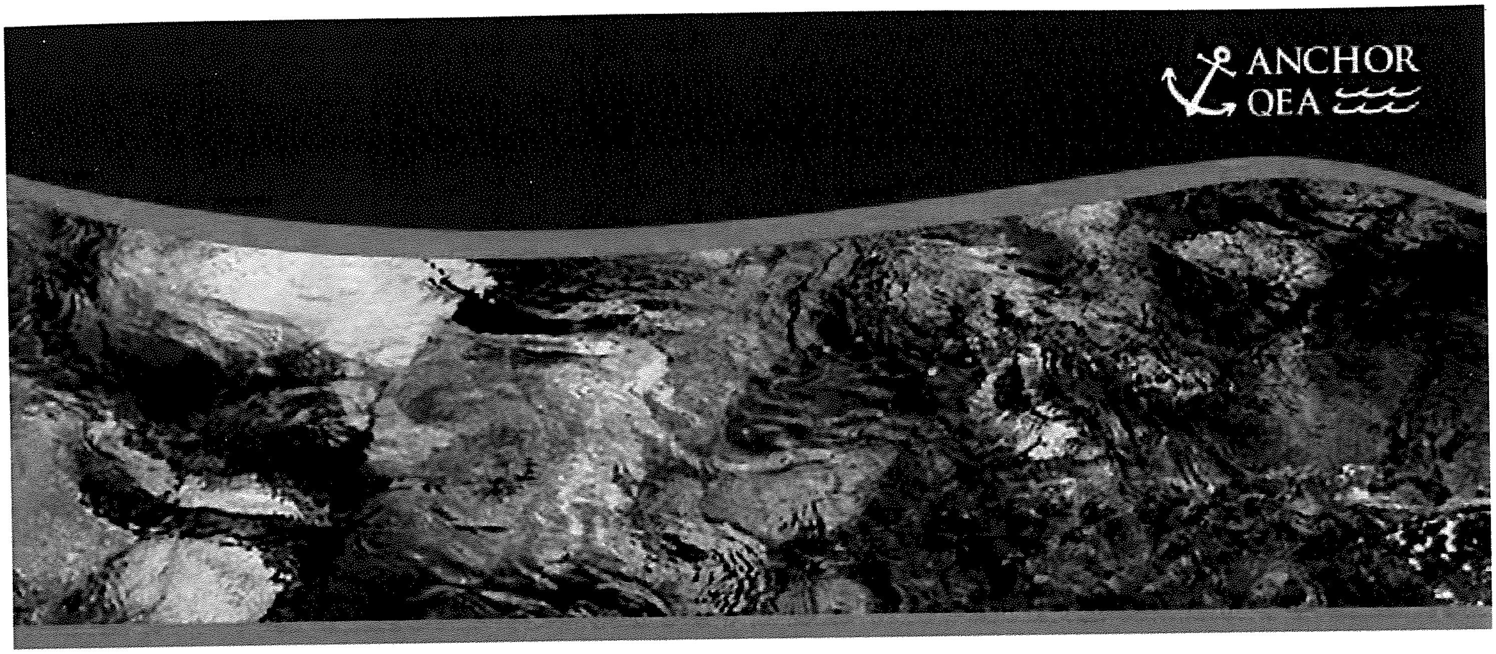

STUDIES TO EVALUATE ACHIEVEMENT OF FRESHWATER INFLOW STANDARDS AND ECOLOGICAL RESPONSE
FINAL REPORT

Prepared for

Texas Water Development Board

Colorado and Lavaca Rivers and Matagorda and Lavaca Bays Basin and Bay Area Stakeholder Committee

Prepared by

Anchor QEA, LLC

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PURSUANT TO SENATE BILL 1 AS APPROVED BY THE 83RD TEXAS LEGISLATURE, THIS STUDY REPORT WAS FUNDED FOR THE PURPOSE OF STUDYING ENVIRONMENTAL FLOW NEEDS FOR TEXAS RIVERS AND ESTUARIES AS PART OF THE ADAPTIVE MANAGEMENT PHASE OF THE SENATE BILL 3 PROCESS FOR ENVIRONMENTAL FLOWS ESTABLISHED BY THE 80TH TEXAS LEGISLATURE. THE VIEWS AND CONCLUSIONS EXPRESSED HEREIN ARE THOSE OF THE AUTHOR(S) AND DO NOT NECESSARILY REFLECT THE VIEWS OF THE TEXAS WATER DEVELOPMENT BOARD.

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LIST OF ACRONYMS AND ABBREVIATIONS

Abbreviation	Definition
°C	degrees Celsius
ac-ft	acre-feet
ANCOVA	analysis of covariance
BBASC	Basin and Bay Area Stakeholder Committee
BBEST	Basin and Bay Expert Science Team
CCA	canonical correspondence analysis
cfs	cubic feet per second
CRD	Colorado River Delta
DCI	dermo condition index
DO	dissolved oxygen
EAMB	eastern arm of Matagorda Bay
FINs	freshwater inflow needs
g/m^2	grams per square meter
GIWW	Gulf Intracoastal Waterway
HSI	habitat suitability index
II	infection intensity
LCRA	Lower Colorado River Authority
LEM	low estuarine marsh
LNRA	Lavaca-Navidad River Authority
LRD	Lavaca River Delta
LSWP	Lower Colorado River Authority – San Antonio Water System Water Project
m^2	square meters
MaxOC	maximum commercial-sized oyster count
MaxWP	maximum dermo weighted prevalence
MBHE	Matagorda Bay Health Evaluation

ME	marsh edge
mg/L	milligrams per liter
MI	marsh interior
mm	millimeters
mS/cm^3	millisiemens per cubic centimeter
NMDS	non-metric multidimensional scaling
NMFS	National Marine Fisheries Service
NTU	Nephthelometric Turbidity Unit
OCI	oyster condition index
PI	percent infection
RFTM	Ray's Fluid Thioglycollate Method
SAWS	San Antonio Water System
SB3	Senate Bill 3
SNB	shallow nonvegetated bottom
STP	South Texas Nuclear Project
TAC	Texas Administrative Code
TCEQ	Texas Commission on Environmental Quality
TDWR	Texas Department of Water Resources
TPWD	Texas Parks and Wildlife Department
TWDB	Texas Water Development Board
USACE	U.S. Army Corps of Engineers
USFWS	U.S. Fish and Wildlife Service
USGS	U.S. Geological Survey
UTMSI	University of Texas Marine Science Institute
WAM	Water Availability Model
WP	weighted prevalence
WUA	weighted usable area

EXECUTIVE SUMMARY

This report describes scientific investigations regarding freshwater inflows and associated ecosystem responses for Matagorda and Lavaca Bays. This work was funded by the Texas Water Development Board (TWDB) and performed on behalf of the Colorado and Lavaca Rivers and Matagorda and Lavaca Bays Basin and Bay Area Stakeholder Committee (BBASC). The purpose of this effort is to provide information and guidance to the BBASC and Texas Commission on Environmental Quality (TCEQ) regarding potential re-evaluation or future study relative to existing freshwater inflow standards for Matagorda and Lavaca Bays found in Subchapter D of Chapter 298 of the Texas Administrative Code (TAC).

Senate Bill 3 (SB 3) of the 80th Texas legislative session (2007) established a framework for identifying and promulgating environmental flow standards throughout Texas. As part of this process, the BBASC (comprised of regional stakeholders) and a Basin and Bay Expert Science Team (BBEST; comprised of regional scientific experts) were established. The BBEST submitted a report containing environmental flow recommendations in March 2011, and the BBASC submitted their report in August 2011. Following a public comment period, TCEQ adopted flow standards for the Colorado and Lavaca basins, effective August 30, 2012. These flow standards include freshwater inflow standards for the Colorado River (into Matagorda Bay) and the Lavaca River and Garcitas Creek (into Lavaca Bay).

These existing inflow standards for Matagorda Bay recommended by the BBEST considered the science and results generated during the Matagorda Bay Health Evaluation (MBHE), which was one of a suite of studies implemented to evaluate a previously proposed water sharing agreement called the Lower Colorado River Authority – San Antonio Water System Water Project (LSWP). These studies generally used data through 2007; however, since 2007, significant additional data have been collected, much of it during an historic drought. Likewise, the existing inflow standards for Lavaca Bay were developed using a similar methodology but based on evaluating fewer ecological indicators than for Matagorda Bay. In this study, similar methods were applied in both bay assessments to the extent possible. The objective of this study is to update and expand the analyses developed through the MBHE and BBEST efforts that related to bay health and key indicator species, including oysters and

dermo (an oyster parasite), marsh vegetation biomass, and juvenile finfish and shellfish density.

Oysters and dermo are both responsive to salinity, which is a function of freshwater inflows, making these species helpful indicators for a determination of freshwater inflows suitable for achieving a sound ecological environment. The long term Texas Parks and Wildlife Department (TPWD) coastal fisheries oyster database that was used for the MBHE effort was updated through 2014. Similarly, the Oyster Sentinel[1] dermo database was updated through 2011, which is the last year of consistently measured dermo data in Texas. A field program was implemented in 2014 through this study to provide additional dermo data; samples were collected from 12 reefs in August and September 2014 and from 6 reefs in November 2014.

In the MBHE effort, a multiple linear regression was developed to relate dermo to salinity and temperature. Because of the limitations of the dataset from Matagorda and Lavaca Bays, in the MBHE study, data from these bays plus Galveston Bay and San Antonio Bay were combined to construct the final regressions.

For the present study, a much larger dermo dataset for Matagorda and Lavaca Bays was available. An examination of these data with the MBHE regressions indicated a general lack of fit of the MBHE regressions with the Matagorda and Lavaca Bays data. Therefore, the regressions for dermo were re-analyzed, and superior fits were obtained based solely on the Matagorda and Lavaca Bays data. The final regression indicated a strong negative relationship between dermo and freshets; the antecedent 5-year probability (i.e., frequency) of salinity less than 2 parts per thousand (ppt) at an oyster reef was a strong predictor of dermo conditions, with an increased frequency leading to reduced dermo. Two additional, statistically significant terms influencing dermo conditions included the following:

- 3-month rolling average temperature lagged by 1 month (recent high temperatures promote dermo and vice versa)

[1] The Oyster Sentinel program was known as the DermoWatch program at the time of the Matagorda Bay Health Evaluation efforts, but will be referred to in this document using the current name of Oyster Sentinel. Data from this program and additional information are available at www.oystersentinel.org.

- 2-year rolling average salinity lagged by 1 year (high antecedent salinities promote dermo and vice versa)

This regression explained 66% of the variance in the dermo data, which is considered very strong for ecological data.

A larger oyster dataset was also available for the present study, relative to the MBHE effort. The updated regression analysis with the full dataset resulted in identical predictor terms to the MBHE regression: commercial-sized oyster counts (i.e., abundance) was a nonlinear function of antecedent 2-year salinity (with an optimal salinity of 20 ppt and decreasing counts at salinities further from 20 ppt) and a freshet term defined as the average number of months between events where the average salinity for a reef location was less than or equal to 2 ppt (with an optimal number of months between freshets of 18). Due to the high variability in oyster counts, and the influence of factors other than salinity on oyster health (e.g., harvest pressure, food supply, and turbidity), the oyster regression explained a minority of the variance in the oyster data (33%).

Similar results were found using long-term average dermo (2004 to 2009) and oyster (1996 to 2014) data across reefs from all three bays.[2] In these cases, the long-term salinity was found to be the dominant predictor of dermo and oysters, while the long-term freshet predictor was the secondary factor.

In addition to extensive oyster and dermo analyses, a series of biological field sampling activities were conducted in Matagorda and Lavaca Bays (specifically, the Colorado River Delta [CRD] and Lavaca River Delta [LRD]) during summer and fall 2014, followed by data analysis and interpretation. Field activities beyond oyster collection included the following ecological categories: 1) marsh vegetation, 2) juvenile finfish and shellfish, and 3) *Rangia* clams. In addition to documenting the presence and status of these organisms during freshwater inflow conditions experienced in these basins in 2014, this additional field collection and data analysis provided further support of certain underlying biological

[2] This report uses the phrase "three bays" to refer to reefs from San Antonio, Matagorda, and Lavaca (considered as one bay system in this context), and Galveston Bay.

relationships developed during the MBHE study. The marsh biomass results confirmed that marsh vegetation does serve as an important ecological indicator, with a detectable response to freshwater inflow. The marsh vegetation results also support a multi-level flow criteria and achievement guideline approach as is currently in place for both bays.

Overall, the species diversity and abundance of juvenile finfish and shellfish species collected in both the CRD and LRD in 2014 were similar to results reported from previous MBHE studies. Unfortunately, nothing in the throw trap data analysis suggests that it is possible to quantify a direct response of freshwater inflow to individual key species or throw trap juvenile finfish and shellfish communities. The juvenile finfish and shellfish analysis does independently support the premise that the throw trap community is influenced by the presence and abundance of vegetative cover and provides statistical evidence that vegetative cover is influenced by salinity. As such, the biological analysis provides additional support for using habitat as a key indicator relative to freshwater inflow. Lastly, the limited investigation conducted during this study for the presence of *Rangia* clams in both delta areas yielded only a single dead shell. Under the current circumstances in which *Rangia* was not present in either bay system, their use as an indicator species was not recommended for this study.

It is acknowledged that the limited 2014 field sampling and analysis associated with this study represents only a snapshot in time for each bay system and should be interpreted with caution. As noted throughout this report, complexities with antecedent inflow conditions and ecological responses of the marsh and juvenile finfish and shellfish communities make additional long-term monitoring and statistical analysis necessary prior to being able to conduct a rigorous validation of the inflow criteria. In order to start alleviating the caveats in this limited biological evaluation for these species, several recommendations for future monitoring and applied research are provided in the final section of this report for BBASC consideration.

The outputs from a hydrodynamic circulation and salinity transport model were used to development regression equations that relate inflows to salinity. Inflow-salinity regression equations are necessary to determine what flow is required to achieve the desired salinity criteria that have been determined from the oyster, dermo, marsh vegetation, and juvenile finfish and shellfish assessments. In general, the development of salinity inflow regression relationships followed the methodology employed in the MBHE study; however, the current study used a different model, TWDB's TxBLEND, which includes a longer period of record and has been updated through 2014. The existing environmental flow standards for Lavaca Bay were developed by the Colorado Lavaca BBEST, and while the BBEST generally applied an approach similar to the MBHE, some differences existed, and these values have now been updated based on the same model and methods that are used for Matagorda Bay.

The results from this study generally support the existing freshwater inflow standards for Matagorda and Lavaca Bays. The updated dermo monthly regression, as well as the oyster monthly regression and the dermo and oyster long-term average regressions, all identify the importance of large freshets, which are not explicitly included in the inflow standards. The BBASC and TCEQ may wish to use these results to consider explicit inclusion of a freshet component in the inflow standards, similar to the high flow pulses that exist in the instream flow standards for the Colorado and Lavaca basins.

The Matagorda Bay inflow standards include a long-term average inflow value. The results described herein support that value. The Lavaca Bay inflow standards do not have a long-term average inflow. The results from this study support a value of 480,000 acre-feet per year for a long-term average inflow value for Lavaca Bay, based on ecological targets consistent with the MBHE and BBEST.

1 INTRODUCTION

This report describes scientific investigations regarding freshwater inflows and associated ecosystem responses for Matagorda and Lavaca Bays. This work was funded by the Texas Water Development Board (TWDB) and performed on behalf of the Colorado and Lavaca Rivers and Matagorda and Lavaca Bays Basin and Bay Area Stakeholder Committee (BBASC). The purpose of this effort is to provide information and guidance to the BBASC and Texas Commission on Environmental Quality (TCEQ) regarding potential revisions to existing freshwater inflow standards for Matagorda and Lavaca Bays found in Subchapter D of Chapter 298 of the Texas Administrative Code (TAC).

1.1 Overview of the Senate Bill 3 Environmental Flows Process

Senate Bill 3 (SB 3) of the 80th Texas legislative session (2007) established a framework for identifying and promulgating environmental flow standards throughout Texas. As part of this process, the BBASC (comprised of regional stakeholders) and a Basin and Bay Expert Science Team (BBEST; comprised of regional scientific experts) were established. The BBEST submitted a report containing environmental flow recommendations in March 2011, and the BBASC submitted their report in August 2011. Following a public comment period, TCEQ adopted flow standards for the Colorado and Lavaca basins, effective August 30, 2012. These flow standards include freshwater inflow standards for the Colorado River (into Matagorda Bay) and the Lavaca River and Garcitas Creek (into Lavaca Bay).

Senate Bill 3 has provisions for continued evaluation of environmental flow standards. In support of this effort, the 83rd Texas legislature set aside $2 million to assist the TWDB and several BBASCs with further evaluations of environmental flows and the associated standards. This report documents the efforts of one such study, with a focus on oyster and marsh ecological responses to freshwater inflows in Matagorda and Lavaca Bays.

1.2 Study Area

Figure 1-1 illustrates the Matagorda and Lavaca Bays system. Highlighted on this figure are the riverine inflows for which standards have been promulgated, including the Colorado River and the Lavaca River plus Garcitas Creek, which are combined in the inflow standards. Figure 1-1 also shows three reef locations and the Shell Marker B Datasonde in the Eastern Arm of Matagorda Bay (EAMB). Salinity trends for these locations are discussed in Section 1.4.

In this report, Matagorda Bay refers to that portion of Matagorda Bay west of the old river channel; therefore, the term does not include the portion to the east of the old river channel, which is commonly referred to as East Matagorda Bay.

1.3 Overview of Existing Freshwater Inflows Studies for Matagorda and Lavaca Bays

Section 1.3 provides a brief overview of studies related to Matagorda and Lavaca Bays. This section includes citations for some key reports and a brief purview of how certain MBHE studies influenced subsequent investigations and recommendations of freshwater inflow standards for the Matagorda and Lavaca Bays.

Pre 1975 dynamics and conditions in Matagorda Bay were studied by McGowen and Brewton (1975). Other reports with relevant historical information on the Matagorda Bay system include USACE (1981), Ward and Armstrong (1980), and Simon (2005).

In the late 1970s and early 1980s, the Texas Department of Water Resources (a predecessor agency to the TCEQ and TWDB) identified freshwater inflow needs (FINs) for the major bays and estuaries of the Texas coast, including Matagorda and Lavaca Bays (referred to as the Lavaca-Tres Palacios estuary system;[3] TDWR 1980, 1982). This work evolved into a series of more recent FINs studies, which are documented in Longley (1994), Martin (1987), Martin et al. (1997), Powell et al. (2002), and LCRA et al. (2006). Federal agencies also evaluated freshwater inflow needs, including a study for Matagorda Bay performed by the U.S. Fish and Wildlife Service (USFWS) and the National Marine Fisheries Service (NMFS; Mueller and Matthews 1987). While some criticisms of the FINs studies and underlying

[3] Prior to 1992, the Colorado River flowed through what is now referred to as the old river channel, carrying a substantial portion of its freshwater out to the Gulf of Mexico, particularly during high flow conditions. In 1991 to 1992, Colorado River flows were rerouted so that the majority of freshwater flows into the Eastern Arm of Matagorda Bay (Wilbur and Bass 1998; Jensen and Lee 2006).

hydrodynamic-salinity TxBLEND model have been posed (e.g., Ward (2004), SAC[4] (2004, 2009), Bao et al. (1989), Wassenich (2004, and citations therein), and Monismith 2005, respectively), the FINs studies greatly advanced the understanding of ecological dynamics in Texas bays and estuaries and set the stage for subsequent efforts.

In the early 2000s, the Lower Colorado River Authority (LCRA) and the San Antonio Water System (SAWS) agreed to fund a series of studies to evaluate the implications of a proposed water sharing agreement known as the LCRA-SAWS Water Project (LSWP). With oversight by a Scientific Review Panel and input from stakeholders, a team of consultants and agency staff generated a series of reports covering a range of environmental and socio-economic issues, including freshwater inflows to Matagorda Bay. The most relevant freshwater inflows reports include those prepared by the Matagorda Bay Health Evaluation (MBHE) team and Texas A&M University-Corpus Christi:

- Hydrodynamic and salinity modeling (MBHE 2006a)
- Nutrients and algal production (MBHE 2007a)
- Biological statistics (MBHE 2006b)
- Marsh and oyster habitat, abundance, and productivity (MBHE 2007b)
- Benthic macroinvertebrates (TAMU-CC 2007)
- Final freshwater inflow recommendations report (MBHE 2008a)

Through the public participation process, and to the benefit of the final reports, a number of state agencies, non-governmental organizations, and other stakeholders provided a variety of formal and informal comments and criticisms. Ultimately, using a combination of ecological science and historical hydrology, the MBHE team recommended a suite of inflows consisting of magnitudes and attainment frequencies for several inflow levels and averaging periods.

In developing their freshwater inflow recommendations, the BBEST relied heavily upon the MBHE reports: “The recommended suite of Matagorda Bay Inflow Criteria for the Colorado River . . . was adopted from the MBHE study” (BBEST 2011). The BBEST also

[4] The Science Advisory Committee (SAC) that authored the 2004 report is not the same SAC as has been involved in Senate Bill 3 efforts, although three members served on both SACs (Dr. Robert Brandes, Dr. Paul Montagna, and Dr. George Ward).

adopted data and analysis approaches similar to MBHE methods in their development of freshwater inflow recommendations for Lavaca Bay (note, MBHE did not address freshwater inflows for Lavaca Bay). The BBASC relied heavily upon the BBEST recommendations: "The Committee agreed to recommend that the BBEST recommended values, with certain limited adjustments, should be included in the environmental flow standards" (BBASC 2011). Finally, the promulgated TCEQ standards relied on the BBASC recommendations: "The proposed . . . standards for Matagorda and Lavaca Bays generally track the recommendations of the stakeholders" (TCEQ 2011). As a result, the science underlying the MBHE studies had a strong influence on the adopted freshwater inflow standards for Matagorda and Lavaca Bays.

