum, spotted trout, shrimp, blue crab, and oysters). Experiments will investigate the extent Hg can bind to plastic and its potential role as a source of mercury to biota.
- Zhanfei Liu, University of Texas – Microplastic concentration in sediments and waters of Matagorda and San Antonio Bays: Initial assessment and mitigation plans
    - This study will map the concentration of microplastic particles in the water and the surface sediments (top 10 cm) in order to identify areas with high microplastic concentrations, identify the most likely sources for microplastics (recent and past), and understand the microplastics' transport pathways in the estuaries as floating in the water column, primary sources (entering the bay), and secondary sources (within the bay) such as erosion/resuspension and re-sedimentation. Maps of microplastics distribution and any "hot spots" are expected by 2024.
- David Hala, Texas A&M University-Galveston – The fate and toxicity of microplastics and persistent pollutants in the shellfish and fish of Matagorda Bay
    - This study will investigate: 1) microplastics in surface water and ingested by shellfish and fish in partnership with an existing program studying food webs in the bay, 2) persistent pollutant (PAHs, PCBs) bioaccumulation along with pollutant levels in surface waters and adsorbed to microplastic particles, and 3) the health effects of microplastics and pollutants using toxicity tests with embryo-larval fish.

Conclusions

Circulation and Salinity

A hydrodynamic and salinity modeling study for Matagorda Ship Channel (MSC) channel improvement project was performed by the U. S. Army Engineer Research and Development Centers' (ERDC) Coastal and Hydraulics Laboratory (CHL) Coastal Engineering Branch (HN-C), Navigation Division. The modeling study (Lin et al. 2018) investigated the impact on water levels, currents, and salinity in and around Matagorda Bay from the proposed ship channel widening/deepening project.

Lin et al. (2018) have concluded that the general effect of the proposed channel improvement projects on water surface elevations, current velocities, and salinity changes is relatively small to cause any navigation and environmental issues. However, the small changes described by Lin et al. (2018) have the potential to alter the morphology, water quality, and secondary production in Matagorda and Lavaca bays. For instance, a 5 cm increase in the water level could shift intertidal marsh and tidal flats to open water environments, an increase in the current speed of 0.1 m/sec is enough to keep fine-grained sediment suspended and potentially increase turbidity, and a 2 ppt increase in long-term average salinity can alter water quality and decrease secondary production of suspension feeders.

Furthermore, our assessment has identified a few issues regarding the modeling procedure of Lin et al. (2018) that could be improved. Improved model performance and predictions could be obtained if the model was calibrated and/or validated using recent datasets and by incorporating multiple validation simulations capturing multiple wind regimes. Similarly, incorporating a future scenario of potential landscape changes due to SLR in the modeling could have given different results.

Finally, additional modeling studies are needed to investigate the impacts of channel improvement projects on storm surge water levels, sediment transport, and the potential increase of channel shoaling rate. Table 8 summarizes the issues in each component of the Lin et al. (2018) modeling study and our recommendations to improve the predictions of water levels, currents, and salinity. It also summarizes additional modeling studies necessary for a holistic assessment of the impacts of the channel improvement projects in and around Matagorda Bay.

Groundwater

Puncturing upper confining beds creates areas of groundwater-surface water interaction that may result in salinization of the surficial aquifer or disruption of groundwater pressure and flow fields and discharge further offshore. The existing ship channel already intercepts the shallow aquifer. However, deepening and widening the ship channel will extend the area of exposed aquifer thus increasing the potential influence of groundwater-surface water interactions in the shallow subsurface and the bay. How the groundwater-surface water interaction occurs, will be site specific and depend on local conditions, such as hydraulic gradients, sediment compositions, clay layers and lenses, and hydroclimatic conditions. Due to the site-specific nature of this issue, more detailed information and monitoring is needed for Lavaca-Matagorda Bay to determine the impacts of deepening and widening the ship channel on groundwater.

- Assessment of shallow subsurface for localized confining layers that may be compromised
- Long-term monitoring of shallow, < 30.5 m (100 ft) groundwater quality (e.g., salinity and contaminants)
- Evaluation of groundwater discharge in and around the existing ship channel along with an evaluation of groundwater hydraulic heads in the shallow aquifer to establish rates of groundwater flow and ascertain risks of saltwater intrusion.
- Assessment of mercury solubility and mobility from site specific dredge spoils and dredge spoil placement area sediments.

Benthos

Habitats of concern include oyster reefs, seagrass beds, and the open bay bottom. At risk areas are comprised of valued ecosystem components (VECs) that may be impacted indirectly by means of sedimentation or directly by physical removal. Spoil placement and construction areas overlap with known habitats such that 834 acres of oyster reef and 1,1017 acres of seagrass may potentially be affected by operations.

Based on the literature, dredging operations may have immediate adverse effects on Lavaca and Matagorda Bay’s open bay bottom habitats. It is likely that the maintenance dredging will result in sedimentation/burying of the benthos. Based on existing literature, recovery of the benthic communities is expected to occur in the range of 1 – 3 years.