1.4 Overview of Historical Hydrology and Salinity

In the years since the completion of the MBHE inflow report, Texas has experienced a significant drought, including, but not limited to, the record setting year of 2011 (Hoerling et al. 2013). More recently, the LCRA has suggested that the current drought in the Colorado Basin is the worst on record, which has significant implications for water supply and environmental flows (http://www.lcra.org/about/newsroom/news-releases/Pages/Drought-conditions-worsen-along-Highland-Lakes.aspx). From 2012 through 2014, irrigation districts did not receive deliveries from the LCRA (excepting the Garwood irrigation district, which receives water under a different contractual arrangement than the other districts). At the time of this writing (summer 2015), substantially wetter conditions have returned to Texas; even so, drought has not been broken in the LCRA's Highland Lakes, and the irrigation districts are once again not receiving water deliveries this year.

Figure 1-2 illustrates the combined annual inflows to Matagorda and Lavaca Bays from the Colorado River, Lavaca River, and Garcitas Creek from 1977 through 2014. The TCEQ inflow standards for Matagorda Bay are based on "the most downstream point in the Colorado River Basin for Matagorda Bay" (TAC §298.305). For the purposes of this study, this is interpreted as the Colorado River near Bay City U.S. Geological Survey (USGS) gage, minus the South Texas Nuclear Project (STP) diversions and other downstream diversions, plus downstream return flows, plus the estimated watershed runoff downstream of Bay City that enters the Colorado River above the Gulf Intracoastal Waterway (GIWW). Similarly, the TCEQ inflow standards for Lavaca Bay are based on “the most downstream point in the Lavaca River Basin and at the most downstream point on Garcitas Creek” (TAC §298.305). For the purposes of this study, this is interpreted as the Lavaca River near Edna (USGS flow gage), plus releases from Lake Texana, plus Garcitas Creek near Inez (USGS flow gage), plus downstream return flows, plus estimated watershed runoff downstream of these gages that enters the Lavaca River and Garcitas Creek, and minus downstream diversions. These inflows are described in greater detail in Section 5.

Figures 1-3a through 1-3i illustrate daily records of inflows from the Colorado River and salinity from 1996 to 2014. Flow data were obtained from the USGS and TWDB (see Section 2). Salinity data were obtained from two sources as follows:

- Daily salinity data were downloaded for the Shell Marker B Datasonde from http://waterquality.lcra.org/ (this datasonde is located on Shell Island Reef; Figure 1-1).
- Monthly salinity data were obtained from the Texas Parks and Wildlife Department (TPWD) for Sammy’s Reef, Mad Island Reef, and Shell Island Reef (see Section 2), all of which are located in the EAMB (Figure 1-1).

As shown in Figures 1-3a through 1-3i, recent inflows through 2014 have been relatively low, and salinity has been relatively high, especially during 2009, 2011, and 2013. The highest salinity period observed is found in late 2011, where salinities in excess of 35 parts per thousand (ppt) occurred, with few interruptions, for more than 3 months. Given that average ocean salinity is about 35 ppt, and the Gulf of Mexico is often lower than this, salinities in excess of 35 ppt can be considered hypersaline: dissolved salt concentrations are higher than ocean water. Hypersaline conditions occur when evaporation exceeds freshwater inflows; these conditions are relatively common in secondary and tertiary bays in south Texas (Forbes and Dunton 2006).

Annual inflows from 2008 to 2014 have all been below average. The MBHE effort, which used data through 2007, included many average and above-average years, interspersed with some relatively short dry periods. Accordingly, the analysis described herein using data collected during the recent drought provides an informative and useful expansion of the MBHE work.

1.5 Purpose of this Study

Since the culmination of the MBHE effort (which used data through 2007), additional data related to marsh habitat and productivity, and oyster abundance and parasitism, have been collected. The purpose of this study is to compile and collect new data (through the end of 2014) and to update and expand upon the marsh productivity and oyster evaluations that were performed in support of the MBHE. This information is expected to either corroborate the existing inflow standards or suggest new relationships between freshwater inflows and ecosystem response that the BBASC and TCEQ can use to guide a potential re-evaluation of those standards.

2 OYSTER HEALTH EVALUATION

The eastern (also called American) oyster, *Crassostrea virginica*, is one of the key species being evaluated by the project team. The work presented here updates analyses conducted for the MBHE in 2006 through 2008 (MBHE 2006c, 2007b, and 2008a; generally termed the 'MBHE effort') that developed oyster suitability criterion models to describe the linkage between oyster population trends and salinity regimes in the Matagorda Bay system.

Section 2 begins with a brief overview of the effects of salinity on oysters and the common and debilitating oyster parasite *Perkinsus marinus*, commonly known as 'dermo'. Next, a review of the analysis approach and results of the MBHE effort is provided. Finally, the current updated analysis approach and results are described, including incorporation of new data through 2014, review of data trends, and presentation of updated suitability criterion model results. Results are presented with a focus on the impacts of the recent, severe, multi-year drought on oysters and dermo in Matagorda and Lavaca Bays. The implications of these results for freshwater inflow criteria are discussed in Section 6.

2.1 Influence of Environmental Factors on Oysters and Dermo

The ecology of oysters and dermo, especially as related to freshwater inflows and salinity, has been described at length in the literature. Accordingly, this section provides a brief synopsis coupled with numerous citations containing additional information.

Estuarine habitats, including those within Texas' bays, are characterized by large changes in salinity over a broad range of temporal and spatial scales. To survive in such variable conditions, many estuarine species, including oysters, have adapted to be euryhaline, i.e., tolerant of a wide range of salinity conditions. Oysters can survive in salinities ranging from about 5 to 40 ppt, but growth is stunted below 7.5 ppt (Kennedy et al. 1996). Oysters have been reported to grow optimally at average salinities between 10 and 20 ppt (Cake 1983); 10 and 15 ppt (Soniat et al. 2013); and 14 and 28 ppt (Kennedy et al. 1996). The variation reported in the literature may be due to local acclimatization by the oysters and localized salinity effects on food organisms, competitors, predators, and parasites. When subjected chronically or episodically to low salinities due to excessive freshwater runoff, oyster reefs may experience degraded conditions ranging from complete or partial population mortality to stunted growth. However, reefs that are located in regions of chronic or seasonally high salinities (greater than 25 ppt) will have a greater mortality due to predation and dermo[5] (Kennedy et al. 1996).

High levels of infection by the parasite dermo (the term 'dermo' describes both the infection and the organism, formerly classified as *Dermocystidium marinum*), are common in Gulf of Mexico oyster reefs. The parasite can severely reduce oyster reproduction and survival rates and decrease the market value of oysters by degrading their meat.

Dermo infections are strongly affected by temperature and salinity regimes, although in the Gulf of Mexico, salinity is considered to be more important than temperature (Soniat et al. 2009). The gross growth and mortality rates of dermo cells within oysters are related to both temperature and salinity of the surrounding water (Hofmann et al. 1995). Figure 2-1, adapted from Hofmann et al. (1995), illustrates the net growth of dermo as a function of temperature and salinity.

As shown in Figure 2-1, at salinities greater than 10 ppt, new growth is primarily affected by, and increases with, temperature; however, below 10 ppt salinity, dermo net growth (i.e., cell growth in excess of cell mortality) decreases sharply with salinity, becoming negative, regardless of temperature, below approximately 5 ppt. An inspection of data from the Oyster Sentinel program shows that, in western Gulf of Mexico bays, oyster populations on reefs closer to freshwater inflow sources typically have lower dermo infection intensities than reefs closer to the seawater source at the bay's mouth.[6] However, salinities low enough to curtail dermo populations can also stress oysters, so the ideal salinity regime to control dermo is one with episodic freshwater inflows that reduce salinity below 10 ppt, rather than continuously low salinities (Hofmann et al. 1995, Kennedy et al. 1996).

Temperatures in the Gulf of Mexico do not get low enough to cause oyster mortality; however oyster growth rate is slower in colder temperatures. The combination of physiological effects of low temperature and limited food supply, in general, can cause

[5] A historical perspective on the discovery of dermo is provided in Ray (1996) and a review of the organism can be found in Villalba et al. (2004).
[6] For example, compare North Reef and Indian Point Reef in Lavaca Bay on OysterSentinel.org.

individual oysters to lose biomass in winter months (Hofmann et al. 1992). Optimal temperature for oysters is approximately 25 degrees Celsius (°C), and temperatures over 30°C can cause cessation of filter feeding (Kennedy et al. 1996). Oysters achieve commercial size (3 inches or 76 millimeters [mm]) approximately 2 years after spat set in the northern Gulf of Mexico (Ingle and Dawson 1952, as cited in Wilbur and Bass 1998).

Dermo infections tend to be most intense in summer when temperatures are highest. Temperatures above 25°C are associated with high oyster mortality from dermo, and the persistence of high-intensity dermo infections is dependent on extended periods of high temperature (Kennedy et al. 1996). Such periods are common during summers in Texas bays.

Once an oyster is infected with dermo, it will never lose the infection completely (Hofmann et al. 1995; Ragone Calvo and Burreson 1994). Dermo acts as a drain on the oyster's energy reserves, often leading to reduced growth (Villalba et al. 2004). However, if conditions (e.g., salinity, temperature, food supply) are optimal for the oyster, the oyster can grow faster than dermo, thereby diluting the infection and delaying oyster mortality (Hofmann et al. 1995; Soniat et al. 2012). Ultimately, as the oyster ages and its growth rate declines, the dermo infection will intensify and may cause mortality. Oyster mortality rates of 50% in the second summer of infection, 80 to 90% by the third summer, and near 100% within 4 years of infection have been reported (Eastern Oyster Biological Review Team 2007). It has been estimated that 50% of the yearly mortality of commercial-sized oysters in the Gulf of Mexico is due to dermo (Hofstetter et al. 1977, as cited in Britton and Morton 1989).

The standard assay for determining the level of dermo parasitism is the Ray's Fluid Thioglycollate Method (RFTM; Ray 1966). A small piece of tissue is removed and assayed for disease after incubation in fluid thioglycollate and antibiotics for approximately 1 week. Dermo intensity is scored using a 0 to 5 scale developed by Mackin (1962), where 0 is no visible infection,[7] and 5 is an infection in which the oyster tissue is almost entirely obscured by the parasite (Craig et al. 1989). Only live oysters can be measured for dermo, and in Texas, it is rare to find live oysters with an infection intensity of 5 (a study of 1,688 oyster samples across five bay systems in Texas resulted in just one oyster with a Mackin score of 5; TPWD and TWDB 2011).

Dermo infections tend to be highly variable among a population. To allow for a reasonable representation of a population, 12 oysters are typically collected at each location. Calculations are made of percent infection (PI), weighted prevalence (WP), and infection intensity (II). PI is the number of infected oysters divided by the number of oysters assayed times 100. WP is the sum of the Mackin disease code numbers divided by the total number of oysters in the sample:[8]

$$WP = \frac{\sum Mackin\ scores}{Number\ of\ oysters\ tested}$$

WP is the summary statistic used in this report to describe dermo in a group of oysters. A third descriptor for dermo in a group of oysters is Infection Intensity (II), which is calculated as the sum of the Mackin codes divided by the number of infected oysters in the sample. Because II ignores uninfected oysters, this study did not use II as a descriptor of dermo conditions and instead focused on WP as a primary indicator variable.

A WP of 1.5 is considered the level at which significant disease-related mortalities are occurring (Mackin 1962; Bushek et al. 2012). Mackin (1962) states that a population of live oysters with a WP of 2.0 "contains an intense epidemic, and more than half of the population may be in advanced stages of the disease, with all of the individuals infected." Similarly, Bushek et al. (2012) state "relatively high [annual] mortality (≥25%) occurred where median [WP] routinely exceeded 2.0." As a final point of reference, for the data from Matagorda and Lavaca Bays used in this study, the maximum WP ever recorded was 2.9 (Mad Island Reef, September 2010).

For any given bay system, significant temporal and spatial variations in salinity and temperature occur. Because both oysters and dermo respond to changes in salinity and

[8] Terminology describing summary statistics for dermo is variable in the literature. The weighted prevalence used herein is synonymous with "weighted incidence" used in MBHE (2008) and Soniat (1996), and "sample intensity" used in Bushek et al. (2012).

temperature, oyster reefs tend to accrete in locations where conditions are generally favorable for oysters; however, reef locations may not be favorable at all times for oyster health due to dynamic conditions within bays. Oyster reefs may also occur in locations that generally have water that is more fresh than optimal, and these locations can provide refuge during drought. Other reefs may occur in locations that are generally more saline than optimal and these locations provide refuges during floods. An optimally functioning bay system includes multiple reef locations such that under extreme climate regimes, at least one reef has suitable conditions for survival and can provide adequate recruitment to recolonize other reefs when conditions normalize.

The literature reports that salinity impacts both oysters and dermo; however, this impact is not expected to be instantaneous. For dermo, lags between increases (or decreases) in salinity and corresponding increases (or decreases) in dermo have been identified in the literature of 6 months (Soniat et al. 2005) and 1 year (Bushek et al. 2012). Similarly, for commercial-sized oysters, a lag of 1 to 2 years between decreased salinity and increased oyster abundance has been identified for Galveston Bay (Buzan et al. 2009). These observations in the literature suggest that a statistical analysis of dermo and oysters versus salinity should consider lag terms rather than simply the salinity value at the date and location of sampling.

2.2 Summary of Matagorda Bay Health Evaluation Efforts

The MBHE habitat assessment identified baseline environmental conditions within Matagorda Bay and investigated relationships between key aquatic species and habitat based on biological, chemical, and physical variables. These variables included freshwater inflow, salinity, temperature, inundation regime, physical features and/or substrate type, and organism abundance. The MBHE effort on the Eastern oyster developed condition indices and suitability criterion models to describe the biological linkage between oyster condition as indicated by commercial-sized oyster counts and dermo WP to salinity regime and temperature in the Matagorda Bay system.

An overview of the MBHE effort is provided in this section, including the development of long-term data series for 20 individual reef locations, oyster and dermo condition indices, and suitability criterion models.

2.2.1 Data and Reef Locations

The datasets chosen to analyze trends in oysters and dermo, and the reasons for these choices, are described in detail in MBHE (2006c). Two datasets are used: TPWD coastal fisheries oyster dredge data and Oyster Sentinel program data.

These two datasets, from long-term programs with frequent sampling events across all seasons, were chosen because their sampling programs had key characteristics needed to analyze trends of oyster and dermo response to salinity conditions throughout Matagorda and Lavaca Bays. In addition, the programs sampled across widespread areas, allowing results to be compared for many different reef locations with differing salinity regimes within Matagorda, Lavaca, and other nearby bays.

Temperature, salinity, and average live oyster counts (total and commercial-sized) per dredge pull were obtained from TPWD's coastal fisheries oyster dredge database. Dermo data plus additional temperature and salinity data were acquired from the Oyster Sentinel monitoring program.

Following the approach detailed in MBHE (2006c), results from TPWD and Oyster Sentinel stations were grouped to improve temporal continuity of data at individual reef locations for this study. Both the TPWD and the Oyster Sentinel monitoring programs operated on a monthly frequency, but not all locations were sampled in all months. The Oyster Sentinel reefs were sampled at a maximum frequency of once per month, but missing months were common. TPWD sampling also occurred once per month, but only at a randomly selected subset of locations (sampling grids) in each bay for each month, so any single TPWD grid was only sampled a few times per year. All Oyster Sentinel sampling occurred at locations that were also sampled by the TPWD program, and, typically, multiple TPWD sampling grids existed per Oyster Sentinel reef. To improve the chance of having data for each reef in any given month, contiguous TPWD sampling grids were grouped with each other and with

Oyster Sentinel stations in the same reef areas in Matagorda, Lavaca, Galveston, and San Antonio bays. Thirteen reefs were based on established Oyster Sentinel locations (these 13 reefs correspond to all of the reefs in these bays with dermo data from Oyster Sentinel except leased reefs in Galveston Bay, which were excluded from the analysis). Seven additional reefs were chosen in Matagorda, Lavaca, and San Antonio Bays, where there were contiguous TPWD grids with relatively continuous oyster data, thereby providing sufficient data to support the statistical analyses (Figure 2-2; MBHE 2006). These reefs are listed in Table 2-1.

For each reef, any available data in each calendar month from either a TPWD sampling grid or from Oyster Sentinel was averaged to obtain a monthly average for that reef (grids and presence of Oyster Sentinel stations for each reef are indicated in Table 2-1).

Each reef had from 1 to 9 TPWD sampling grids and was potentially an Oyster Sentinel sampling site. This resulted in a range from 2 data sources (at Shell Island Reef, which had one TPWD grid and was an Oyster Sentinel location) to 10 data sources per reef (Table 2-1). However, even after combining data from both programs and multiple sampling grids in this way, all reefs had some months without data, and the smaller reefs, which generally had fewer sampling stations, had more months with missing data. Despite the resulting occasional data gaps, this approach to data organization provided for a relatively continuous monthly time series, particularly for temperature and salinity records, which were available from both monitoring programs.

These monthly averages form the basis of many of the statistical analyses performed in this study. It is important to recognize that these are not monthly averages based on daily data, but are averages based on a few discrete measurements during each month. As seen in Figure 1-3a through 1-3i (Section 1.5), which includes monthly average TPWD data from Shell Island Reef and co-located daily salinity measurements from the LCRA Shellfish Marker B Datasonde, the monthly averages may occasionally miss short-term fluctuations, but they capture most of the patterns shown in the daily salinity data.

Figure 2-3 illustrates reef locations in Matagorda and Lavaca Bays that have historically been sampled for oysters and have sufficient data to support the analyses described in this report.

In the eastern portion of the figure is the EAMB, the eastern end of which contains the mouth of the Colorado River and the Colorado River Delta (CRD). Reefs in the EAMB, from east to west, include Shell Island Reef, Mad Island Reef, and Sammy's Reef.[9] In the western portion of the figure is Lavaca Bay, the north end of which contains the mouth of the Lavaca River and the Lavaca River Delta (LRD). From north to south, the Lavaca Bay reefs include North Reef (sometimes referred to as Upper Lavaca Reef), Route 35 Bridge (sometimes referred to as Lap Reef), Gallinipper Point, and Indian Point (sometimes referred to as Middle Ground Reef). Matagorda Bay also included three central reef locations, not clearly tied to input from the CRD or the LRD. These are Half Moon Reef[10] and reef groups in Tres Palacios Bay and Carancahua Bay.

2.2.2 Oyster and Dermo Condition Index Development

A prerequisite to developing oyster suitability criterion models is the definition and development of an index to describe oyster reef condition. The MBHE effort developed two condition indices based on the commercial-sized oyster population at each reef. These are the Oyster Condition Index (OCI), based on the count of live oysters greater than or equal to the legal harvest size of 76 mm (commercial-sized oysters) over the course of a 30-second dredge pull,[11] and Dermo Condition Index (DCI), based on the dermo infection severity in commercial-sized oysters. These two indices were used for this study.

The indices were developed for commercial-sized oysters (rather than total oysters, which includes oysters of all sizes) because attainment of an adult, harvestable population, and the heath of that segment of the population, may be considered the best indication of oyster reef population quality (Soniat 2005). Oyster count alone does not fully describe oyster reef condition, especially given the tendency for oyster size and growth to be stunted at very high densities due to resource competition between oysters (Hofmann et al. 1992, 1995). Stunted oyster populations result in limited commercial value and poor reproduction. Furthermore,

[9] Shell Island Reef and Mad Island Reef are historical reefs. Sammy's Reef was constructed in 1995 by the U.S. Army Corps of Engineers (Culbertson 2008).
[10] Half Moon Reef in this document refers to the historical reef that has been sampled by TPWD, which has also been called Oyster Lake Point in the MBHE reports. Half Moon Reef does not refer to the recent oyster reef restoration project led by The Nature Conservancy, which is located nearby.
[11] The Oyster Condition Index used herein is different from the oyster condition index used in some literature (e.g., La Peyre et al. 2003) that is based on the ratio of dry weight of the oyster tissue to dry weight of the shell.

the tendency for dermo infections and subsequent mortality to worsen with oyster age and size may result in reefs with high numbers of juvenile oysters but few or no individuals of harvestable size and reproductive age.

The OCI is based on commercial-sized oyster abundance, log-transformed and normalized to a 0 to 1 scale (analogous to a Habitat Suitability Index, or HSI; Soniat et al. 2013):

$$OCI = \frac{\log_{10}(CommercialOysterCount + 1)}{\log_{10}(MaxOC + 1)}$$

This index relies upon a maximum observed commercial-sized oyster count (MaxOC): if the oyster count equals MaxOC, then this would be the best condition observed; therefore, OCI equals 1. The MaxOC for the MBHE effort was 82. This maximum is from the three-bay system for 1996 to 2007.

The DCI is dermo WP in commercial-sized oysters, log-transformed and normalized to a 0 to 1 scale.

$$DCI = 1 - \frac{\log_{10}(DermoWP + 1)}{\log_{10}(MaxWP + 1)}$$

Maximum dermo WP (MaxWP) for the MBHE effort was 3.25.[12] DCI decreases as dermo intensity increases. A higher DCI indicates a more parasite-free oyster population, hence, a better condition.

The DCI was only calculated for times and locations when dermo was measured, which is a subset of the data available for OCI. Only 13 of the 20 reef locations were sampled for dermo, and Oyster Sentinel data are only available (depending on the reef) from 1998 to

[12] The DCI calculation, including the value of MaxWP, is used to scale the dermo WP values to allow for a more normally distributed dataset, which has statistical advantages. The choice of MaxWP is largely irrelevant (as long as it exceeds the maximum of the dataset), because the final dermo WP results and interpretations are based on a back-transformation of the DCI statistics; hence the MaxWP value cancels itself out.

2011 in Galveston Bay and from 2003 or 2004 to 2011 in Matagorda, Lavaca, and San Antonio bays.[13]

2.2.3 Previous Suitability Criterion Models

The MBHE oyster effort focused on development of oyster suitability criterion models based on use of statistical multiple regression modeling to predict OCI and DCI. This section summarizes the work performed in the MBHE. Separate analyses were performed for OCI and DCI to determine the best predictive regression model for each index. These best models were the designated suitability criterion models for oyster reef condition in Matagorda Bay as reported in MBHE (2007b). Following discovery of some Oyster Sentinel database errors, the DCI model was updated in MBHE (2008a).

To prepare data inputs for the suitability criterion models, average monthly salinity, temperature, dermo WP, and oyster counts were calculated for each month for each reef location from 1996 through 2007 (Section 2.2.1). Then, as detailed in MBHE (2006c), the monthly statistics were used to calculate longer-term temperature and salinity variables (e.g., 2-year average salinity, 3-month average temperature). Oyster reef condition may be dependent on temperature and salinity forcing functions operating on a variety of temporal scales, including seasonal, annual, and multi-annual. The approach for suitability criterion model development compared oyster condition in current months to temperature and salinity in prior months or years in order to provide the best predictive model. Variables indicating the frequency of freshwater inundations in prior time periods were also constructed. These variables are described in detail in MBHE (2006c). Additional variables constructed and tested in the current effort are described in Section 2.8. The variables were constructed as rolling averages so that, for example, a 2-year salinity average for a particular month would be the average of salinity values at that reef location for the 24 months including and prior to that month. As discussed in Section 2.2.1, all reefs had some months without data for salinity, temperature, oyster counts, and/or dermo WP. Missing data rules were developed such that no rolling average was constructed for time periods with too few monthly data results in the averaged period. Missing data rules are also detailed in MBHE (2006c).

[13] The dermo program was terminated in September 2011 by TPWD due to budget cuts.

Multiple regression model-building techniques (MBHE 2006c) were then used to develop models using combinations of temperature and salinity parameters that had the best ability to predict observed OCI and DCI results. The objective was to develop relatively simple models (few terms) that had high predictive power and were consistent with known physiological characteristics. The use of 3-month rolling averages of OCI (3MRA OCI) and DCI (3MRA DCI; both metrics were only calculated where all 3 months of data were available) rather than monthly OCI and DCI results reduced some of the month to month "noise" of environmental heterogeneity and resulted in stronger regression models.[14]

The single best predictor of 3MRA DCI was the prior 2 years' salinity average, with dermo increasing, and therefore DCI decreasing, as salinity increases. This result is consistent with the known biological response of the parasite to higher salinities. The best multiple regression model found during the MBHE effort included two additional temperature terms with this salinity term: the prior 2 years' spring temperature average (higher spring temperatures decreased dermo presumably due to better oyster growth; MBHE 2008a), and the prior 3 months' rolling temperature average (higher recent temperatures increased dermo).

The final multiple regression model for 3MRA OCI included terms for 10-year low salinity event frequency (calculated as the average number of months between events where the average salinity for a reef location was less than or equal to 2 ppt) and the prior 2 years' salinity average. Both of these terms were entered as 2-term polynomials, indicating that intermediate values in the ranges of average salinity and freshet frequency were optimal for OCI. An additional term in the model was the prior 2 years' average winter temperature (higher winter temperatures presumably increased oyster growth).

Even with the use of 3MRA OCI rather than monthly OCI, the variability in oyster counts was high and the OCI suitability criterion model did not have as high an explanatory power as the DCI model (multiple regression R^2 equaled 0.35 and 0.56, respectively; MBHE 2007b,

[14] Stronger regression models are those that explain more of the variance in the data, as quantified by the coefficient of determination [R^2]).

2008a). Consequently, the decision was made to only retain the DCI model predictions for the development of inflow standards (MBHE 2008a).

2.3 2014 Oyster Field Collection

The termination of the Texas portion of the Oyster Sentinel monitoring program in mid-2011 resulted in a lack of information about the dermo response in Matagorda and Lavaca Bays to the long-term drought, which continued for several more years without an intervening wet period. In 2014, field sampling efforts were undertaken to help address this data gap.

2.3.1 Methods and Materials

Oyster sampling took place during two separate field events, targeting late summer (late August and early September) and late fall (mid-November) 2014. The late summer event was intended to capture the typically high dermo levels seen at the end of the hot season, and the late fall event was intended to capture either continuation of relatively high dermo levels or a reduction in dermo following seasonal cooling and/or a high inflow event. Fall 2014 was warm and dry without a high inflow event.

During late summer, oysters were sampled at 4 locations in Lavaca Bay (Figure 2-4) and 8 locations in Matagorda Bay (Figure 2-5). Sampling locations were chosen to capture data along a salinity gradient based on long-term salinity averages of each bay system. In Lavaca Bay, oyster sampling sites included Gallinipper Point, Indian Point, Route 35 Reef, and North Reef. In Matagorda Bay, the oyster sampling sites chosen were Sammy's Reef, Mad Island Reef, Shell Island Reef, and five sites from the previous MBHE study conducted within the CRD. Of these 12 original sites, 6 were chosen for resampling during late fall, including Sammy's Reef, Indian Point, Mad Island Reef, North Reef, CRD 5, and CRD 6.

Sites were sampled using either a BIO-WEST custom fabricated steel-frame oyster dredge (Figure 2-6) or a 1/3-square meter (1/3-m^2) quadrat basket sampler, depending on water depth at each site. The dimensions of the oyster dredge are 33 inches (80 centimeters) long by 18 inches (47 centimeters) wide by 11 inches (29 centimeters) deep with a 0.5 inch (1.3 centimeters) wire mesh lined collection basket to retain small shell hash and bivalve/benthic species. The oyster dredge was towed via boat for 30 seconds at a speed of 2 miles per hour. The start and end location of each dredge tow was recorded using a Trimble® GeoXT 7000 real-time DGPS. Upon retrieval of each dredge tow and quadrat sample, contents were described and documented by photograph. Photographs of the oyster sampling quadrat and the oyster dredge equipment and deployment in the field are provided in Appendix A.

At each sample site, sampling was performed with the goal of obtaining 15 live commercial-sized oysters (greater than or equal to 76 mm in length) to send to the laboratory for dermo analysis. This goal was met for all but one site (see Section 2.3.2). All live commercial-sized oysters were measured in 5 mm incremental size classes and recorded. All collected oysters were rinsed in ambient water, double-bagged in gallon size plastic bags along with site information, and stored with frozen gel packs in a cooler. Samples were delivered overnight to Dr. Tom Soniat at the University of New Orleans in Louisiana. Only commercial-sized oysters were collected because dermo data from juvenile oysters were not used in suitability criterion modeling efforts and because adults typically are more heavily infected than juveniles. Dr. Soniat and his staff performed dermo assays individually on oysters (as per the scope of work, dermo assays were performed on three fewer oysters than were delivered, for each site) to determine dermo WP. All samples were assayed within 1 week of delivery. These methods are the same as those performed for the Oyster Sentinel program and previously during MBHE investigations.

Standard water quality parameters were collected at each sample location using a YSI 6920 multi-parameter water quality datasonde. The parameters measured included temperature (°C), specific conductivity (millisiemens per cubic centimeter [mS/cm^3]), salinity (‰), dissolved oxygen (DO; milligrams per liter [mg/L]), pH (standard units) and turbidity (Nephelometric Turbidity Units). Water depths (feet) were measured and recorded using a non-vented strain gauge on the YSI 6920 datasonde.

2.3.2 Oyster Sampling Results

Late summer sampling was conducted on August 25, 2014, and September 1, 2014, at each of the 12 targeted locations in Lavaca and Matagorda Bays. All sample sites in Lavaca Bay were sampled using the oyster dredge. Within Matagorda Bay, Sammy's Reef, Mad Island Reef, and Shell Island were sampled using the oyster dredge. The remaining five sites in the CRD were accessed by airboat and hand-sampled utilizing the 1/3 m^2 quadrat basket sampler. The quadrat basket sampler was used for these samples because CRD oyster reefs are too shallow to navigate to via motorboat. All sites except Sammy's Reef yielded the targeted 15 live commercial-sized oysters. After a concerted effort, ten live commercial-sized oysters were collected at Sammy's Reef. All oysters were packaged and sent to Dr. Soniat's laboratory for dermo analysis. Water depths at the sampling locations ranged from 5.5 to 7.0 feet at the Lavaca Bay sites and 1.3 to 6.0 feet at the Matagorda Bay sites. Salinities at Lavaca Bay sample locations ranged from approximately 21 to 31 ppt, and salinities at the Matagorda Bay sites ranged from approximately 26 to 32 ppt (Table 2-2). Water quality information collected at each of the sites is presented in Appendix B.

Late fall oyster sampling was conducted on November 15, 2014, at the six sample sites chosen for resampling. Sites were chosen based on the expected salinity gradients within each bay and to include the larger fringe reef sites located east and west of the CRD. Two Lavaca Bay sample sites were chosen for resampling (Indian Point and North Reef), with each being sampled using the oyster dredge. Sites resampled in Matagorda Bay included Mad Island Reef, Sammy's Reef, CRD 5, and CRD 6. Oyster sampling for all four sites utilized the oyster dredge due to high water levels at the time of sampling. Therefore, no sites during the late fall were sampled by utilizing the quadrat sampler. The late fall oyster sampling event yielded the targeted 15 live commercial-sized oysters at all six sites. Water quality parameters were recorded at each of these six sites. Water depths at the sampling locations ranged from 4.0 to 5.0 feet at the Lavaca Bay sites and 1.5 to 7.0 feet at the Matagorda Bay sites. Salinities at Lavaca Bay sample locations ranged from approximately 25 to 29 ppt, with salinities at the Matagorda Bay sites ranging from approximately 25 to 31 ppt (Table 2-3). Water quality information collected at each of the sites is presented in Appendix B.

2.3.2.1 Dermo Analysis Results

For the summer oyster sampling event, PI ranged from 0 at North Reef to 91.7% at CRD 4, CRD 6, and CRD 8 (Table 2-2). The WP ranged from 0 at North Reef to 1.28 at CRD 8 (Table 2-2). The II ranged from 0 at North Reef to 1.92 at Indian Point (Table 2-2). While there may be some dermo-related mortalities occurring at reefs with higher WP and II values, none of the reefs have disease levels indicative of outright epizootic[15] conditions (Mackin 1962; Bushek et al. 2012; Powell et al. 1996).

Disease levels from Sammy's Reef, Gallinipper Point, Indian Point, Mad Island Reef, North Reef, and Shell Island are available from Oyster Sentinel for this study and previous assays. These records allow a comparison of present disease levels to past trends. Of the reefs common between this study's field collection and the Oyster Sentinel program, Sammy's Reef, Gallinipper Point, Indian Point, and Mad Island Reef had lower disease levels, whereas North Reef and Shell Island had similar levels when compared to the most recent summer data available (2011; e.g., compare values in Table 2-2 to summer 2011 in Figures 2-7a through 2-7t, as discussed below).

Table 2-3 lists the results for the second round of sampling, which was conducted in November 2014. Percent infection ranged from 8% at Mad Island Reef to 83% at CRD 5 and CRD 6. Weighted prevalence ranged from 0.14 at Mad Island Reef to 1.28 at CRD 6. Infection intensity ranged from 0.76 at Indian Point to 1.67 at Mad Island Reef.

For the stations where a long-term record is not available from Oyster Sentinel, comparisons of fall to summer sampling conditions were made. The fall values from Sammy's Reef were about the same as the summer levels, whereas Indian Point oysters showed the expected cooler weather decrease in disease. CRD 5 showed a decrease in WP as compared to the summer sample, whereas CRD 6 showed an increase. Particularly noteworthy is North Reef, which is commonly uninfected with dermo. It shows a highly elevated level of disease in the fall as compared to the summer survey and to long-term trends. Salinity was high throughout the system in November, and dermo was evident in the upper portion of Lavaca Bay. These elevated salinities may have caused this increasing dermo over the fall months. Infection rates on Oyster Sentinel reefs at some stations have changed as much between monthly samplings. Additionally, North Reef is extensive, thus spatial variability in the dermo data cannot be ruled out as having some influence on these observations as well.

[15] The term 'epizootic' refers to widespread infection associated with significant mortality.

2.4 New Data Acquisition and Processing

In addition to the 2014 field data collection, new data from the same long-term monitoring programs used in the MBHE effort were acquired and analyzed. The reason for updating the oyster suitability criterion model analysis was based on the combination of a multi-year severe drought and the availability of new data to track what occurred during that drought. Data through 2007 were used in the MBHE effort, and both the TPWD coastal fisheries monitoring and Oyster Sentinel program continued after that, although the later program ended regular monitoring in Texas bays in mid-2011.

New data, referring to the years 2008 through 2014 for TPWD oyster dredge data and to the years 2008 through 2011 for Oyster Sentinel data, were acquired and processed following the same procedures described in MBHE (2006c, 2007b) for reefs located in Matagorda, Lavaca, Galveston, and San Antonio Bays. In the MBHE effort, the processed analysis database had monthly average data from 1986 through 2007. Rolling averages were calculated and analyzed for the suitability criterion models for months starting in 1996 (10-year rolling averages of salinity and temperature were considered in the regressions, hence 1996 was the first year of oyster data used). The updated analysis database has the same starting period but now runs through 2014.

The new data represent a substantial increase in quantity over the prior available data, particularly for the Oyster Sentinel data in Matagorda and Lavaca Bays. Dermo monitoring only began in 2003 or 2004 in these bays, and the new data more than doubled the available dermo data in Matagorda and Lavaca Bays relative to what was available for the MBHE effort. The TPWD data collection efforts began much earlier, hence the new data represented approximately a 60% increase (1996 through 2014 versus 1996 through 2007) in the processed analysis database. An additional result of acquiring new data is that North Reef is now included as one of the Oyster Sentinel reef locations. In the MBHE effort, dermo monitoring had just been initiated at North Reef, so there was not sufficient data to include it in the dermo analysis. The inclusion of North Reef brings the number of dermo reefs up to 14 in the three-bay system and up to 6 in Matagorda and Lavaca Bays. Inclusion of North Reef, which is close to the Lavaca River inflow and is therefore relatively fresh, expands the salinity range of reef monitored for dermo in these bays. While North Reef is typically relatively fresh, during extended periods of low inflows, the salinity can increase significantly, even reaching hypersaline conditions (e.g., see summer of 2011 in Figure 2-7i). Such conditions are included in the analyses herein where data are available. Unfortunately, dermo data for North Reef are not available in the summer of 2011, so this hypersaline period is not represented in the analyses herein.

2.5 Data Trends

Table 2-4 shows the data quantity and average values for the 20 reef locations across the three-bay system in the processed analysis database. The averages presented in Table 2-4 are the average of monthly averages (where available) calculated from 1996 through 2014. Data availability at each reef depended on reef size (larger reefs have more sampling grids, and are accordingly sampled more frequently in TPWD's randomized sampling approach) and the duration of the Oyster Sentinel monitoring at the reef. The salinity averages demonstrate that the reefs exhibit a wide range of salinity regimes. Because of the recent drought, these long-term averages are generally 2 ppt higher than were reported in MBHE (2006c). Table 2-4 also lists the ranges in average dermo and oyster counts across reefs. The relationships between the long-term reef averages for oyster counts and dermo versus salinity are examined further in Section 2.6.

Figures 2-7a through 2-7t show the time series of monthly results for oyster counts, dermo WP, temperature, and salinity from 1996 through 2014 for each reef in the three-bay system. These time series plots show all months with available data as points and show connecting lines where data are present in consecutive months. The time series plots include the dermo results collected at Matagorda and Lavaca reefs during the 2014 field sampling efforts described in Section 2.3. Dermo data collected in the CRD for both the 2007 MBHE and 2014 field sampling efforts are shown at Shell Island Reef, which is the closest submerged reef to the CRD fringe reef locations.

The reef time series plots (Figures 2-7a through 2-7t) clearly demonstrate the effect of the recent drought on the salinity at each reef location. Reefs with more exposure to freshwater inflow events have historically (i.e., pre-drought) had more frequent downward spikes in salinity and lower overall salinity. These downward spikes, which demonstrate the occurrence of high freshwater inflow events (freshets), substantially decrease or disappear completely during the drought years of 2008 through 2014, and the overall salinity trend lines are higher at each reef during this period than they were in most years prior to 2008.

The gradient of relatively fresh water to relatively saline water in Matagorda Bay (Shell Island to Mad Island to Sammy's Reefs) and in Lavaca Bay (North to Route 35 Bridge to Indian Point and Gallinipper Reefs) can be observed by comparing the time series plots for these reefs. The salinity gradient and temporal trend in Matagorda Bay is also compared using annual averages at Shell Island, Mad Island, and Sammy's Reefs in Figure 2-8. Figure 2-8 shows that the effect of more severe drought periods is to reverse the typical salinity gradient in Matagorda Bay. During wet periods, Sammy's Reef often exhibits higher salinity than Mad Island Reef and Shell Island Reef. However, during recent dry periods, this pattern reverses, with Sammy's Reef (which is closer to the tidal inlets) actually exhibiting lower salinity and Mad Island Reef and Shell Island Reef exhibiting higher salinity, possibly due to evaporation exceeding the freshening effects of river inflows.

Figure 2-8 illustrates average annual salinities based on TPWD data from Sammy's Reef, Mad Island Reef, and Shell Island Reef. This figure shows that in 3 of the 4 years with the highest salinities, Shell Island, which is closer to the CRD inflow, switches from being the least saline to the most saline of the three EAMB reefs, demonstrating the effect of prolonged low inflows on bay-wide salinity gradients. This trend is of particular concern as reefs closer to freshwater inflows are typically refuges during high salinity/low flow periods.

These data, combined with Figures 1-2 and 1-3a through 1-3i, indicate that the last wet year was 2007 and years 2008 through 2014 all had below-average inflows and relatively high salinities, with 2011 exhibiting the lowest inflows and highest salinities of any year evaluated.

The biological data at each reef (i.e., oyster counts and dermo WP) exhibit a high level of month-to-month variability, likely due to both spatial and population-stage driven heterogeneity, making temporal trends harder to observe. To make broad trends easier to observe, oyster and dermo annual averages for all reefs in Matagorda and Lavaca Bays combined were calculated (Figure 2-9). Several semi-quantitative trends based on

Figures 2-7a through 2-7t, Figure 2-8, and Figure 2-9 are identified here; the most robust statistical trends are identified in Sections 2.6 and 2.8 below.

The following trends are shown in Figures 2-7a through 2-7t, Figure 2-8, and Figure 2-9 regarding dermo WP:

- Dermo is highly variable.
- Some stations generally have low dermo WP (e.g., Shell Island Reef, North Reef). Typically, low dermo WP is correlated with lower salinity; however, low oyster counts may also be a contributing factor to the low observed dermo WP in some reefs, due to reduced transmission of dermo. Oyster-to-oyster transmission via water is the predominant mode of the spread of dermo (Ray 1987), and would be expected to be lower in reefs with low oyster densities.
- Many reefs have relatively low dermo in winter 2008 and winter 2009, which may be a result of significant inflows and reduced salinity in 2007 and/or low temperatures during these winters (low temperatures reduce dermo growth and consequently make dermo cells more difficult to identify using the RFTM approach [TPWD and TWDB 2011]).
- Some stations with generally low dermo exhibit an increase in dermo during the recent drought (e.g., Shell Island Reef). In particular, the November 2014 sample in North Reef was the highest ever recorded at that location. This may be a consequence of consistently high salinity over the antecedent 4 years. However, variability is high, as evidenced by the zero dermo WP measured at this reef in August 2014.
- Even stations with relatively high dermo often exhibit some samples with a dermo WP of zero. This is likely a reflection of antecedent high inflows but also seasonality (winter generally has lower dermo than other seasons) and the possibility of false negatives in the RFTM dermo assay.
- Reefs with relatively high dermo historically don’t show clear evidence of increases during the recent drought (e.g., Sammy’s Reef, Gallinipper Point).
- In the 2014 data, some locations had lower dermo than expected (e.g., Sammy’s Reef, Indian Point, and Gallinipper Point). This may be because oyster counts at these reefs have been low for a few years, inhibiting transmission of dermo.

- Dermo increased in 2010 and 2011 as the drought progressed following the hot and dry year of 2009.

The following trends are shown in Figures 2-7a through 2-7t, Figure 2-8, and Figure 2-9 regarding oyster counts:

- Both total and commercial-sized oyster counts are highly variable.
- Some reefs in Galveston Bay were impacted by Hurricane Ike in September 2008. Accordingly, temporal trends for those reefs are difficult to interpret without additional analysis that was out of the scope of this project. Figures 2-7a through 2-7t include the Galveston Bay reefs results; Figures 2-8 and 2-9 focus only on Matagorda and Lavaca Bays.
- The highest total oyster counts are often seen in 2008 and 2009. This may partly be a result of high inflows in 2007, suggestive of a 1 to 2 year lag between high inflows and increased oyster counts.
- Commercial-sized oysters increased in 2009 and 2010 following the increases in total oyster counts in 2008 and 2009.
- Total oysters decreased from 2009 to 2014 as the drought progressed.
- Commercial-sized oysters decreased from 2010 to 2014 as the drought progressed.

In general, these trends suggest a dependence of dermo WP and oyster counts on antecedent salinity, possibly including a lag of 1 to 3 years. These trends are consistent with the literature and are quantified more robustly in the sections that follow.

2.6 Relationship between Long-Term Oyster and Dermo Reef Averages versus Salinity

The relationships between long-term salinity regimes at each reef location versus oyster counts and dermo were explored using regression analysis. For this long-term analysis, reefs from San Antonio Bay, Matagorda and Lavaca Bays, and Galveston Bay were used. Average statistics for each reef location (Table 2-4), which were calculated from monthly averages (when available) from 1996 through 2014, showed a relatively wide range of salinity, oyster counts, and dermo WP. Dermo is tolerant of a wide salinity range but does best in higher salinities, and reefs with higher average salinity were expected to have higher average dermo

WP. Oysters are also tolerant of varying salinities, but reefs in locations with intermediate salinity values were expected to be ideal for oyster populations, while reefs in fresher or saltier locations may be refuges during low inflow and high inflow periods, respectively, but would not have the largest oyster populations because they spend more time in suboptimal conditions.