Ways to mitigate the effects of dredge operations include careful pre-and post- construction environmental studies, a means to control turbidity, dredging during periods of low benthic populations, and thoughtful disposal of spoil (Johnston, 1981). However, resuspension by ships is likely to enhance suspended sediment concentrations and siltation (Aarninkhof, 2008).

Turbidity

As turbidity levels increase there is a decrease of optical quality within the water, and a decrease in the distance that light can reach within the water. Natural occurring turbidity can reduce sunlight for a short period of time, but an increase in turbidity can result in the disruption flora and fauna within the environment. Analyses determined that short term dredging does not result in light limitations or reduction, of a common bioindicator, phytoplankton.

Turbidity may impact filter feeders by clogging the feeding appendages with inorganic material with low nutritional value or diluting suspended organic food. Increased turbidity and increased sediment concentration may lead to harmful impacts to benthic organisms because these organisms are at risk for slowed growth and tissue abrasion. Indirect effects can also occur. The release of nutrients to the water column from harmed organism tissue includes organic matter such as carbohydrates, fats, and lipids (Zarilla et al. 2009). The release of these nutrients results can stimulate bacterial metabolism and lead to hypoxia or anoxia as a result from increased oxygen consumption locally.

Bivalves in the area can be greatly influenced by the TSS and the velocity of water. With an increase in water velocity and high levels of TSS, mussels experienced a 40 - 60% increase in clearance rate (Tuttle-Raycraft and Ackerman 2019).

Turbidity will reduce the efficiency of hunting prey for planktivorous fish, and loss of benthic prey due to the clogging of feeding structures. The sediment of clay or mud will increase sinking rate of pelagic eggs, increasing the mortality of eggs that reach the bottom before development is complete.

Mercury

The assessment has identified four areas of concern, short-term effects of the mobilization of Hg, long-term effects of the mobilization of Hg, bioaccumulation/biomagnification, and a change in Hg dynamics due to nutrient loading.

Hg concentration assessments in bays sediments have not been conducted recently, and the last sampling collection was conducted in 2002. A new Hg concentration assessment should be conducted to accurately assess the current location and concentration of the Hg in Lavaca Bay, Texas.

The conclusions by Bloom et al. (2004) are still valid: “In Lavaca Bay, because of high sedimentation rates and low remixing, the highest levels of Hg are buried 10–30 cm below the surface. Combined with previous observations of strong Hg methylation after dredging activities, this argues for leaving Hg-contaminated sediment in place, to be buried by the deposition of cleaner sediments.”

After completion of the dredging of the MSC, Hg mitigation may be conducted to aid the efforts in reducing the amount of Hg in the water column and keep it from being ingested by organisms of concern. Due to high sedimentation of Lavaca Bay and the low release rates of methylated Hg from the sediment, redirecting uncontaminated spoils onto the contaminated areas of Lavaca Bay is the most cost-effective and time-effective mitigation method. Redirecting uncontaminated spoils is essentially “burying” the mercury in the lower sediments. (Bloom 1997) inspects alternative solutions to Hg mitigation involving extraction of Hg from, the sediment. (Bloom 1997) found aqueous distillation to be the most practical approach for Hg extraction. Aqueous distillation was found to avoid “negative interferences” in the extraction process as opposed to other forms of distillation.

Plastics and Microplastics

Plastic and microplastic contamination may also play a significant role in the Lavaca-Matagorda Bay system in relation to contaminant transport (i.e., mercury and PAHs) and food webs. It is known that plastics can negatively impact organisms of different trophic levels, from plankton through bivalves, fishes, birds, and up to large mammals and turtles, through contact or ingestion (Santos-Echeandía et al. 2020 and references therein). However, the effects of microplastics on organisms vary widely because there are many variables to consider and there is still much to learn about transport, uptake, and stress mechanisms.

In areas with metal or organic matter contamination of the sediments, such as Lavaca Bay (mercury and PAHs), higher contaminant concentrations may sorb onto the plastics, particularly polystyrene, due to prolonged contact and greater weathering of the plastic. While in the sediments, the plastics may pose a risk to benthic sediment feeders or detritus feeders (Karlsson et al. 2017, Panti et al. 2015, Priscilla et al. 2019). Resuspension events potentially reintroduce contaminated plastics into the water column where they may once again come into contact or be ingested by estuarine organisms or disperse/transport the plastic and associated contaminants to elsewhere in the environment. Given the known mercury and PAH contamination in Lavaca Bay and their genotoxic (Santos-Echeandía et al. 2020), carcinogenic, mutagenic, and toxic effects (Guven et al. 2018), this raises concerns about microplastics in the sediments and the potential for them to act as contaminant transport vectors.

The role of plastics and microplastics in Lavaca-Matagorda Bay as a vector for contaminant transport is not well constrained. The Matagorda Bay Mitigation Trust has funded research to resolve the prevalence of plastics and measure mercury concentrations, to map the concentration and distribution of microplastic particles in water and sediment, and to assess the toxicity and bioaccumulation of persistent pollutants that adsorb to microplastic particles.

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