2.6.1 Average Dermo and Salinity

Dermo was not measured throughout the entire database period; however, all 13 reefs[16] with dermo data had dermo results from roughly 2004 through 2009. In order to compare dermo to average salinity results across reefs in the same climate conditions, averages were constructed for just the 2004 through 2009 time period for those 13 reefs (Table 2-5). Average dermo WP was then compared to average salinity at each reef and also to average temperature and average oyster counts. Positive relationships between reef averages of dermo versus temperature, total oyster count, and commercial-sized oyster count were noted. All of these would be expected: increased temperature and host availability should benefit dermo, but none of the relationships was strong (regression R^2 value of 0.22, 0.11 and 0.17, respectively). In contrast, the relationship between average salinity and dermo WP was both positive and very strong (the strongest of all relationships tested, with an R^2 of 0.75; Figure 2-10). Residuals (i.e., unexplained variability) from the dermo versus salinity regression had no clear relationships versus average temperature or oyster counts, and these variables were not further investigated as predictors of average dermo WP at reefs.

It was noted qualitatively that residuals around the salinity versus dermo WP regression line appeared to be high for reefs with lower freshet frequency and vice versa. This was formally tested using a measure of long-term freshet frequency at each reef—proportion of monthly

[16] Frenchy's Reef (Galveston Bay) was not used in this analysis because oyster counts dropped to zero or near zero from 2001 to present, and dermo disappeared on this reef, likely due to lack of hosts (Figure 2-7l). Confederate Reef was included in this analysis. Confederate Reef is unique in that Dr. Sammy Ray collected oysters from this reef for Oyster Sentinel, and many of the oysters may have been from inter-tidal locations. The response of dermo to salinity and temperature may be different at this location because inter-tidal oysters are exposed to different physical conditions than the sub-tidal oysters collected at other reefs. To test this possibility, all analyses in this section were performed both with, and without, Confederate Reef. The exclusion of Confederate Reef had a negligible impact on the analyses and no impact on the conclusions; accordingly, only the results with Confederate Reef are presented.

averages with salinity less than or equal to a particular threshold. The first threshold tested was 2 ppt. In general, reefs in Galveston Bay had the lowest proportion of months with salinity less than or equal to 2 ppt, while reefs in San Antonio Bay had the highest. The regression exhibited an inverse relationship—as the proportion of months with less than or equal to 2 ppt increased, average dermo WP decreased (R^2 equaled 0.58; Figure 2-11). The fairly strong regression coefficient—although not as strong as for average salinity—indicated that increasing frequency of freshet events led to lower dermo. Additionally, the residuals of the regression of average salinity versus dermo also had a strong negative relationship with proportion of months less than or equal to 2 ppt (R^2 equaled 0.40), indicating that freshet frequency was not just a proxy for average salinity, but had additional predictive power for dermo across reefs. Other salinity thresholds tested (1, 3, 4, 5, and 10 ppt) had a similar relationship versus residuals, but did not have as strong an effect on the regression as did 2 ppt. The threshold at 2 ppt appears consistent with literature, e.g., La Peyre et al. (2009) tested effects of exposure to 1 and 5 ppt and showed substantially greater dermo reductions in oysters exposed to 1 ppt for 3 weeks as compared to oysters exposed to 5 ppt.

A multiple regression of dermo versus average salinity and proportion of months with salinity less than or equal to 2 had greater predictive power than either variable alone (R^2 equaled 0.88; Figure 2-12); therefore, the combination of average salinity and freshet frequency is an extremely good indicator of typical dermo conditions in reefs throughout the three-bay study area. It is notable that the maximum long-term average dermo WP was 1.7 (at Confederate Reef in Galveston). This is supportive of the general opinion in literature cited in Section 2.1 that dermo WP above 2.0 would result in population die offs. On a short-term basis, values above 2.0 occur in some reefs, including several in Matagorda and Lavaca Bays, but dermo at this level does not persist long term in the bays studied herein.

2.6.2 Average Oysters and Salinity

The average commercial-sized oyster count for the entire database period of 1996 through 2014 was compared to average salinity at each reef for the same period. The commercial-sized oyster count had a non-linear relationship to salinity with the highest oyster counts at intermediate salinities. This relationship was modeled as a two-term polynomial versus salinity (functionally, a multiple regression with salinity and salinity squared), which while not as strong as the relationship between dermo and salinity, was still able to predict 50% of the variability in commercial-sized oysters across reefs (average salinity was the strongest of all relationships tested, with an R^2 equaling 0.50; Figure 2-13). The optimal average salinity for oysters was approximately 20 ppt for the polynomial model. This is somewhat higher than optimal salinities reported in some literature for growth of adults, but it is within the range of other citations and also matches the reported optimal salinity for reproduction (Section 2.1).

In addition to average salinity, the suitability criterion model from the MBHE effort also included a term describing freshet frequency (MBHE 2007b); therefore, the proportion of months with salinity less than or equal to 2 ppt was also tested against average commercial-sized oyster counts. As with average salinity, an intermediate optimum was expected for this variable—some freshet activity will reduce dermo levels on the oysters, but too much will kill the oysters themselves. A two-term polynomial fit to oyster counts versus proportion of months with salinity less than or equal to 2 ppt shows the highest commercial-sized oyster counts occurred at a proportion of approximately 0.055, which is one freshet event in 18 months (R^2 equals 0.26; Figure 2-14).

A multiple regression of commercial oyster count versus average salinity and proportion of months with salinity less than or equal to 2 ppt (four terms: salinity, salinity^2, proportion less than or equal to 2 ppt, and proportion less than or equal to 2 ppt^2) had greater predictive power than either variable alone (R^2 equals 0.66; Figure 2-15); therefore, as with dermo, a representation of salinity regime combining average salinity and freshet frequency was a good indicator of expected oyster counts in reefs throughout the three-bay study area. No other freshet thresholds tested (1, 3, 4, and 5 ppt) had a better fit to the data than 2 ppt.

In summary, this analysis of long-term average data (2004 to 2009 for dermo and 1996 to 2014 for oysters) identified both long-term average salinity and proportion of months with salinity less than or equal to 2 ppt as meaningful predictors of dermo and commercial-sized oyster counts. In both cases, the long-term average salinity had a higher predictive power than the proportion of months less than 2 ppt; also in both cases, both terms were statistically significant in the multiple regression model. This consistency between the two organisms, and between these results and expectations from the literature, is remarkable and increases confidence in the results.

2.7 Matagorda Bay Health Evaluation Suitability Criterion Models Updated With New Data

Several years of severe drought conditions provided ample new data within Matagorda and Lavaca Bays representing prolonged, extremely high salinity conditions. While the MBHE suitability criterion models theoretically should predict oyster and dermo responses to these conditions, the new salinity data went beyond the range of the old salinity data from Matagorda and Lavaca Bays, although not beyond the range of old salinity data from Galveston Bay. Uncertainty as to whether the MBHE models would appropriately represent Matagorda and Lavaca Bays during extreme drought was a primary motivator for the updated analysis undertaken and presented in this report.

The MBHE suitability criterion model for 3MRA DCI (built using data from San Antonio Bay, Matagorda and Lavaca Bays, and Galveston Bay) did a poor job of predicting actual 3MRA DCI for the new data in all three bays (Figure 2-16). Specifically, comparing blue symbols (new data) to black symbols (old data), the actual 3MRA DCI using new data is often higher than predicted (note the blue symbols in the bottom right of the figure)—oysters are in better condition with respect to dermo infection because dermo is not as severe as predicted. To try to determine the cause of the poor fit in the new data and provide guidance for development of an updated model, old and new data were plotted against the individual regression terms in the model. This demonstrated that the dermo response to the 2-year average salinity in the new data deviated from the pattern of the old data—dermo was not as high as expected for the high salinity levels that occurred in the new data (Figure 2-17; note the blue symbols in the upper right of the figure).

The DCI results from the 2014 field data are also plotted on Figure 2-17. These results are not 3MRA (one sampling event occurred in August and September and another occurred in November 2014, so a 3-month average could not be calculated), so they would be expected to have more variability in their response to the prior 2 years' average salinity than 3-month averages. Instead, the field results were invariably higher than the old trend line, indicating the following:

- Dermo WP in Matagorda and Lavaca Bays was lower than expected given the recent salinity history.
- The 2-year salinity average, as constructed in the MBHE effort, was not a good predictor for dermo WP in the new data.

For 3MRA OCI, there is generally more spread when comparing model predicted values to actual values (Figure 2-18), reflecting the lower predictive power of this model. Comparing blue symbols for new data to black symbols for old data, the 3MRA OCI for the new data mostly overlapped the lower part of the old data cloud—oyster condition as indicated by commercial-sized oyster counts was lower in recent years in the three-bay system than in the old data (Figure 2-18). Because the new data fell within the cloud of the old data, there was not an obvious problem with the model, in contrast to the case with 3MRA DCI, but the fit to the new data was relatively poor compared to the old data, indicating an update to the model might prove useful.

2.8 Updated Monthly Model Construction

The failure of the old DCI suitability criterion model to predict dermo for the new, drought period data and the relatively poor fit of the new OCI data to the old OCI suitability criterion model indicated a need for better models. Model updates that included the new data were undertaken with the goal that the new models work equally well for both new and old data periods.

Model reconstruction was taken as an opportunity to narrow the geographic range of the suitability criterion models to just Matagorda and Lavaca Bays for both DCI and OCI. During the MBHE effort there was not enough data in these bays to develop robust models. The salinity data were relatively limited on the high end, and bringing in Galveston Bay allowed incorporation of responses to higher salinity. These responses were not in Matagorda and Lavaca Bays, however, and the salinity regime is generally different in Galveston Bay. For dermo, the lack of salinity range was combined with a shorter time period (regular dermo monitoring began in 2003 and 2004 in Matagorda and Lavaca Bays, respectively), so the dermo data in Matagorda and Lavaca Bays were fairly limited in quantity and represented a relatively narrow range of salinity conditions. Bringing in additional Oyster Sentinel reefs from San Antonio Bay, and particularly Galveston Bay, where dermo monitoring started in 1998, greatly expanded the available data for model development. The current effort includes 7 and 4 additional years of oyster and dermo data, respectively, in Matagorda and Lavaca Bays and substantially expands the salinity range of dermo data within the bays, hence addressing both of the data limitations that motivated the expansion to adjacent bays in the MBHE effort.

The multiple regression results for the MBHE 3MRA DCI suitability criterion model are displayed graphically in Figure 2-19. Matagorda and Lavaca Bays results are displayed with different symbols on Figure 2-19 and show that the MBHE model for 3MRA DCI did a relatively poor job of predicting the limited DCI data in Matagorda and Lavaca Bays. For example, the trend of dermo WP versus two year salinity does not appear as strong for Matagorda and Lavaca Bays as it is for data from the other two bays. Also, the Matagorda and Lavaca data taken alone do not have a positive relationship versus two-year spring temperature average. Because these bays are of primary interest and it appears that dermo responds somewhat differently among the bays, it was appropriate to focus this updated effort solely on Matagorda and Lavaca Bays.

Model reconstruction also provided an opportunity to test three types of additional salinity terms as follows:

- New average salinity terms
- New freshet frequency terms
- New time lagged salinity terms

New average salinity periods of 6 months and 4 years were tested. The 6-month period and 4-year period were tested in response to the finding that oyster and dermo population statistics in the Gulf of Mexico appear to correlate with the 4-year El Niño Southern Oscillation (ENSO) period (Soniat et al. 2005, 2009, and 2012) and that dermo outbreaks were initiated within 6 months of an ENSO event (Soniat 2005). New freshet frequency terms were calculated as the proportion of months with salinity less than or equal to a threshold. This calculation method matched the approach used for the long-term reef average regressions discussed in Section 2.6 and made results less affected by the averaging period than the 5-year and 10-year low salinity flood frequency variables tested in the MBHE effort. For this set of variables, several salinity thresholds (2, 3, and 5 ppt) and several time ranges (2, 3, 5, and 10 years) were tested. In response to apparent time lags of approximately 1 year in observed time trends of dermo and oyster counts in response to salinity events, 1-year and 2-year average salinity terms with 0.5- and 1-year time lags from the present month were also tested. Additional variables tested are described in MBHE (2006c).

New multiple regression models for 3MRA DCI and OCI using only Matagorda and Lavaca Bays data were constructed using the same general approach for term selection as described in MBHE 2006c. These new models evaluated the utility of the new terms described here as well as the existing salinity and temperature terms from the MBHE effort. Calculation of the OCI and DCI values includes a term for MaxOC and MaxWP in the database (Section 2.2.2). With the addition of new years of data, these maxima were increased from 82 to 84 for MaxOC and from 3.25 to 4.0 for MaxWP.

2.9 Updated Model Results

2.9.1 Updated Dermo Model

The new suitability criterion model for 3MRA DCI using only Matagorda and Lavaca Bay data included three multiple regression variables in the following order:

- P ≤ 2 ppt 5YR: Proportion of months with salinity less than or equal to 2 ppt in the prior 5 years
  - Increasing freshet frequency decreases dermo and therefore increases DCI
- 3MRA T lag 1m: 3-month temperature, lag 1 month
  - Increasing temperature increases dermo and therefore decreases DCI
- 2YR S lag 1YR: 2-year average salinity, lag 1 year
  - Increasing salinity increases dermo and therefore decreases DCI

The updated Matagorda and Lavaca Bays-only model (Figure 2-20) explains slightly more variability in 3MRA DCI than the old (MBHE) model: R^2 is now 0.66 as compared to 0.56.

The first term in the model is P ≤ 2 ppt 5YR, which explains 45% of the variance (Partial R^2 of 0.45). It is notable that a very similar term, P ≤ 2 ppt from 2004 to 2009 (6 years), was one of the predictors of dermo WP for the long-term reef average analysis presented in Section 2.6. In the monthly analysis, the term can change at any given reef in response to freshet events in the prior 5 years.

The first term was chosen because it had the highest R^2 versus 3MRA DCI (and a balanced residual pattern). Intuitively, shorter durations than 5 years might be expected to be more predictive, but, for average salinity terms (with and without time lags) and freshet frequency terms, longer time periods were better predictors (higher R^2) of dermo response in Matagorda and Lavaca Bays. The explanation may be that while dermo can grow quickly within an individual oyster, dermo levels take time to respond (e.g., build up in response to favorable conditions for dermo) within the oyster population.

The second and third terms in the model are 3MRA T lag 1m and 2YR S lag 1YR (Figure 2-20; partial R^2 equals 0.11 and 0.09, respectively). Following standard procedures for multiple regression model construction, these two terms were the best fits to the remaining variability in the data, respectively, after the first term had been entered into the model. These are similar to terms in the MBHE model, 2-year rolling salinity average and 3MRA temperature, except with time lags. The 1-month time lag for temperature (3MRA T lag 1m) predicted 3MRA DCI slightly better than the 3MRA T (with no lag), likely because the temperature measured in the same month as the dermo will not yet have impacted the dermo infection levels. The 1-year lag between increased average salinity and subsequent increases in dermo is less intuitive, but matches the qualitative response patterns in dermo levels noted on the reefs.

Figure 2-21 shows that the model effectively captures the range of dermo responses at different reefs. Additionally, actual 3MRA DCI versus model-predicted 3MRA DCI for a relatively fresh (Shell Island) and relatively salty (Indian Point) reef are highlighted in Figure 2-21. The points generally fall around the 1 to 1 line as expected, but 3MRA DCI values from Shell Island vary within the high end of the DCI range, while 3MRA DCI values from Indian Point vary within the low end of the DCI range.

An important goal for the updated DCI model was for the response of new data and old data to be the same. Figure 2-22 demonstrates that this goal was achieved. This figure shows the same data as Figure 2-20 except the response of the older data is highlighted. A comparison of data points from both periods on Figure 2-22 shows that they follow the same patterns versus the three-model regression terms, and trend lines for the whole data period and old data period for each plot lie essentially on top of each other.

While the updated model does not severely underestimate 3MRA DCI during long-term high salinity conditions (underestimates did occur when applying the MBHE model to the new data; Figure 2-18), the model does appear to underestimate 3MRA DCI when P ≤ 2 ppt 5YR equals 0. This pattern, which occurs for both Oyster Sentinel data and the 2014 field data (Figure 2-22, top panel), is observable as data points lying above the trend line at the 0 value on the x-axis. A possible explanation for these data not fitting the overall trend is as follows: if there have been no freshets in the last 5 years, it is likely that the oysters themselves are struggling and have low densities thereby potentially hindering the proliferation of dermo. In fact, a general decline in oyster counts in recent years in Matagorda and Lavaca Bays is discussed in Section 2.5 and shown in Figures 2-7a through 2-7t and 2-9. Thus, while high salinity conditions generally favor dermo, it is possible that lower dermo levels, as a response to a reduced host population, follow prolonged periods with no freshets and consistently high salinity. Complex relationships, such as the one posited here, could be identified and quantified using additional terms in the regression model or more complex statistical approaches, but they were beyond the scope of this effort.

2.9.2 Updated Oyster Model

The 3MRA OCI Model from the MBHE effort was updated using only Matagorda and Lavaca data, including the new data, and applying a standard multiple regression model building approach. The salinity and temperature terms that best explained 3MRA OCI, and were therefore used for the model, were unchanged from the MBHE effort (Section 2.2.3), although the coefficients changed, resulting in an overall better fit for the new data as well as the old data. The final multiple regression model for 3MRA OCI included terms for 10-year low salinity event frequency (calculated as the average number of months between events where the average salinity for a reef location was less than or equal to 2 ppt) and the prior 2 years' salinity average. Both of these terms were entered as 2-term polynomials, indicating that intermediate values in the ranges of average salinity and freshet frequency were optimal for OCI. An additional term in the model was the prior 2 years' average winter temperature (higher winter temperatures presumably increased oyster growth). Also in keeping with MBHE results, this model still explains a relatively small amount of the oyster count variability (R^2 equals 0.33), but it agrees with long-term average regressions (Section 2.6.2; Figure 2-13) in that the optimum salinity is approximately 20 ppt (Figure 2-23).

It is notable that both a freshet frequency variable and a long-term average salinity variable feature in both these 3-month average DCI and OCI models and the long-term average dermo WP and oyster count models (described in Section 2.6). The consistency of these results provides the following benefits:

- They further support that salinity impacts oysters both as an average condition and as an episodic influence.
- They agree with literature findings (e.g., LaPeyre et al. 2009) on the benefits of periodic naturally occurring freshets, especially in high salinity conditions.
- These results support the management of inflow conditions in the context of both average salinity and freshet events, as will be discussed in Section 6.

3 MARSH PRODUCTIVITY EVALUATION

In addition to oysters and dermo, the MBHE effort and associated inflow standards were informed by measurements and analyses related to marsh productivity, including marsh vegetation biomass and juvenile finfish and shellfish densities. In 2014, additional data related to these organisms (marsh vegetation, juvenile finfish, and shellfish) were collected to do the following:

- Evaluate the underlying vegetation relationships used in MBHE
- Test MBHE habitat model predictions that provided guidance in developing freshwater inflow criteria

3.1 2014 Marsh Productivity Field Collection

The marsh vegetation biomass sampling effort focused on low and high estuarine marsh and evaluated aboveground plant biomass for wetland plant species common throughout each delta area. As described in MBHE (2006c), a low estuarine marsh (LEM) is regularly flooded by diurnal tides, while a high estuarine marsh (HEM) is infrequently flooded by high tides and storm events. The LEM is extremely important to aquatic organisms as it provides higher quality aquatic habitat that is more frequently available due to inundation, while HEM can be extremely valuable for waterfowl and shore birds, while also serving as temporary habitat for aquatic species.

The juvenile finfish and shellfish sampling focused on the key MBHE project species (other than oysters) within shallow water habitats. Key MBHE project species expected in the Matagorda and Lavaca Bays with sufficient abundance in marsh habitats include brown shrimp, white shrimp, blue crab, Gulf menhaden, and Atlantic croaker. Due to the life cycles of each of these organisms, some of the key species are more prevalent in the bays at different time periods (Table 3-1; MBHE 2006c). As such, the timing of the 2014 sampling efforts (September and October)—driven by the contract schedule—only allowed for an evaluation of white shrimp based on expected time in bay and blue crab, which fortunately were present with adequate abundance in fall 2014.

Marsh productivity biological sampling (marsh vegetation biomass and throw trap for juvenile finfish and shellfish) was conducted along three transects in both the LRD

(Figure 3-1) and the CRD (Figure 3-2). In the CRD, sampling locations were chosen to match those used in the MBHE studies. Specific methods for sampling for each component are discussed in the following sections.

3.1.1 Marsh Vegetation Biomass Sampling

Marsh vegetation was sampled in October 2014. Each location included a transect extending from the marsh edge (ME; defined in this study as LEM that is less than 5 meters [m] from open water) to the marsh interior (MI), which can either be low or high estuarine marsh depending on elevation but is always greater than 5 m from open water. Transects were generally approximately 100 m in length to the interior marsh community, but varied between sites. The aim was to sample both edge and interior zones during the study. At each transect, duplicate vegetation plots (0.25 square meters [m^2]) were randomly selected in the ME and MI plant communities. All standing plant material within each plot was clipped at ground level and placed in plastic bags. Live and dead plants were separated, and live plants were further separated by species and counted. All harvested plant material was oven dried at 60°C to constant weight. Aboveground biomass (grams per square meter [g/m^2]) was determined for live plant species individually. Figure 3-3 shows the crew conducting the marsh vegetation biomass sampling at a MI site in the CRD.

3.1.2 Throw Trap Juvenile Finfish and Shellfish Sampling

Two throw trap sampling events were performed: one in September 2014 and one in October 2014 (the October event was coincident with vegetation sampling). The throw trap sampler consists of a 1 m by 1 m by 1.5 m aluminum frame with 1 mm square mesh on all four side panels (Figure 3-4) and is designed to trap juvenile finfish and shellfish. The size of the trap and deployment method (hand thrown) typically limits the number of larger, more mobile fish captured. The bottom of the sampler has a thin, 6-inch metal edge along all four sides that cuts into the sediment, holds the sampler steady, and prevents fauna from escaping the enclosure at the sediment surface. Fauna are collected with a 0.9 m^2 dip net that is swept along the sediment surface from one end of the throw trap to the other. A total of ten sweeps were conducted within each throw trap sample, unless an organism was caught in the dip net on the tenth sweep. Then, an additional three sweeps were conducted until no organisms were caught on the last sweep.

During each sampling event, three replicate 1 m^2 throw trap samples were collected in shallow nonvegetated bottom (SNB) habitat and in vegetated ME habitat at each of the three transects (Figures 3-1 and 3-2). This yielded 72 samples total for the 2014 efforts (two habitats at six locations multiplied by three replicate samples, all performed during each of two efforts). Due to increasing water depths (rendering this sampling technique infeasible) further out in the bay, SNB was sampled approximately 3 to 5 m away from edge habitat. By definition (edge), ME habitat was sampled less than 5 m from open water. The same water depth considerations and edge criteria used for this sampling effort were employed during all MBHE studies, allowing equal comparison of datasets. Fauna were collected in 250 to 1,000 milliliter, high density, polyethylene bottles and preserved in 10% formalin solution.

Samples were transported back to the laboratory, identified to species, and the first 30 individuals of each species per sample were measured to the nearest millimeter. Shrimp were measured from the tip of the rostrum to the end of the telson (total length), and carapace width was measured for crabs. Any organisms larger than the bottle were identified, measured, and released in the field. Additionally, information on the wetland vegetation cover present in each throw trap sample was collected at all sites. Finally, water quality parameters (depth, temperature, salinity, pH, conductivity, and DO) were recorded at each throw trap location.

3.2 Matagorda Bay Health Evaluation Habitat Model Overview

To provide context for this study's results and subsequent discussion, this section presents a high level overview of the MBHE habitat model, pertinent underlying biological relationships, and MBHE freshwater inflow criteria characteristics.

The MBHE marsh productivity assessment involved a series of computer programs that comprised the MBHE habitat model, which was subsequently used to evaluate Matagorda Bay habitat conditions and predict potential changes. These predictions informed and influenced the development of the existing freshwater inflow standards for Matagorda Bay. A detailed characterization of the development and subsequent utilization of the MBHE habitat model can be found in MBHE 2006, 2007, and 2008.

As a high-level summary, the main inputs to the habitat model are physical habitat and salinity for each MBHE key species (MBHE 2006c). Specific to marsh vegetation, relative productivity was also evaluated within the physical habitat input file based on each salinity input file. The marsh biomass relationships with salinity are presented in MBHE 2006c, 2007b. For freshwater inflow criteria development, the MBHE team conducted extensive model runs to evaluate the amount of weighted usable area (WUA) for each juvenile organism and marsh vegetation versus a range of freshwater inflow scenarios represented by a change in salinity regime. The WUA versus salinity relationships were plotted graphically for three trophic levels (which encompass the key species and marsh, shellfish, forage fish, and estuarine marsh) within each of the model segments. The MBHE WUA to salinity relationship developed for the CRD is presented in Figure 3-5.

For inflow criteria development, MBHE habitat model results for areas most representative of those likely to be impacted by Colorado basin inflows were used. This included areas that encompassed a wider range of salinity conditions and physical habitats than what is available in the CRD alone. Habitat quality was subsequently ranked by percentage of maximum WUA for each individual organism as follows:

90 to 100%	Selected
75 to 90%	Good
50 to 75%	Fair
25 to 50%	Poor
Less than 25%	Refuge

The 90 to 100% Selected category represents the best habitat conditions (e.g., preferred or optimal). The other four categories, Good, Fair, Poor, and Refuge, are descriptive of their respective habitat conditions. As stated in MBHE 2006c, professional judgment was used to assign these categories. The use of these categories was retained from the definitions determined for the MBHE for consistency between the two studies. For comparison later in this assessment, habitat quality rankings associated with calculated WUA reflecting existing (at the time of MBHE) physical habitat conditions and modeled salinities are presented in Table 3-2 relative to salinity ranges. Table 3-2 was generated based on MBHE habitat model output as described in MBHE 2006c.

Figure 3-6 shows how the salinity values presented in Table 3-2 were subsequently characterized into habitat quality for white shrimp (MBHE 2008a); similar plots for the remaining species are available in MBHE (2008a). It is important to note that refuge conditions for several key species were extrapolated, since the simulation did not include any periods where average salinity (at a model node near the center of each segment) was greater than 30 ppt. The extrapolation is based upon the chemical suitability function for each species (MBHE 2007b), which shows various levels of decreasing trends in suitability for salinity above 25 ppt. Figure 3-6 also demonstrates that the habitat model predicts lower levels of habitat quality for shellfish at low salinities. Although a valid relationship, this result is not applicable to the current assessment.

Several key observations were noted during MBHE habitat modeling, including the following:

- Importance of LEM habitats to shellfish
- Sharp decline in habitat availability for most species (brown shrimp excepted) as conditions shift from estuarine to marine
- Decrease in habitat availability at the salinity extremes

Additional MBHE habitat model analysis (MBHE 2007b) allowed a spatial observation of changes and total WUA for the CRD along with the entire habitat model area (based on salinity conditions within the CRD). Also noteworthy is that condition changes within the CRD are driven primarily by Colorado River flows, while changes in East Matagorda Bay are more controlled by localized freshwater inflow. Therefore, habitat suitability across the entire bay area can vary spatially according to inflow, and different areas can potentially exhibit large differences in suitability during the same time period.

Another component of the analysis was the calculation of WUA for LEM (annually) and each species per each month (during time in bay) for the long-term hydrodynamic/salinity model run (1995 to 2002). A detailed discussion of these results is also presented in MBHE 2007b. Overall, the results followed the same trends as observed in the model runs discussed above. Larger reductions in WUA relative to dryer conditions were observed for blue crab, white shrimp, Atlantic croaker, and Gulf menhaden. In contrast, brown shrimp observed the largest declines in WUA during fresher conditions. A review of the MBHE habitat model output for the long-term validation run by the month assisted the understanding of spatial distribution of habitat under different inflow regimes from 1995 to 2002 and also confirmed the importance of localized freshwater inflow into the Matagorda Bay system.

As described further in Section 3.3, the amount of freshwater inflow into Matagorda Bay experienced during 2014 sampling is characterized as MBHE level 2 according to MBHE criteria. Based on MBHE 2008a, MBHE level 2 is described as follows:

> The goal for MBHE 2 is to sustain conditions of oyster health, benthic condition, marsh productivity, and shellfish and forage fish habitat. During these relatively dry conditions, the mid-bay region would experience lower quality ecological conditions for each trophic level. Depending on inflows from the Lavaca Basin, it is also likely that during these conditions the reefs, benthic habitat, low estuarine marsh, and shellfish and forage fish habitat would be largely reduced further west into the Matagorda Bay system. (p. 3-5)

As shown in Table 3-3 taken from MBHE 2008a, MBHE level 2 is predicted to provide 24 to 26 ppt salinity near the CRD, good marsh condition, and selected to poor shellfish habitat condition depending on the species.

3.3 Results and Discussion

This section focuses on the results of the marsh productivity sampling, which encompasses the marsh vegetation biomass and juvenile finfish and shellfish field sampling components and analysis.

The intent of the 2014 marsh productivity sampling effort was to provide for the collection of habitat utilization and wetland plant productivity data in the CRD and LRD during a period of extended drought and limited freshwater inflow. The overarching objective of this field data collection was to independently evaluate and validate the relationships developed for the key species and habitats used in the MBHE habitat model, which was previously used to assist in the development of freshwater inflow standards for Matagorda Bay. Similar to previous MBHE fieldwork efforts, specific objectives were to 1) quantify juvenile finfish and shellfish densities and community composition within shallow water habitats in CRD and LRD in relation to physical habitat and salinity, and 2) support the evaluation of the role of salinity and inundation relative to marsh vegetation dynamics in these systems.

3.3.1 Marsh Vegetation Biomass Sampling

A total of 24 marsh vegetation samples were harvested (12 in the CRD and 12 in the LRD). All marsh vegetation samples collected at ME or MI locations in the CRD consisted of *Spartina alterniflora* (smooth cordgrass). Within the LRD, all ME samples were comprised of *S. alterniflora*, but MI samples included a mix of *S. alterniflora*, *Batis maritima* (saltwort), *Juncus roemerianus* (black needlerush), and *Symphyotrichum tenuifolium* (perennial saltmarsh aster). Table 3-4 displays the aboveground biomass (g/m^2) measured at each sample plot during the October 2014 sampling event. Marsh vegetation biomass in October 2014 within the LRD is approximately half of that observed in the CRD. Although less in overall biomass, the LRD transects support a more diverse marsh vegetation community. The difference in biomass and diversity between CRD and LRD is largely caused by the elevation along each transect. The LRD transects are steeper, allowing for a more diverse vegetation community nearer to the ME, which is a direct result of shallower inundation depths and reduced inundation frequency. In contrast, the CRD transects are very flat, resulting in a consistent inundation depth and frequency pattern that allows *S. alterniflora* to flourish throughout the marsh. These observations represent delta-specific conditions and are not considered better or worse relative to each other. Both delta systems support productive marsh habitat, resulting in similar species and densities of juvenile finfish and shellfish. With only this 2014 sampling event available for Lavaca Bay (no other vegetation biomass datasets are available for the LRD), the subsequent analysis will focus on CRD marsh vegetation biomass results.

The CRD 2014 end of growing season marsh vegetation biomass data were combined with the 2006 to 2008 MBHE end of growing season data (MBHE 2007b, 2008b) and the 2009 (BIO-WEST 2009) end of season dataset that was collected during a period of limited freshwater inflow and elevated salinities in the CRD. All datasets were collected by BIO-WEST using the same methodologies. These data were then plotted against the freshwater inflow to Matagorda Bay over the corresponding time period (Figure 3-7). In general, observations in 2014 support the underlying relationships used in the MBHE habitat model. For example, Figure 3-8 (MBHE 2006c) shows the MBHE relationship of decreasing productivity in LEM with increasing salinity. However, one must remember that salinity (shown in Figure 3-8) is only one part of the equation, with inundation frequency also playing an important role in the underlying MBHE marsh vegetation to inflow relationships. As points of reference, Figure 3-8 also illustrates the relative productivity as a function of salinity of high interior marsh (HIM) and low interior marsh (LIM); these communities were included in MBHE (2006c) but were not further evaluated in the current study.

An examination of Figure 3-7 shows a corresponding decrease in vegetation biomass with reductions in freshwater inflow when specifically evaluating the 2006, 2007, and 2014 end of growing season results. In each of these 3 years, the level of inflow appears to be a good indicator of the amount of marsh biomass present at the end of those respective growing seasons. For these years, lesser freshwater inflow resulted in lesser end of growing season biomass. The end of growing season data from 2008 and 2009 does not match this trend because the relationship between freshwater inflow and marsh vegetation biomass is more complex than just comparing the inflow at or just prior to any given sampling event. As evident in Figure 3-7 and described further below, antecedent conditions (of up to several years potentially) can play a role in the marsh vegetation biomass observed during an actual sampling event.

For example, Figure 3-9 provides an overview of resulting MBHE criteria levels associated with the same time period of inflows. An examination of Figure 3-9 and Figure 3-7 show that both the 2006 and 2014 end of growing season sampling events reflect growing seasons that experienced MBHE Level 2 conditions. First, it is encouraging that the salinity data collected during the October 2014 survey averaged just under 26 ppt, which accurately represents MBHE level 2 predictions. Second, it is also encouraging that the marsh vegetation biomass documented during two independent MBHE level 2 rated periods—2006 approximately 600 g/m^2) and 2014 (approximately 350 g/m^2)—are relatively similar. One can speculate with some confidence that 2014 is likely lower than 2006 (albeit the same MBHE level) because of the extended period of low inflow that occurred leading up to the 2014 sampling.

Before strong conclusions can be drawn regarding the possibility of quantifying an inflow to marsh biomass relationship, it is important to explain that 2008 (approximately 1,400 g/m^2) was sampled during MBHE level 1, and 2009 (approximately 900 g/m^2) was sampled at the conclusion of MBHE Threshold conditions. In both cases, the amount of biomass exceeded both the 2006 and 2014 data points collected at higher freshwater inflow levels. One can speculate that this is simply a result of the extreme production of marsh vegetation in 2007 coupled with extraordinarily fresh conditions in the marsh sediments as a result of extremely high 2007 inflows, and had 2008 or 2009 inflow levels occurred in 2014 following the extended drought, those biomass numbers would have been considerably lower.

The point to all this is that antecedent conditions (sometimes spanning multiple years) logically appear to play a major role in the quantity of marsh biomass in the CRD. With the existing data, it is currently not possible to quantify the biomass ranges within each MBHE criteria level as they have the potential to be quite large as demonstrated in the previous example. As such, we are not able to formally validate what the MBHE habitat model would have predicted for marsh biomass in October 2014. However, with that said, both the predictions of salinity conditions during October 2014 and resulting decreasing trend in biomass with reductions in freshwater inflow provide support of MBHE methodologies.

Additionally, the marsh vegetation biomass assessment from this study contributes additional support for a multi-level flow criteria and achievement guideline approach to Texas estuaries as is currently in place for both bays. Higher levels of marsh biomass resulting from higher levels of freshwater inflow are important for these bays and requires the following:

- Inflow criteria go beyond only minimal inflow levels that maintain refuge conditions to keep species alive
- Some type of achievement guideline is in place in order to have these periods of higher inflow moving forward

Fortunately, both the Matagorda and Lavaca Bays criteria have these requirements. The range of marsh biomass experienced in the CRD and continued productivity of that system documented in 2014 after several years of intense drought also highlights that higher levels of inflow and resulting high abundances of marsh biomass don't need to happen all the time.

In fact, having a diversity of habitat conditions over time is how nature maintains healthy and productive systems.

For Lavaca Bay, Figure 3-10 shows the 2000 to 2014 inflows and inflow criteria levels. This figure indicates that field sampling in the LRD in 2014 was conducted under subsistence inflow conditions as described in the BBEST report. However, with only 1 data point, it is impossible to make reference to expectations of biomass or relationships to inflow conditions in that bay system. As noted at the start of this section, lower biomass conditions were reported from the LRD than from the CRD in 2014 field efforts. This may hint that the lower inflow levels in the Lavaca Bay system were responsible for this result, but without additional data from this system, that is pure speculation at this time.

All marsh vegetation results data are provided in Appendix C. Water quality data for the ME sites were collected as part of the throw trap sampling effort and can be found in Appendix B.

3.3.2 Throw Trap Juvenile Finfish and Shellfish

Two throw trap sampling events took place during 2014: one in early September and one in late October. Throw trap sampling in 2014 resulted in the capture of over 5,100 individuals representing 33 species of fish and invertebrates. Table 3-5 shows the average water quality conditions measured at the time of each sampling event in each delta. Table 3-6 shows the overall number of individuals and species collected in 2014 from both the LRD and CRD. A total of 2,064 individuals and 21 species were collected at CRD ME habitat compared to 836 individuals and 19 species collected at CRD SNB habitat. Similar results were found at the LRD sample locations. A total of 1,775 individuals and 20 species were collected in LRD ME habitat and 486 individuals and 16 species were collected in LRD SNB habitat. Based on the individual organism totals at each site for each collection effort, the three most abundant species among all locations are grass shrimp (*Palaemontes vulgaris*), followed by darter goby (*Gobionellus boleosoma*) and then white shrimp (*Litopenaeus setiferus*; Table 3-6). Species diversity and density are consistently higher in ME habitat than in SNB habitat. Results also showed a higher number of total individuals and number of species during the October sampling versus the September sampling (Appendix D). Overall, the species diversity and abundance of juvenile finfish and shellfish species collected in both the CRD and LRD in 2014 were similar to results reported from fall samples taken in Matagorda Bay during MBHE studies in 2006 through 2008 (MBHE 2006c, 2007b, and 2008b). Site-specific water quality data, juvenile finfish and shellfish species information, and vegetation cover data are presented in Appendices B, D, and E, respectively.

3.3.2.1 Matagorda Bay Health Evaluation Key Species

As previously discussed, the timing of the 2014 sampling efforts (September and October), driven by the contract schedule, only allowed for an evaluation of 2 of the 5 MBHE key marsh juvenile species (white shrimp and blue crab). The first objective for this data (including both the LRD and CRD results) was to test the underlying MBHE habitat model relationships relative to the habitat suitability of LEM versus shallow nonvegetated bottom. Figure 3-11 shows the relationship for the combined National Marine Fisheries Service (NMFS) database (MBHE 2007b), each MBHE sample year, and this 2014 study. Overall, the observation in 2014 holds well with previous years' data and supports the importance of ME habitat in the MBHE habitat model. In each data set (including 2014) more juvenile finfish and shellfish were collected in ME (defined as low estuarine marsh within 5 m of open water) habitat than in SNB (e.g. open water) areas.

Figures 3-12 and 3-13 show the white shrimp and blue crab densities plotted against the freshwater inflow to Matagorda Bay over the corresponding time period for CRD. As for marsh vegetation biomass, with only this 2014 sampling event available for Lavaca Bay (no other throw trap datasets are available for the LRD), the subsequent analysis will focus on CRD throw trap key species and community results.

Upon examination of Figures 3-12 and 3-13, there do not appear to be any discernible trends with decreasing freshwater inflow and reduced densities of white shrimp and/or blue crab. This interesting result stimulates some key additional questions, including:

1. Is density of mobile organisms really the best indicator to evaluate estuarine condition?
2. Is the MBHE level 2 criteria for white shrimp and blue crab overly protective in that conditions reported for these species in 2014 (characterized as average densities) are

higher than the poor to average habitat conditions the MBHE habitat model predicted for this corresponding criteria level?

3. Could the reduction in the amount (biomass) of marsh vegetation and likely reduction in spatial expansion (although not mapped for this study) actually have caused a condition of clumping of organisms in remaining quality habitat?

In the case of question 1, the fact that physical habitat (e.g., marsh vegetation) does show a negative response with decreasing inflows suggests that maybe "habitat" is a more appropriate indicator than mobile organism abundance or density. Relative to question 2, the MBHE criteria levels were based on weighted usable area calculations, not organism density. As such, the conditions presented in MBHE 2008a (e.g., selected, good, average, poor, and refuge) represent a combination of both physical habitat and salinity and include a spatial consideration. With this understanding, it is difficult to conclude from the 2014 sampling that the MBHE level 2 criteria are overly protective for these two species. Finally, if the answer to question 3 is yes, then this analysis strongly supports the concept of needing criteria that support refugia conditions in these deltas. Nature has repeatedly shown that habitat can only be reduced so much before the clumping effect hits its threshold and a rapid organism decline ensues (Gilpin and Soule 1986; Lacy 2000; Kaczensky et al. 2011).

All three are important questions worthy of consideration. However, none of them can be quantitatively answered within the confines of this scope of work. Additional years of sampling coupled with a re-evaluation of the existing NMFS original database for this potential clumping effect might solicit answers to these fascinating questions. At this time, all that can be surmised by this result is that the MBHE level 2 conditions (including antecedent conditions) experienced in and leading up to late summer and fall 2014 were protective of these two species.

3.3.2.2 Throw Trap Community Analysis

In order to delve into the database beyond just the two MBHE key species, canonical correspondence analysis (CCA; ter Braak 1986) was used to examine patterns in juvenile finfish and shellfish communities across throw trap samples from 2006 to 2008 and 2014 as well as evaluate their relationship to environmental variables. The CCA is a method of direct gradient analysis that uses environmental variables as constraints on the correlation of sample and species scores. This method allows for testing of the significance of model solutions and effects of environmental factors through the use of permutation tests (iteratively randomizing the data). Non-metric multidimensional scaling (NMDS; Kruskal 1964; Prentice 1977) was also used to evaluate relationships in the community data. NMDS is a method of indirect gradient analysis that utilizes only the differences in species by sample (a distance matrix) and is not reliant on assumptions of linearity. Like most multivariate ordination methods, these methods seek to map the data such that correlation in mathematical distance and plot distance is maximized, producing a solution in which existing correlations are readily observed. Thus, NMDS allows similarities or groups to be defined by the species alone, while CCA allows for investigation of correlations to pre-defined factors of interest (such as environmental variables).

Data preparation for CCA and NMDS is as follows. Because 2014 sampling occurred during the fall season, only fall throw trap (juvenile finfish and shellfish) samples from 2006 to 2008 were used under the assumption of strong seasonal patterns, which is supported by a plethora of coastal research (Tolan 2013). Species that were observed less than 5 times across sample periods were excluded from analysis, reducing the number of species from 33 to 21. Species counts were averaged across replicates at each transect/site/sample event to produce an average abundance for each site at each sampling event.

Environmental variables included salinity (ppt), DO (mg/L), temperature (°C), percent vegetative cover (*S. alterniflora*), vegetative biomass (g/m^2), and open versus ME habitat. Permutation tests were used to evaluate the significance of CCA solutions and the importance of environmental variables. No significant relationships were found in the full model (p equals 0.56) or any CCA other solution (Figure 3-14), and no apparent meaningful grouping or alignment of the species or sights along the environmental gradients or CCA axes was observed. The first (“x”) axis in Figure 3-14 represents the alignment of the species and sample scores that has the highest correlation between the two. The 2d (“y”) axis does the same, but without being correlated to the first. If relationships or structure existed in the distribution of these species and samples, then groups of data would be observed in an ordination (CCA or NMDS). For instance, if some species only occurred at low temperatures and the others at higher temperatures, then two distinct groups of data (corresponding to groups of species) would be identifiable in an ordination plot. Likewise, no discernible pattern was observed in community composition among sites and samples using NMDS analysis (Figure 3-15). It is noticeable in the ordination plot from the NMDS that the species and sample scores are not distributed in any particular pattern—since this is an unconstrained analysis, it further demonstrates that these data lack any strong structure.

Additional analyses were then conducted on a reduced dataset using only fall throw trap species for which greater than 100 specimens were observed over all samples, resulting in selection of five species (bay anchovy, blue crab, darter goby, grass shrimp, and white shrimp). A significant CCA solution was found when percent vegetation cover was included as a covariate of the reduced throw trap sample community (p equals 0.038; Figure 3-16). In the plot of this CCA, samples and species (albeit to a lesser extent) can be seen to be distributed along the vegetative cover gradient.

In an additional analysis, differences in dominant vegetation (*S. alterniflora*) biomass (SPAL; g/.25 m^2) in LEM samples were log transformed and compared among sites, sampling events, and salinity values using analysis of covariance (ANCOVA; Huitema 1980). ANCOVA is an extension of the familiar and commonly used analysis of variance (ANOVA) method that allows for the incorporation of a continuous variable (salinity) along with categorical variables (site, sampling event). When all sites were considered, significant differences in SPAL were observed among sites and sampling trips (p less than 0.001). SPAL was not observed to vary with salinity (p equals 0.55); however, the interaction term for salinity and site was significant, suggesting that the effect of salinity was not consistent. When only the CRD samples were considered, as they were the most consistently sampled over the long term, SPAL was found to differ by sampling event (p less than 0.001) and to decrease with increasing salinity (p equals 0.038; Table 3-7).

The MBHE habitat model predicts that under high salinity conditions, the key throw trap species habitat condition (weighted usable area) will be less than under more average salinity conditions. Nothing in this preliminary throw trap community analysis suggests that it is possible to quantify a community health index or condition directly related to salinity or other water quality parameters. Instead, this community analysis independently supports that the throw trap community is influenced by the presence and abundance of vegetative cover. Furthermore, it provides statistical evidence that vegetative cover is influenced by salinity and thus could affect the structure of the throw trap juvenile finfish and shellfish community. The lack of a strong statistical association between juvenile finfish and shellfish with freshwater inflow is not surprising when considering recent work published for the Texas coast (Tolan 2013; Tolan and Nelson 2013).

4 *RANGIA* INVESTIGATION

Rangia cuneata (*Rangia*, a species of bivalve clam native to Texas estuaries) were not explicitly considered by either the MBHE team or the BBEST and BBASC in their deliberations regarding freshwater inflows. However, other basins (e.g., Guadalupe-San Antonio) have used *Rangia* as an indicator species with which to help identify suitable freshwater inflow standards.

The study plan for this effort included the collection of live *Rangia* and analysis of growth rings in the laboratory by Dr. Bryan Black of The University of Texas Marine Science Institute (UTMSI). As described below, live *Rangia* were not found in Matagorda and Lavaca Bays; accordingly, the growth ring analysis was abandoned.[17]

The remainder of this section describes the field efforts that were implemented in an attempt to find live *Rangia* in Matagorda and Lavaca Bays.

4.1 Limited *Rangia* Surveys

In conjunction with marsh sampling efforts conducted in late 2014, BIO-WEST investigated areas believed to support *Rangia* clams in both the Matagorda and Lavaca Bay systems. Each of these two systems present their own unique challenges for benthic sampling, including water depths and obstructions. Low water depths make it difficult to get a boat into some locations, while obstructions wreak havoc on a tow dredge sampler. Adaptive techniques commensurate with the existing project budget were implemented for each area as described below.

The Colorado River estuary is unique in that it is bisected by the GIWW before entering into Matagorda Bay. North of the GIWW, the lower Colorado River possesses a deep channel for approximately 15 miles upstream to allow barge traffic to the Celanese Chemical Company facility. The lower section of the river is shunted across the GIWW via a hydraulic lock system through a man-made diversion channel into a deltaic formation in the bay. Along

[17] A similar study was implemented in San Antonio Bay where live *Rangia* were found; results of the growth ring study for that system will be delivered to the TWDB in fall 2015 under a separate study.

this diversion channel, large amounts of debris descending from the watershed accumulate, creating an abundance of snags and log piles.

Therefore, as a result of these obstacles, short, concise dredge tows were conducted using a modified oyster dredge in locations along the lower Colorado River, the diversion channel, GIWW, and Culver Cut (Figure 4-1).

Rangia investigations occurred within the Matagorda Bay system on December 4, 2014. Dredge tows were conducted utilizing the BIO-WEST custom fabricated oyster dredge. The dredge was towed in areas suspected to contain *Rangia* shell. Areas selected for limited surveys were based on local fishermen guidance as well as areas upstream in the river deltas with lower salinities. Fifteen recorded dredge tows were performed, and many more were aborted due to the abundance of snags within the bay system. Efforts yielded only a single, old *Rangia* valve (lacking a periostracum layer) that was recovered north of the GIWW in the Colorado River.

Due to the vast extent of the Lavaca Bay system, shallower waters, and diminishing results from the Matagorda Bay efforts, a more efficient effort was implemented to cover a broader area. Repetitive soundings or poling using a PVC pole with an end cap to probe the substrate is a common technique for locating bivalve colonies. BIO-WEST has successfully implemented this technique on past projects to locate *Rangia* in similar systems, and follow-up dredge tows were used to verify sounding results. Using this poling technique, a large area was investigated in the lower Lavaca River, Redfish Lake, Swan Lake, and near-shore areas along Lavaca Bay (Figure 4-2). These efforts did not identify the presence of any *Rangia*. Due to a lack of *Rangia* present in these two bay systems, this supplemental sampling component was abandoned.

5 HYDRODYNAMIC AND SALINITY MODELING

This section describes the development of regression equations that relate inflows to salinity. Inflow-salinity regression equations are necessary to determine what flow is required to achieve the desired salinity criteria that have been determined from the oyster, dermo, marsh vegetation, and juvenile finfish and shellfish assessments. Both the MBHE study, which developed the flow standards for the Colorado River inflow to the EAMB, and the BBEST analysis, which developed the flow standards for the Lavaca River and Garcitas Creek inflow to Lavaca Bay, used regression equations to estimate flows needed to meet specific salinity criteria at specific locations. In each of these studies, salinity was derived from a hydrodynamic circulation model. The primary objective of the current analysis is to update the period of record for the hydrodynamic salinity model and the regression equations developed from these model outputs. For the Colorado River, this involves shifting to a different hydrodynamic model (from RMA, which was used in the MBHE study, to TxBLEND, which is maintained and updated at the TWDB). A secondary objective is to make consistent, where possible, the methods that are applied to both bays.

The development of inflow-salinity regressions involves the following steps:

1. Updating and executing a hydrodynamic salinity model
2. Extracting the appropriate salinity results
3. Reviewing the freshwater inflow time series and separating low flow periods from normal periods
4. Calculating regression parameters appropriate for low and normal conditions
5. Applying these regression equations to determine the flows required to produce specific salinity criteria

At each of these steps, various technical decisions are made related to model selection, time step, period of record, characterization of salinity both temporally and spatially, and the form that the final regression will take. Whenever possible, this study has elected to maintain consistency with the approach taken in the MBHE study and applied the methods from that study to the Lavaca Bay analysis. Table 5-1 describes the decision points used in the previous analyses and in this study. The cells shaded in grey highlight areas where the analysis in this study differ from what was used previously; each of these items is described in more detail in the sections that follow.

5.1 Hydrodynamic Model

This study uses the TxBLEND computer model developed by the TWDB to simulate water circulation and calculate salinity conditions in Texas estuaries (Matsumoto 1992). TxBLEND is a finite element model that employs triangular elements and simulates hydrodynamics and transport in two dimensions (circulation and salinity distributions of vertical-mean parameters in the horizontal plane). Water circulation is simulated by solving the continuity equations and the momentum equation, jointly referred to as the shallow water equations. Salinity condition is calculated by solving the mass transport equation or the convective-diffusion equations. The TxBLEND model has been applied to all of the major bays in Texas as part of the state's freshwater inflow needs evaluation program (Longley 1994; LCRA et al. 2006) and has been used by most of the SB3 BBEST groups in the development of their recommendations (Trinity-San Jacinto, Colorado-Lavaca, Guadalupe-San Antonio, and Nueces). TWDB has documented the model calibration and validation for its application in Matagorda and Lavaca Bays (Schoenbaechler et al. 2011).

The original MBHE study used the RMA model family (a family of finite element models supported by the U.S. Army Corps of Engineers [USACE]) to perform hydrodynamic-salinity transport modeling for an 8.5-year period of record (July 1995 through December 2003; MBHE 2008). The RMA was selected by the MBHE team because it is capable of simulating the wetting and drying that occurs in marsh areas. As it is also part of larger family of models, it could theoretically be linked to these in the future to assess other estuarine properties. While the marsh response was, and still is, believed to be a be an important consideration for overall estuarine health, by the completion of the MBHE study, the technical modeling challenges associated with executing the marsh models made them impractical for the purpose of including marsh salinity responses to flow in the development of the final inflow recommendations. A 30-year TxBLEND simulation is very stable (i.e., it reliably runs to completion) and executes in 20 to 30 hours. More importantly, unlike the TxBLEND model, which is maintained and updated by the TWDB, the RMA model is limited to the 8.5-year period that was developed in the MBHE study. The time and resources that would be required to update and execute RMA are not justified within the confines of this study.

Before finalizing the decision to shift from the RMA model to TxBLEND, the first step was to verify that the results produced by TxBLEND were comparable to those produced by RMA. To verify this, the process to develop regression equations that was used in the MBHE study was replicated with TxBLEND consistent with all of the decisions included in Table 5-1 under the column MBHE (Colorado). The results of this exercise indicated that salinities predicted in the EAMB were not appreciably different, and the flows that would have been determined had the MBHE team elected to use TxBLEND instead of RMA would have been very similar. Figure 5-1 shows a comparison of the regression lines derived from the TxBLEND versus the curve derived from the RMA (MBHE 2008a) for the Delta Edge Transect. The final inflow results that would have been calculated had the TxBLEND model been employed in the earlier study are discussed in Section 5.5 and Tables 5-9 and 5-10.

The application of the TxBLEND model to Matagorda and Lavaca Bays is based on a model with 8,340 nodes, which are three points of the triangular element, to construct a mesh consisting of 13,389 elements (Figure 5-2). The model predicts salinity at each of nodes on a 30-minute time step. These results are summarized as daily average values for use in this study.

The input data to the model consist of two parts as follows:

- The first part includes the static physical properties, including estuarine bathymetry, and transport parameters such as dispersion and roughness coefficients.
- The second part includes the dynamic data (which changes with time), including river inflows, tides, wind, evaporation, and precipitation.

The model is routinely updated by the TWDB to included recent hydrologic and metrological data; prior to this study, the data had been updated through 2009. For this study, the model includes a period of record from November 1986 to December 2014 (although the most recent years contain some provisional data as will be discussed below). Metrological data, including winds, tides, evaporation, precipitation, and off-shore salinity at the gulf boundary, were updated by TWDB staff following the approach documented in Schoenbaechler et al. (2011).

Inflows to the bays are represented at 15 point locations representing the primary subwatersheds draining to the bays (Figure 5-3). River inflow data were provided by the TWDB (disaggregated by subwatershed) as the following:

- Gaged flow (g)
- Ungaged flow (modeled using the TWDB TxRR rainfall runoff model; m)
- Diversions (d)
- Return flows (r)

TxBLEND surface water inputs for each of the 15 rivers and streams were calculated by the following equation.

$$Inflow = g + m - d + r$$

The data from the gaged and ungaged sources are complete through 2014, the diversion data from TCEQ are only available through 2013, and the return flow data are only available through 2012.

While there were several apparent inconsistences within the TxBLEND input files (discussed below in Section 5-6), verifying and correcting the data was beyond the scope of this project; therefore, the TxBLEND input files were for the most part not adjusted. However, a review of the various data subsets raised some questions, and, in a few cases, resulted in modifications to some of the values provided by the TWDB. This report assumes that the existing datasets, upon which the model was calibrated and validated, are generally correct; where significant anomalies were identified, adjustments have been made to the updated data input files in order to maintain consistency with early input files. For example, return flow estimates from the Lavaca Delta input point for the period from 2010 to 2012 were reported as two orders of magnitude higher than the return flows in the previous 20 years. A review of some of the discharge records used to develop these return flow estimates suggests that some of these reported return flows may include stormwater, which is already included in the modeled inflow from the rainfall runoff model. Therefore, these return flow estimates were excluded from the calculation of total inflow. If, however, the recent estimates of return flows prove to be accurate, this would raise questions as to the accuracy of the return flow estimates in the existing dataset (1986 to 2009) and might suggest the need to recalibrate the model and thus could change the design flow estimates.

Although relatively minor, the USGS gage data provided by the TWDB appear to have included provisional data for the last several months in 2009. These values have been replaced with the final flow records from USGS. Particular attention was also given to the data used to estimate inflows to the LRD and CRD. The Lavaca-Navidad River Authority (LNRA) provided a time series of reservoir releases from Lake Texana, which were compared with the values provided by TWDB, and some adjustments were made to the LNRA data based on the TWDB data. Similarly, a time series of diversions from the Colorado River for the STP was provided by LCRA, and these data replaced diversion data provided by TWDB.

In the late 1980s and early 1990s, the Colorado River was reconfigured to allow more water into the EAMB. The bathymetry data for TxBLEND are based on the reconfiguration and are consistent with the current status of the system. Inflows from the Colorado River prior to the reconfiguration (pre-1988) were adjusted to account for the fact that a portion of those inflows were not entering the EAMB, but this adjustment is not necessary for the recent hydrology update (2009 to 2014).

Finally, there appear to have been rather significant increases in the return flows reported between 2009 (existing dataset) and 2010 (new dataset) for the LRD and Guadalupe River input points (Table 5-2). Since these return flows are used for new updated inflows, the project team has decided, in consultation with the TWDB coastal hydrologist, that, in order to be more consistent with existing inflow sets, the return flow estimates of these two nodes would be ignored. In the LRD watershed, return flows have typically been very small relative to other inputs. The Guadalupe River has generally included higher return flow amounts; however, this input node is reasonably distant from the areas of concern in this study (see next section); therefore, their exclusion would not be expected to impact salinities in the LRD or CRD.

5.2 Design Area

This study focuses on the areas of Matagorda and Lavaca Bays that are most directly influenced by inflows from their respective watersheds. For Matagorda Bay, this design area was identified as the EAMB and is specifically represented by average salinities along three transects identified as Delta, Shell Island/Tripod,[18] and Mad Island (MBHE 2008a). Figure 5-4 depicts the nodes along the transects (from shore to shore) at which daily salinities were extracted from the TxBLEND model results. Daily modeled salinities from these nodes were averaged along each transect to produce a time series of daily average salinities. While there is some variation in salinity across the transects (on average, 0.5 ppt to 1.25 ppt, depending on transect), the objective of the analysis is to determine the flows that will maintain average salinity, on the upstream side of each transect, below the design criteria salinity that has been specified for each flow level. The use of average salinity across the transect meets that objective.

For the Lavaca Bay analysis, the BBEST used monthly average salinities at individual reef nodes (Figure 5-5), rather than the daily average salinities across transects, in the development of salinity regressions. To make the Lavaca Bay analysis more consistent with what was done with Matagorda Bay, including the development of regression equations based on daily average salinity across transects, the same transect approach was applied to Lavaca Bay. The Route 35, Gallinipper, and Indian Point transects correspond closely with reef nodes used by the BBEST; the North Reef transect is an additional transect corresponding to oyster and dermo analyses described herein.

5.3 Salinity Inflow Regressions

5.3.1 Salinity Time Series

Daily time series of salinity for selected nodes within the TxBLEND model domain were extracted from the model results. For each transect identified in Figures 5-4 and 5-5, salinity was averaged across each transect to produce a single time series of daily average salinity (Figures 5-6 and 5-7).

[18] The Shell Island/Tripod transect is conterminous with Shell Island Reef and includes the Shell Marker B (SMB) and West Bay Tripod (WBT) datasondes operated by LCRA.

5.3.2 Inflow Time Series

To obtain inflows for the regression of inflows versus salinity, MBHE 2008a used stream flow from the USGS gage at Bay City, adjusted by downstream withdrawal for the STP. To conform to the location at which TCEQ evaluates freshwater inflow standards, for the analysis in this report, the inflow variable was adjusted slightly from that used in MBHE (2008a). Instead of the Bay City gage minus STP, this study uses the estimate of total inflow from the Colorado River that was developed for the TxBLEND model. The only difference in these two values is that the TxBLEND number includes an estimate of ungaged runoff downstream of the Bay City gage. This value is small compared to the gaged inflow minus STP, so the change has a negligible effect on the regressions developed in the next section. This change was made to be consistent with the way freshwater inflow standards are defined in TCEQ rules and interpreted by the TCEQ when they are included in the Water Availability Model (WAM). In the WAM, the freshwater inflow targets are compared to the total inflow from the Colorado River (WAM control point M10000; TCEQ 2001), including ungaged runoff downstream of the gage. In the MBHE study, the Bay City gage minus STP made sense because that study was solely focused on a water development project associated with the LCRA, whereas the total freshwater inflow standard defined in TCEQ rules applies to all entities seeking water rights permits.

To be consistent with the regression approach developed for the MBHE, it is necessary to classify the time series of inflows into low and normal conditions. This classification was accomplished by determining the average monthly flow over the period of record and then parsing out low periods as those in which the flow remained below the average for 20 or more consecutive months in the previous (MBHE) analysis. This length was based on professional judgment and appeared to reasonably separate low and normal conditions. In this study, the daily inflow period of record available for TxBLEND is from 1977 to 2014, which included 1977 to 2009 plus the update performed as part of this study. With the much longer period of record available in this study, including flows from the recent drought, the long term average monthly flow changed from 171,600 acre-feet (ac-ft) per month for the 8.5-year period available in the MBHE to 143,191 ac-ft per month in the 37-year period used in this study. Applying the same 20-month consecutive standard produced the illogical result that the droughts beginning in about 2008 through 2014 were classified as normal periods. This is because within that time frame there were several months in which inflows exceeded the new average of 143,191 ac-ft. In this study with a longer period of record, 14 (or more) consecutive months below the average monthly flow is a better reflection of low versus normal conditions. The 14-month duration generally classifies the 8.5-year period used in the MBHE analysis in the same categories as in the MBHE analysis, while also classifying much of the recent drought as a low flow period (Figure 5-8).

For Lavaca Bay, similar issues related to the appropriate calculation of inflows used to develop regression equations are encountered. The BBEST report (2011) states that "Releases from the [Texana] reservoir were summed with flows from USGS gage Lavaca River at Edna (08164000), and USGS gage Garcitas Creek at Inez (08164600) for purposes of this inflow analysis." A careful review of these three inflow sets suggest that the values represented as the USGS Garcitas Creek at Inez may instead have been the ungaged runoff predicted by the TxRR model for the watershed below this gage. The project team has decided that to be consistent with TCEQ freshwater inflow standards, the appropriate inflows are the total inflows from Garcitas Creek and the Lavaca River as calculated by TWDB to produce the inputs for the TxBLEND model.

When applying the same approach for the Lavaca River and Garcitas Creek inflows, a period of 6 (or more) consecutive months with flows less than the average appears to effectively partition low and normal inflow periods. Figure 5-8 and 5-9 depict time series of total inflows from the Colorado River and Lavaca/Garcitas, respectively, in which the background shading distinguishes low and normal inflow periods.

5.3.3 Regressions

Regression functions were developed for each transect (three in the EAMB versus total inflow from the Colorado River and four in Lavaca Bay versus total combined inflow from the Lavaca River and Garcitas Creek) for all data (normal) and low flow data only (described above). These regression equations take the following form where S is the daily average salinity across the transect, and Q is the total inflow volume in the antecedent 30 days:

$$S = a * Ln(Q_{30}) + b$$

As noted in MBHE 2008a, “The choice to characterize flow as cumulative antecedent 30-day volume represents a compromise.” While there have been times when the salinity response to flow was longer or shorter and other times when the contribution from the other watershed had an effect on salinity, this study did not attempt to analyze alternative formulations, in large part due to the following:

- Because the current inflow standards are based on monthly and seasonal inflows
- Increased computational complexity (e.g., a 30- to 60-day antecedent inflow term) would necessitate substantial revisions and added complexity in the standards themselves

Figures 5-10 and 5-11 show regression equations based on all data and low flow only data for the Colorado River (Delta Edge transect) and Lavaca/Garcitas (Indian Point transect), respectively.

Since the salinity response to inflow is slightly different during low flow times than it is during normal conditions, the next step is to combine the regressions developed for normal conditions and for low flow conditions. Following the approach taken in the MBHE study, blended regression curves for the two design areas are presented in Figures 5-12 and 5-13.

Tables 5-3 and 5-4 are tabular summaries of the flows that relate to certain salinity values based on the blended regression equations. Shaded cells are based on an interpolation between the low flow and normal flow regressions.

5.4 Design Criteria Inflows

Should the project team or the BBASC determine that salinity criteria should be adjusted based on recent data collections and analysis, the blended regression curves can be utilized to estimate the flow needed to produce that salinity. Assuming that the salinity criteria are unchanged, the design flow estimated to meet existing salinity requirements from the Colorado River are based on producing salinities of 31, 27, 24, 20, and 15 ppt for Threshold and Levels 1 to 4, respectively, at the Delta Edge transect and then confirming that those flows produce acceptable salinities at the other two transects. Based on the updated inflow-salinity regression equations, these flow values are presented in Table 5-5.

The flows estimated to meet the existing salinity requirements from the Lavaca River and Garcitas Creek are based on producing salinities of 30, 25, 22, and 20 ppt for subsistence, base dry, base average, and base wet, respectively, at the Indian Point (Middle Ground Reef) transect and then confirming that those flows produce acceptable salinities at the other two transects. Based on the updated inflow-salinity regression equations, these flow values are presented in Table 5-6.

The final step in this process is to convert the design flows identified above into seasonal values. This is accomplished in the same manner as was implemented in MBHE (2008a): by converting the design flow into an annual value by multiplying it by 12 and then distributing this annual value across seasons based on historical hydrologic patterns.[19] After extensive evaluations, the MBHE team based the seasonal distribution for development of the inflow criteria for the Colorado River on 38% of the annualized recommended criteria in the spring period, 27% in the fall period, and the remaining 35% in the intervening 6 months. Appling these percentages to the updated design criteria flows results in the annual and seasonal values shown in Table 5-7.

The BBEST repeated this process for the Lavaca system and assigned 45% of the annualized recommended criteria in the spring period, 32% in the fall period, and the remaining 23% in the intervening 6 months. Appling these percentages to the updated design criteria flows for the Lavaca River and Garcitas Creek results in the annual and seasonal values shown in Table 5-8.

Note that the seasonal distributions were not recomputed in the current study. The original MBHE and BBEST estimates were based on long records, and the addition of a few years of new data were not expected to meaningfully change the percentages.

[19] In MBHE (2008a), this seasonal distribution of flows was used to specify spring and fall freshets with a duration of three months each. In the current study, the term freshet is generally used for short-term inflow events, because the dermo results and associated literature highlight the importance of shorter, more intense (i.e., lower salinity) freshet events.

5.5 Comparison with Previous Studies

The decision to change the underlining hydrodynamic salinity model for the Colorado River and to change the development of regression equations for the Lavaca River and Garcitas Creek does result in changes to the estimates of inflow that would be needed to meet the salinity design criteria in each of these systems.

For the Colorado River, this update involved changing the underling hydrodynamic model from RMA to TxBLEND and extending the period of record from 8 years and 6 months to 29 years and 2 months. The effects of these two changes can be observed in Table 5-9 (the middle table “TxBLEND 8.5 year” uses an identical period of record to the top table “MBHE Study”).

As noted in Section 5.1, the salinities predicted by the two models were very similar; however, as can also be seen in Figure 5-1, there is significant scatter around the regression lines that are generated from the hydrodynamic model results. Once these regressions are blended into a single curve from which the design criteria flows are calculated, the results show a small difference—in this case, a decrease in the amount of inflow that would be need to produce the design criteria salinities. Inclusion of additional years of data, notably lower flow data especially during recent droughts, produces results that indicate slightly less water is needed to produce these design criteria salinities.

In Lavaca Bay, this study includes three changes in the development of inflow-salinity regressions as follows:

- Change 1: A change in the characterization of inflow from Lake Texana releases plus the gage flows from Lavaca River at Edna and Garcitas Creek at Inez (also noting that there may have been an error with the Garcitas data in the original BBEST analysis) to a total inflow from these two watersheds
- Change 2: A change from monthly average salinity at an individual point to daily average salinity across a transect Change 3: An extension of the period of record (was November 1986 to July 2009 in the BBEST report and is November 1986 to December 2014 in this study)

The impacts of these updates produce flows (Table 5-10) that are less straightforward to interpret, though some of the values do change. The flows needed to produce subsistence-level salinity conditions have decreased, relative to the BBEST study flows, and the flows required for the base flow levels have increased. The decrease in subsistence flows, coupled with an increase in base flows, is due to the combination of changes that were implemented. When the same temporal characterization (Change 2) and same period of record is used (Change 3) and only the characterization of inflows are changed (Change 1), design flow of the base flows remain about the same, while the subsistence flow decreases. Accordingly, Change 1 leads to the decreases in required subsistence flows, while Changes 2 and 3 lead to the increases in base flows.

The BBEST analysis was intended to mimic the approach taken in the MBHE analysis for Matagorda Bay. The BBEST decision to develop regressions based on monthly average flows without considering different responses during low and normal conditions allowed for the analysis to be completed within the limited time allotted to the BBEST for this analysis. The analysis presented in this report is intended to bring the BBEST analysis approach in line with the more rigorous approach taken in the MBHE analysis. The observation that the values differ somewhat is not surprising.

5.6 Possible Future Work

There are two areas in which future analysis might be directed. The first generally would be to continue work with the TWDB to improve the hydrodynamic modeling and specifically the input data used to drive the model. The second would be to incorporate some of the lessons learned from the other BBEST analyses that have been conducted since the completion of MBHE.

Until recently, the TxBLEND models have been used to predict salinities over very broad geographic areas (whole bay) and at coarse time steps (monthly averages). These whole bay outputs are relatively insensitive to errors in estimates of return flows and diversions downstream of the USGS gages, because the return flows and diversions are generally small relative to the gaged flows. Since the passage of SB3, the analyses of Texas estuaries has begun to focus on smaller delta areas or on strategies to supplement or otherwise augment inflows to meet strategy frequency targets. At these finer levels of analysis the potential effect of errors in diversion and return flow data may become more significant.

A review of reported return flow data by subwatersheds (Table 5-11) suggests that there may have been some inconsistencies as to how diversion and return flow data have been updated through time. Several patterns in these data raise questions as to their accuracy. For example, for most of the watersheds, there appears to be a significant shift in return flows between 1999 and 2000. Given that some of the databases from which these data are derived are stored in separate files by decade, this shift may reflect an accounting error rather than a real change in return flows.

Similar issues may exist in the estimates of diversions. In general, return flows and diversions are less than 3% of the total inflows and, in many cases, cancel each other out; it is, therefore, unlikely that errors in these data would have a significant effect on modeled salinities, especially when considering the whole bay at monthly time scales. If, however, the recent estimates of very large return flows for the Lavaca River and Guadalupe River discussed in Section 5.1 (which are oftentimes larger than 3% of the total inflows) prove to be accurate, then a review and possible revision to the existing dataset could significantly impact results, particularly when focused on small delta areas or in considering strategies to augment inflows to meet attainment frequency targets. Updating the existing datasets is a substantial effort. Diversions are self-reported, and the databases that store this information contain gaps and duplications. In addition, for the South Texas Watermaster areas, including the Lavaca and Guadalupe basins, the data are maintained separately and in different formats from the data for the rest of the state. The return flow data carry their own set of challenges as do issues related to specific types of water use such as irrigation and water for power plant cooling. Reviewing older records and trying to rectify existing inflow estimates for the period from 1986 to 2009 would be a significant challenge. Once completed, it might require a recalibration of the existing TxBLEND model, which was calibrated and validated based on the existing datasets.

Another issue for which there are concerns about the TxBLEND input data has to do with the estimates of gaged flow at the Bay City. During low flows (less than 2,000 cubic feet per second [cfs]), tidal effects make the readings at this gage unreliable. Therefore, the USGS no longer publishes these low flow estimates, and LCRA calculates inflow based on a gage that they maintain at Lane City. The Lane City site is upstream of Bay City, so additional adjustments are made to account for diversions and returns that may occur between Lane City and Bay City.

The current version of TxBLEND maintained by the TWDB and the version that is used in this study, continue to rely on the Bay City gage, including the use of some older data that USGS has removed from its website. The continued use of the Bay City gage to estimate inflows from the Colorado River adds uncertainty to the estimates of flows needed to maintain threshold and Level 1 salinity conditions.

The LCRA has provided inflow estimates that are based on adjustments to the Lane City gage by subtracting diversions downstream of Lane City and upstream of Bay City (Gulf Coast irrigation districts 1 and 2) when flows are less than 2,000 cfs for the period from March 2004 to May 2015. When these are compared to the Bay City-derived inflows that are used in TxBLEND, the differences generally show that the Lane City-derived inflows are lower than the Bay City values; however, the comparisons are inconsistent, with some days higher and some lower. Given the multiple steps involved in running the TxBLEND model and developing regressions based on antecedent 30-day inflows (filtered for low flows) versus modeled salinity, it is difficult to speculate as to if or how much changing the inflows to estimates based on Lane City would have on the final estimate of flows needed to produce design criteria salinity conditions. This issue should be examined further in a future application of the model.

The second general area towards which future efforts might be directed would be to incorporate some of the approaches that were developed in other basins as part of the SB3 program. The analysis presented in this report attempted to stray as little as possible from the methodology used in the original MBHE study. Since that study was completed in 2008, the Texas Science Advisory Group guidelines on *Methodologies for Establishing a Freshwater Inflow Regime for Texas Estuaries* (SAC 2009) has been applied to most of the other major bays systems in the state. These guidelines incorporated many of the approaches developed in the MBHE study; in fact, given the resource limitation for the SB3 groups, these applications were far more limited than the work conducted in the MBHE study. However, some alternative or modified approaches were developed related to hydrodynamic modeling, which might be worth consideration. One area to consider would be to quantify the spatial extent of the areas of the bays that are within the salinity criteria defined through the oyster, dermo, vegetation biomass, and juvenile finfish and shellfish assessments included in this report.

5.7 Flows Necessary to Decrease Salinity to 2 Parts Per Thousand

The results of the updated dermo analysis have reinforced the importance of intermittent freshets to drive salinity down to very low levels (less than or equal to 2 ppt). In Matagorda Bay, while 30-day antecedent inflow appears to be a good indicator for predicting normal mid-range salinity conditions (15 to 30 ppt), events that drive salinity to very low levels tend to be short duration (less than 1 week, though 3 days appears to be a good predictor; Figure 5-14). Very rarely, if ever, do salinities decline to 2 ppt in response to sustained high flows over more than one week.

Figure 5-15 shows the relationship between the 3-day antecedent inflow to salinity at the Delta transect filtered for events when salinity was less than 5 ppt but greater than 0.5 ppt (to exclude extremely high flows that drive salinity to lower bound of 0 ppt) and which were greater than 10 ppt 1 week previous (to filter out sustained low salinity, which may have been the result of earlier high flows but for which salinity has yet to begin to rise. This analysis suggests that short-duration storm events on the order of 100,000 ac-ft over 3 days will result in salinity at the delta edge transect to fall below 2 ppt.

The Lavaca/Garcitas watershed appears to respond somewhat differently. When salinities fall below 2 ppt, it appears that this occurs most often in response to longer duration high flow events or a series of these events. Figure 5-16 provides a fairly typical picture of this type of event.

The BBEST report recommended a high flow pulse event on the order of 450,000 ac-ft within a one month period and within any season with the goal of dropping salinity to less than 5 ppt for up to 2 weeks every 5 to 10 years. Results from the salinity model suggest that 5 ppt could be achieved with lower inflow volumes however a target of 2 ppt might be achieved with inflows in the 450,000 ac-ft range (Figure 5-17).

6 CONCLUSIONS AND RECOMMENDATIONS

The original MBHE flow recommendations were based on salinity targets associated with desired ecological conditions for a number of organisms. Accordingly, the results herein, which are based on a subset of organisms, should not be considered a complete replacement for the MBHE work. Rather, as discussed in the sections that follow, this work provides some corroboration for the MBHE studies, and also provides some insight as to potential new flow standards that the BBASC and TCEQ may wish to consider, as well as directions for future study.

6.1 Oysters and Dermo

The updated dermo and oyster monthly regression model results are generally consistent with the MBHE results and the existing flow standards, and there does not appear to be an urgent need to modify the existing flow standards which, for dermo, were largely based on a 2-year average salinity regression. The results do suggest two potential modifications to current inflow standards: 1) incorporation of a new freshet component and 2) addition of a long-term inflow criterion for Lavaca Bay similar to the criterion currently employed in Matagorda Bay.

6.1.1 Potential New Freshet Component

Both the monthly regression models and the long-term reef average models highlight the combined importance of average salinity and freshet frequency on both dermo WP and oyster counts. These results provide ecological support for a new freshet component to the flow standard that the BBASC and TCEQ may wish to consider. Because of the importance of freshets, it is possible that management of low to medium levels of inflows in the future could be contingent on recent freshet frequency. For example, if few or no freshets have occurred in the recent past, then good dermo and oyster conditions would require higher subsequent inflows than if multiple freshets have occurred (and vice versa). Such a management option would increase the complexity of evaluations and operations and would require some level of a priori corroboration with ecological indicators other than dermo and oysters. However, despite the added complexity, such a management scheme may have ecological and water management advantages and accordingly may be worth considering in the future.

While the inflow standards currently do not include explicit freshet components (in the sense of a short term freshet), the instream flow standards do include a corollary hydrologic event, termed high flow pulses. If desired by the BBASC and TCEQ, this new freshet component would be scaled to drive the salinity below 2 ppt at the location desired to be protected. For Matagorda Bay, in the case of the CRD, a flow of approximately 100,000 ac-ft in the antecedent 3 days would be required. This inflow is equivalent to the existing high flow pulse specified in the instream flow standards at Wharton (27,000 cfs for 2 days, or 108,000 ac-ft; this high flow pulse was originally identified for the purpose of providing for beneficial sediment transport and has a return frequency of once every two years).

For Lavaca Bay, to reduce salinities at the Gallinipper transect to 2 ppt, a flow of approximately 450,000 ac-ft in the antecedent 30 days would be required. This flow is consistent with a previous recommendation by the BBEST, but no similar flow exists in the standards. The largest pulses (Annual Pulses) included in the instream flow standards for Lavaca at Edna, Navidad at Strane Park near Edna, and Garcitas near Inez have volumes of 18,400, 11,250, and 1,500 ac-ft, respectively. Assuming the pulse events occur concurrently at all three sites (which is not a requirement of the standards), this totals 31,150 ac-ft. Furthermore, assuming base wet conditions would apply on the non-pulse days, this would add an additional 8,600 ac-ft, bringing the monthly required inflow total to 39,750 ac-ft. Under this interpretation, the 450,000 ac-ft over 30 days is 410,250 ac-ft higher than the existing high flow pulses specified in the TCEQ standards. As another point of comparison, at the Route 35 Bridge transect, a flow of 200,000 ac-ft over the previous 30 days would be required to decrease the salinity at that location to 2 ppt. By a similar calculation, this 200,000 ac-ft, is 160,250 ac-ft higher than the existing high flow pulses specified in the TCEQ standards.

It is important to also consider the timing of such a freshet event. Ray (1987) argued that freshets during the summer and early fall may be most beneficial with respect to reducing dermo:

> [Dermo] is most active during the warm periods (summer and early fall). . . Thus, freshwater introduction during this time of year will be most effective in controlling the two most important oyster mortality agents [Dermo and Thais haemastoma, the southern oyster drill] on the Texas coast. Unfortunately, this

> time of year would probably be the most difficult in which to obtain the controlled release of freshwater from reservoirs in the estuaries because of reduced river flow. . . (p. E.14)

Accordingly, freshets during the summer and early fall may provide the most benefit with respect to dermo conditions. Such freshets in the summer should not be numerous however. Gulf oysters spawn primarily from April to November (Hofstetter 1977, as cited in Culbertson 2008), but spawning can occur in any month if temperatures are suitable (greater than 25°C; Britton and Morton 1989). Because the optimal salinity for spawning is considered to be relatively high (approximately 20 ppt), freshet events during spawns (i.e., April to November) should not be recommended to occur so frequently as to hinder spawning and recruitment.

Freshet duration is also a consideration, especially during high temperatures. La Peyre et al. (2003) observed increased mortality of oysters that were exposed for 21 days to a freshet (less than 1 ppt) during summer (28°C), as opposed to spring (18°C) or winter (16°C). La Peyre et al. (2009) showed that 19 days of 1 ppt conditions decreased dermo body burden (number of parasites per gram of oyster tissue) by over 90%, but longer durations of low salinity led to increased oyster mortality at summer temperatures. Hence, a freshet that imposes very low salinity conditions for 2 weeks or so should be sufficient to substantially reduce dermo. Conversely, due to mortality concerns, very large and longer-term freshets (approximately 1 ppt for more than 3 weeks) are contraindicated for oysters during the warmest months.

6.1.2 Potential New Long-Term Average Inflow Criterion for Lavaca Bay

Currently, the Matagorda Bay inflow standards include a long-term average inflow, but the Lavaca Bay inflow standards do not. Tables 5-3 and 5-4 provide inflows corresponding to salinity levels at various reefs in Matagorda and Lavaca Bays. This information may be used by the BBASC and TCEQ to identify long-term average inflows corresponding to desired salinity levels based on the ecological information presented in this report. For example, the relationship between reef average dermo WP and long-term salinity (Figure 2-10), and between average commercial-sized oyster count and long-term salinity (Figure 2-13), may be used to help identify long-term average salinity targets. As an example, for dermo, a long-term average salinity of 21 promotes an average WP of 1.0,[20] which corresponds to a light infection intensity with relatively low mortality (Bushek et al. 2012). Relatedly, a salinity of 20 corresponds to the maximum for commercial-sized oyster counts.

The Matagorda Bay long-term average inflow standard was largely based on the MBHE water quality study (MBHE 2007a). This flow was 1.4 million ac-ft per year, which is equivalent to 120,000 ac-ft per month. Based on the TxBLEND model and associated regressions (Table 5-3), this average flow would generally result in salinity values of 13 ppt, 14 ppt, and 17 ppt at the CRD, Shell Island Reef, and Mad Island Reefs, respectively. These values all would be expected to result in long-term dermo WP averages less than 1.0 and good oyster conditions (Table 2-2).

The Lavaca Bay inflow standards do not include a long-term average inflow and the current study did not perform a water quality modeling evaluation (as was performed in Matagorda Bay to support the long-term average inflow recommendation during the MBHE effort). However, a long-term average inflow recommendation could be constructed for Lavaca Bay based on dermo. As an example of how one might be considered, to maintain a long-term average dermo WP of 1.0 (which corresponds to an average salinity of 21 ppt) at the following reefs, the corresponding average inflows would be required

- North Reef: 4,000 ac-ft per month (48,000 ac-ft per year)
- Route 35 Bridge Reef: 6,700 ac-ft per month (80,000 ac-ft per year)
- Gallinipper Point: 18,000 ac-ft per month (216,000 ac-ft per year)
- Indian Point: 40,000 ac-ft per month (480,000 ac-ft per year)

In the BBEST report, of all the Lavaca Bay reefs identified as being important to protect desired salinity conditions, Indian Point Reef (referred to therein as Middle Ground Reef) was the most distant from freshwater inflows (BBEST 2011). Accordingly, a long-term average inflow standard of 480,000 ac-ft per year from the Lavaca River and Garcitas Creek[21]

[20] All reefs in this study had a long-term average dermo WP below 1.0, except Gallinipper Point at 1.1, and Confederate Reef at 1.7. Confederate Reef has previously been identified as having high levels of dermo (Ray 1987).
[21] This inflow (480,000 ac-ft per year) corresponds to the 31st percentile of historical inflows from Lavaca River and Garcitas Creek.

would promote low oyster mortality due to dermo in Indian Point Reef as well as the remaining key reefs in Lavaca Bay.

6.2 Marsh Productivity

Field data from both bay systems collected in 2014 as part of this study aided in the testing of underlying biological relationships previously used in the development of freshwater inflow criteria for Matagorda Bay. Although the majority of the discussion presented above speaks directly to the CRD, the data collection effort in Lavaca Bay begins the foundation for the baseline record necessary to potentially apply similar methodologies (as those applied in Matagorda Bay) to Lavaca Bay inflow criteria development in the future. Based on the similarities in both marsh and juvenile finfish and shellfish communities in these adjacent systems, an expanded evaluation of Lavaca Bay inflow criteria appears appropriate pending the collection of additional data.

It is acknowledged that this limited 2014 dataset represents only a snapshot in time for each bay system and should be interpreted with caution. However, considering the freshwater inflow conditions experienced during this study (i.e., extended low inflow), the results generated do provide a solid foundation on which the BBASC can build. The independent results from both the marsh vegetation biomass and white shrimp and blue crab habitat use evaluations conducted in 2014 provide additional support of certain underlying biological relationships developed during the MBHE study. Additionally, the results from the marsh vegetation biomass and throw trap community analysis revealed direct relationships between marsh vegetation (e.g., habitat), freshwater inflow, and salinity. These findings support the use of habitat as an indicator in setting freshwater inflow recommendations as well as support the concept of multiple tiers and achievement guidelines as currently in place. As noted throughout this section, complexities with antecedent inflow conditions and varying ecological responses make additional long-term monitoring in these systems and additional statistical analysis of existing databases necessary to conduct a rigorous inflow criteria validation.

The inability to show direct linkages in organism abundances and densities or throw trap community condition with freshwater inflow or salinity is disappointing but not unexpected for mobile organisms in an estuarine environment. Albeit only a limited investigation, the finding of no live *Rangia* clams in either delta area during this study stimulates questions as to why. Speculation ranges from the effects of man-made alterations over time in both these river deltas to only limited sampling efforts being employed during this study. Only additional, more expansive study efforts will answer these questions. Under the current circumstances in which *Rangia* was not present in either bay system, their use as an indicator species was not recommended for this study.

6.3 Salinity Modeling

The modeling transition from RMA2 to TxBLEND has indicated that the results from these models are generally similar. Additionally, the extension of the period of record for TxBLEND has indicated that the model results, as summarized by the inflow-salinity regressions, are generally similar among the different time periods considered. However, "generally similar" does not mean "identical." There are differences, and if the BBASC and TCEQ wish to use the latest TxBLEND model to update the flow recommendations, the appropriate values are shown in Table 5-7.

Similarly, the extension of the period of record for TxBLEND and revision of the inflow-salinity regressions used by the BBEST for the Lavaca River and Garcitas Creek results in a final regression that is generally similar to that used by the BBEST. Should the BBASC and/or TCEQ deem these differences important and wish to update the flow recommendations, the appropriate values are shown in Table 5-8.

The most significant changes in flow recommendations that would be derived from this analysis, based solely on updating the hydrodynamic modeling and salinity-inflow regression, are the changes to the Threshold and Level 1 inflows to Matagorda Bay and the subsistence flows to Lavaca Bay. These lower tiers are capable of being directly impacted by water management operations. It is important to keep in mind that the inflow volumes are only a part of the equation, the other important part is understanding how often these different flow levels should occur or their attainment frequency targets. If the BBASC and/or TCEQ decides that changes to the recommended flows, based solely on changes to the hydrodynamic analysis, are merited, then the attainment frequency targets should also be re-evaluated. In the previous Matagorda and Lavaca freshwater inflow analyses, attainment frequency recommendations were calculated based on the historical annual frequency for which all of the seasonal components of the criteria were achieved. In the Matagorda study these frequencies were based on Bay City gage flow minus STP diversions for the period from 1948 to 2007, the Lavaca/Garcitas study used inflow data for a period for 1940 to 2009. For this current study total inflows based on gage data with corrections for ungaged runoff, diversions and returns are available from 1977 to 2014 and thus this period was used for both inflow watersheds. Tables 6-1 and 6-2 show the frequency of occurrence for each of the flow criteria.

Perhaps more important than the changes to the various flow levels that are calculated from this analysis is the production of a consistent and flexible approach and tool. Prior to this analysis, the RMA hydrodynamic model simulated a limited period of record, and the inflow analyses applied to the two bays differed due to limitations, which prevented the re-running of the RMA model during the BBEST analysis for Lavaca Bay. The approach of using the TxBLEND model in Lavaca Bay presented in this report should be adopted and used in future studies because it will allow for the continued application of a consistent approach that can continue to be updated as additional data and understanding of the estuarine system are developed.

6.4 Recommendations for Future Efforts

Section 6.4 provides suggestions for future efforts that would enhance the BBASC's assessment of freshwater inflow standards for Matagorda and Lavaca Bays.

6.4.1 Marsh and Reef Monitoring

It is not surprising that the organisms with the clearest dependence on inflows and salinity are the sessile organisms: dermo, oysters, and marsh vegetation. These organisms, once established, are relatively easy to sample and cannot move as salinity conditions change. Accordingly, their health, density, and extent form useful indicators for ecosystem health. Importantly, these organisms are not simply valuable in their own right as oyster reefs provide numerous ecological functions beyond simply oyster production (Grabowski et al. 2012), and marsh vegetation provides habitat for a host of other species.

In some cases, the sampling programs carried out for the work presented herein did not find the biota to be as negatively impacted by the drought as expected. This may be encouraging news, or may be due to secondary effects of poor habitat conditions (e.g., lower dermo may result from lower oyster density; higher juvenile finfish and shellfish density may result from clumping in refuge locations). A well-designed marsh and oyster reef monitoring program is a critical cornerstone to ongoing improved understanding of the relationship between inflow and bay health. It is recommended that the BBASC consider a basic long-term monitoring program, including:

1. Re-establishment of dermo monitoring in Matagorda and Lavaca Bays. Dermo provides a robust indicator of ecological condition and is strongly influenced by inflows and salinity. While the information presented herein is sufficient to estimate future dermo conditions without the benefit of future dermo data, such data would be helpful as part of a continuing corroboration and validation effort. Future dermo data would also help identify dermo epizootics when they occur.

   If dermo monitoring was re-established in Matagorda and Lavaca Bays, the quantitative real-time polymerase chain reaction method of assaying dermo infections should be considered due to the somewhat subjective nature of the RFTM method (TPWD and TWDB 2011 describes several advantages and disadvantages of the two methods for quantifying dermo). Multiple seasonally targeted sampling events per year may be more cost-effective than re-instating the full former program, which monitored six reefs once per month.

2. Development of marsh productivity monitoring in each delta system. The program should include marsh vegetation biomass as well as throw trap sampling in each delta to establish the condition of habitats, their inhabitants, and the relationship of each to freshwater inflow over time. Having annual end of growing season marsh vegetation biomass data for 5 to 10 years and corresponding juvenile finfish and shellfish data from these areas will be invaluable in teasing out the complexities with antecedent inflow conditions. In order to encompass more bay species in the analysis, a seasonal spring and fall sampling for shellfish and finfish would be required.

These monitoring programs do not have to be extensive or expensive, just well designed with analysis endpoints clearly established, and consistently conducted over a period of years.

This type of simplified, long-term monitoring data with an eye towards documenting and understanding the intermediary link of habitat is essential, in our opinion, to truly test the applicability of multi-tiered, achievement guideline-based freshwater inflow criteria.

6.4.2 Analysis and Modeling

6.4.2.1 Oysters and Dermo

The health of oysters in Matagorda and Lavaca Bays is influenced by both average salinity and freshet frequency, but current inflow standards are only designed to manage average salinity (with seasonal variations and different levels). Identification of an approach to link freshet effects on oyster health to actionable management options within the existing BBASC and TCEQ frameworks is recommended as discussed in Section 6.1.1. Much of the unexplained variation in oyster and dermo condition may be explained by factors other than temperature and salinity. For instance, oysters thrive (and outgrow dermo) when food is plentiful. Primary productivity in Matagorda and Lavaca Bays is likely nitrogen-limited, and the Colorado River is an important source of nitrogen (MBHE 2007a). Accordingly, freshwater inflows may have an indirect and positive effect on food supply for oysters. An analysis sufficient to identify this effect for oysters in Matagorda and Lavaca Bays may require new data collection as well as additional analyses.

6.4.2.2 TxBLEND Modifications

Possible future modifications to modeling salinities in Matagorda and Lavaca Bays are provided in Section 5.6. In general, there are some inconsistencies in the TxBLEND model and input data bases that should be carefully evaluated and addressed in future studies.

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TABLES

Table 2-1
Summary of TPWD Oyster Dredge Stations and Dermo Watch Stations Combined to Create Reef Locations

Reef Location	Major Bay Area	Reef Name	Stations in Reef	Number of Stations	Oyster Sentinel
1	Matagorda	Sammys Reef	240, 280, OS	3	Yes
2	Matagorda	Mad Island Reef	203, 243, 244, OS	4	Yes
3	Matagorda	Shell Island Reef	205, OS	2	Yes
4	Matagorda	Charancahua Bay	88, 89, 116, 117, 118, 148	6	No
5	Matagorda	Tres Palacios Bay	74, 93, 94, 95, 124, 158	5	No
6	Matagorda	Halfmoon Reef	316, 317, 347, 348, 349, 379	6	No
7	Lavaca	Indian Point	332,364, OS	3	Yes
8	Lavaca	Gallinipper Point	297, 298, 254, 255, OS	5	Yes
9	Lavaca	North Reef	44, 45, 46, 60, 61	4	No[1]
10	Lavaca	Rt 35 Bridge	107, 108, 136, 137	4	No
11	Galveston	Fisher's Reef	112, 113, 114, 133, 134, OS	6	Yes
12	Galveston	Frenchy's Reef	326, 327, 358, 359, OS	5	Yes
13	Galveston	Redfish Reef	343, 344, 345, 346, 370, 371, 372, 373, OS	9	Yes
14	Galveston	Hanna's Reef	376, 377, 378, 379, 402, 403, 404, 405, 406, OS	10	Yes
15	Galveston	Confederate Reef	562, 563, 564, 574, 575, 576, 577, 578, OS	9	Yes
16	San Antonio	V Reef	102, 103, 104, 120, 121, 122, OS	7	Yes
17	San Antonio	2nd Chain Reef	165, 177, 178, 188, OS	5	Yes
18	San Antonio	1st Chain Reef	99, 281, 282, 111, 112, OS	6	Yes
19	San Antonio	Port O'Connor	211, 212, 217, 218	4	No
20	San Antonio	Espiritu Santo Bay	262, 263, 274, 275, 276	5	No

Note:

1 = Regular monitoring for dermo at North Reef began during the MBHE (Matagorda Bay Health Evaluation) project, and this reef is now an Oyster Sentinel (OS) reef for the updated analysis.
TPWD = Texas Parks and Wildlife Department

Table 2-2
Late Summer 2014 Oyster Dermo Results

Station[1]	Collection Date	Size Range (mm)[2]	T (°C)	S (ppt)	PI	WP	II	Number of Oysters Assayed
Sammy's Reef	8/25/2014	76-101	31	31.5	28.6	0.38	1.33	7
Galliniper Point	8/25/2014	83-111	30.5	30.2	66.7	0.75	1.12	12
Indian Point	8/25/2014	76-113	30.4	31	41.7	0.81	1.92	12
Mad Island Reef	8/25/2014	78-141	32	32	66.7	0.72	1.08	12
Route 35 Reef	8/25/2014	81-131	30.6	26.5	50	0.42	0.83	12
North Reef	8/25/2014	90-119	30.2	21	0	0	0	12
Shell Island	8/25/2014	85-124	33.3	31.8	0.8	0.83	1.11	12
CRD 1	9/1/2014	90-150	27.8	27.9	83.3	1	1.2	12
CRD 4	9/1/2014	100-127	28.3	27.2	91.7	1	1.09	12
CRD 5	9/1/2014	78-110	28.8	27.3	75	0.97	1.3	12
CRD 6	9/1/2014	93-129	30.1	26	91.7	0.81	0.88	12
CRD 8	9/1/2014	83-114	29.1	27.8	91.7	1.28	1.39	12

Notes:

Collection information, oyster size ranges, environmental data, and disease levels

1 = Commercial-size oysters (greater than 75 mm) from stations (reefs) in Matagorda and Lavaca Bays

2 = Size range refers to oyster length

°C = degrees Celsius

CRD = Colorado River Delta

II = Infection Intensity

mm = millimeter

PI = percent infection

ppt = parts per thousand

S = salinity

T = temperature

WP = weighted prevalence

Table 2-3
Late Fall 2014 Oyster Dermo Results

Station[1]	Collection Date	Size Range (mm)[2]	T (°C)	S (ppt)	PI	WP	II	Number of Oysters Assayed
Sammy's Reef	11/16/2014	79-130	11.3	30.9	33	0.31	0.92	12
Indian Point	11/15/2014	78-120	11.5	31	58	0.44	0.76	12
Mad Island Reef	11/16/2014	80-99	11	30.4	8	0.14	1.67	12
North Reef	11/15/2014	88-145	9.5	25.5	75	1.22	1.63	12
CRD 5	11/16/2014	79-109	13.5	25.3	83	0.8	0.97	12
CRD 6	11/16/2014	92-130	13.2	28.2	83	1.28	1.53	12

Notes:

Collection information, oyster size ranges, environmental data, and disease levels

1 = Commercial-size oysters (greater than 75 mm) from stations (reefs) in Matagorda and Lavaca Bays

2 = Size range refers to oyster length

°C = degrees Celsius

CRD = Colorado River Delta

II = Infection Intensity

mm = millimeter

PI = percent infection

ppt = parts per thousand

S = salinity

T = temperature

WP = weighted prevalence

Table 2-4
Average Parameters for Reef Locations

Major Bay Area	Reef Location Name	Temperature °C	Months of Temperature Data[1]	Salinity	Months of Salinity Data[1]	Total Oyster Count[2]	Commercial Oyster Count[2]	OCI	Months of Oyster Count Data[1]	Dermo Weighted Prevalence[3]	DCI	Months of Weighted Prevalence Data[1]
Matagorda	Sammy's Reef	23.14	180	21.3	180	22	5	0.28	146	1.04	0.60	97
Matagorda	Mad Island Reef	23.4	204	20.1	204	22	6	0.30	192	0.76	0.69	75
Matagorda	Shell Island Reef	23.1	153	19.3	153	27	5	0.25	98	0.27	0.88	62
Matagorda	Charancahua Bay	22.9	223	21.3	223	15	5	0.31	223			
Matagorda	Tres Palacios Bay	22.7	216	22.5	216	22	6	0.36	216			
Matagorda	Halfmoon Reef	22.6	214	24.9	214	1	0	0.01	214			
Average for bay		**23.0**	**198**	**21.6**	**198**	**18**	**5**	**0.25**	**182**	**0.69**	**0.73**	**78**
Lavaca	Indian Point	22.9	187	23.2	187	30	5	0.22	163	1.06	0.58	82
Lavaca	Gallinipper Point	22.6	201	21.4	201	45	8	0.36	180	1.12	0.56	84
Lavaca	North Reef	22.5	212	16.8	212	8	2	0.15	209	0.06	0.97	30
Lavaca	Rt 35 Bridge	22.6	207	19.0	207	29	5	0.29	207			
Average for bay		**22.7**	**202**	**20.1**	**202**	**28**	**5**	**0.26**	**190**	**0.75**	**0.71**	**65**
Galveston	Fisher's Reef	22.1	193	13.0	194	16	4	0.19	165	0.12	0.95	88
Galveston	Frenchy's Reef	22.1	159	15.3	159	11	4	0.15	124	0.50	0.79	68
Galveston	Redfish Reef	21.9	212	17.1	212	21	5	0.31	197	1.05	0.60	91
Galveston	Hanna's Reef	21.9	213	17.8	214	18	4	0.23	206	0.86	0.67	113
Galveston	Confederate Reef	22.9	205	25.1	205	10	3	0.22	186	1.74	0.39	137
Average for bay		**22.2**	**196**	**17.7**	**197**	**15**	**4**	**0.22**	**176**	**0.86**	**0.68**	**99**
San Antonio	V Reef	22.6	207	16.0	208	16	4	0.25	198	0.02	0.99	55
San Antonio	2nd Chain Reef	23.1	180	18.6	179	24	6	0.31	151	0.69	0.73	87
San Antonio	1st Chain Reef	22.8	203	23.5	203	20	4	0.27	190	0.84	0.68	85
San Antonio	Port O'Connor	23.4	182	27.0	182	7	2	0.12	182			
San Antonio	Espiritu Santo Bay	22.9	198	26.0	198	7	1	0.11	198			
Average for bay		**23.0**	**194**	**22.2**	**194**	**15**	**3**	**0.21**	**184**	**0.52**	**0.80**	**76**
Range of Reef Location Averages												
Minimum		21.9	153	13.0	153	1	0	0.01	98	0.02	0.39	30
Maximum		23.4	223	27.0	223	45	8	0.36	223	1.74	0.99	137

Notes:

Average results are the average of monthly averages, where available, from 1996 through 2014.

1 = Number of months with available data

2 = Average live oysters per individual dredge pull

3 = Average Mackin score for commercial oysters

°C = degrees Celsius

DCI = Dermo Condition Index (varies inversely with weighted prevalence)

OCI = Oyster Condition Index

Table 2-5
Average Parameters for Oyster Sentinel Reef Locations, 2004 to 2009 Data Only

Major Bay Area	Reef Location Name	Temperature °C	Months of Temperature Data[1]	Salinity	Months of Salinity Data[1]	Total Oyster Count[2]	Commercial Oyster Count[2]	OCI	Months of Oyster Count Data[1]	Dermo Weighted Prevalence[3]	DCI	Months of Weighted Prevalence Data[1]
Matagorda	Sammy's Reef	23.3	69	19.7	69	36	7	0.35	45	0.88	0.65	62
Matagorda	Mad Island Reef	23.5	68	18.4	68	26	6	0.29	60	0.48	0.79	41
Matagorda	Shell Island Reef	23.1	67	17.3	67	36	7	0.31	31	0.11	0.95	38
Average for bay		**23.3**	**68**	**18.5**	**68**	**33**	**7**	**0.31**	**45**	**0.49**	**0.80**	**47**
Lavaca	Indian Point	22.9	72	21.2	72	55	6	0.29	53	0.99	0.60	62
Lavaca	Gallinipper Point	22.6	69	19.3	69	54	10	0.40	53	1.10	0.57	64
Lavaca	North Reef	22.9	70	14.6	70	10	2	0.14	67	0.02	0.99	28
Average for bay		**22.8**	**70**	**18.4**	**70**	**39**	**6**	**0.28**	**58**	**0.71**	**0.72**	**51**
Galveston	Fisher's Reef	22.1	63	11.4	64	4	1	0.08	49	0.00	1.00	28
Galveston	Redfish Reef	22.1	69	16.5	69	16	3	0.23	61	0.74	0.69	35
Galveston	Hanna's Reef	22.1	67	16.7	68	6	2	0.16	63	0.65	0.73	46
Galveston	Confederate Reef	23.1	71	24.5	71	11	3	0.24	60	1.66	0.41	67
Average for bay		**22.4**	**68**	**17.3**	**68**	**9**	**2**	**0.18**	**58**	**0.77**	**0.71**	**44**
San Antonio	V Reef	22.6	66	14.7	66	15	2	0.16	58	0.00	1.00	35
San Antonio	2nd Chain Reef	23.5	66	16.6	65	15	4	0.23	49	0.58	0.76	49
San Antonio	1st Chain Reef	23.0	69	23.1	69	25	5	0.31	63	0.93	0.65	51
Average for bay		**23.0**	**67**	**18.1**	**67**	**18**	**4**	**0.23**	**57**	**0.50**	**0.80**	**45**
Range of Reef Location Averages												
Minimum		22.1	63	11.4	64	4	1	0.08	31	0.00	0.41	28
Maximum		23.5	72	24.5	72	55	10	0.40	67	1.66	1.00	67

Notes:

Average results are the average of monthly averages, where available, from 2004 through 2009.

1 = Number of months with available data

2 = Average live oysters per individual dredge pull

3 = Average Mackin score for commercial oysters

°C = degrees Celsius

DCI = Dermo Condition Index (varies inversely with weighted prevalence)

OCI = Oyster Condition Index

Table 3-1
A List of Time Periods for Which Each Key Juvenile Species is Most Abundant in Matagorda Bay

Species	**TPWD Coastal Fisheries Database**
	Bag Seine[1]
Blue crab	February-June
Brown shrimp	April-July
White shrimp	July-November
Atlantic croaker	January-June
Gulf menhaden	April-August

Notes:

1 = Data from low estuarine marsh edge and shallow non-vegetated bottom habitats only

Based on data from the Texas Parks and Wildlife Department coastal fisheries database (MBHE 2006c)

Table 3-2
Habitat Quality Rank for Each MBHE Trophic Level

Trophic Level	Habitat Quality Rank				
	Selected	Good	Fair	Poor	Refuge
	90-100% WUA	75-90% WUA	50-75% WUA	25-50% WUA	<25% WUA
	Salinity Range (PPT)				
Shellfish					
White Shrimp	8-15	15-20	20-25	25-30	>30
Blue Crab	5-15	15-20	20-25	25-30	>30
Brown Shrimp	10-25	7-10, 25-30	30-32	32-35	>35
Forage Fish					
Gulf Menhaden	5-15	15-20	20-23	23-28	>28
Atlantic Croaker	5-15	15-20	20-23	23-26	>26
Low Estuarine Marsh	0-15	15-20	20-25	25-30	>30

Notes:
Weighted usuable area (WUA) calculated for each species includes selection for physical habitat and salinity
MBHE = Matagorda Bay Health Evaluation
PPT = parts per thousand

Table 3-3
Summary of Freshwater Inflow Criteria (MBHE 2008a)

	Threshold	MBHE 1	MBHE 2	MBHE 3	MBHE 4	Long-Term Volume and Variability
Design Area	**Delta**	**Delta Edge to Mad Island Transect**	**Delta Edge to Mad Island Transect**	**Delta Edge to Mad Island Transect**	**Delta Edge to Mad Island Transect**	**EAMB**
Salinity range across area (ppt)	<30[1]	27-29	24-26	20-23	15-18	Average[4]
Trophic Level						
Primary Production	Low	Low	Low	Moderate	High	Normal[5]
Oyster Health	Refuge[2]	Refuge[2]	Poor[2]	Fair	Good	Normal[5]
Benthic Condition	Fair/Poor	Poor	Fair	Good	Peak	Normal[5]
Marsh Productivity	Fair	Fair	Good	Good	Good	Normal[5]
Shellfish Habitat	Good[3]/Poor	Good[3]/Poor	Selected[3]/Fair/Poor	Selected[3]/Fair	Selected[3]/Good	Normal[5]
Forage Fish Habitat	Poor/Refuge	Poor/Refuge	Poor	Fair	Good	Normal[5]

Notes:

1 = This would be typical when no significant local watershed inflows have occurred.

2 = Potentially detrimental to select reefs based on Dermo Condition Index. However, a condition experienced a similar amount of time historically.

3 = Ranking applies to brown shrimp. Blue crab and white shrimp habitats ranks lower.

4 = The long-term average salinity will be in the mid teens but include very low and high periods.

5 = Indicators of productivity and health will be normal, but will experience variations during dry and wet periods.

EAMB = Eastern Arm of Matagorda Bay

MBHE = Matagorda Bay Health Evaluation+A1

Table 3-4
October 2014 Marsh Vegetation Biomass Collected from the CRD and LRD

Marsh Vegetation Above Ground Biomass (g/m^2)					
Colorado River Delta (CRD)			**Lavaca River Delta (LRD)**		
Sites	**Edge**	**Interior**	**Sites**	**Edge**	**Interior**
CRD 1	296	376	LRD 1	120	200
CRD 1 (replicate)	344	440	LRD 1 (replicate)	168	200
CRD 2	376	256	LRD 2	288	368
CRD 2 (replicate)	472	264	LRD 2 (replicate)	192	280
CRD 3	384	384	LRD 3	112	248
CRD 3 (replicate)	225	480	LRD 3 (replicate)	192	136
CRD Average	350	367	LRD Average	179	239

Note:
g/m^2 = grams per square meter

Table 3-5
2014 Water Quality Measurements Collected During Throw Trap Sampling

Lavaca River Delta Averaged Measurements						
Month	**Temperature (°C)**	**pH**	**Salinity (ppt)**	**Conductivity (mS)**	**D.O. (mg/L)**	**D.O. Saturation (%)**
September	29.37	8	23.07	36.65	6.44	95.76
October	24.59	7.94	23.41	36.96	7.42	101.34
Colorado River Delta Averaged Measurements						
Month	**Temperature (°C)**	**pH**	**Salinity (ppt)**	**Conductivity (mS)**	**D.O. (mg/L)**	**D.O. Saturation (%)**
September	31.08	7.71	28.47	44.32	5.92	93.11
October	27.09	7.97	25.75	40.54	6.57	94.89

Notes:

Trap sampling from the Lavaca River Delta and Colorado River Delta

°C = degrees Celsius

D.O. = dissolved oxygen

mg/L = milligrams per liter

mS = millisiemens

ppt = parts per thousand

Table 3-6
2014 Throw Trap Species Abundance in CRD and LRD

Species	Colorado River Delta (CRD)		Lavaca River Delta (LRD)	
	Marsh Edge	Shallow Nonvegetated Bottom	Marsh Edge	Shallow Nonvegetated Bottom
ATLANTIC CROAKER	5	0	0	0
BAY ANCHOVY	26	122	399	332
BAYOU KILLIFISH	11	0	0	3
BLACKCHEEK TONGUEFISH	11	16	16	17
BLUE CRAB	163	48	52	7
BROWN SHRIMP	11	35	21	2
CLOWN GOBY	0	1	0	0
DARTER GOBY	220	182	57	15
FLAGFIN MOJARRA	0	2	0	0
GRASS SHRIMP	1,307	241	1,020	31
GREEN GOBY	5	9	0	0
GULF KILLIFISH	0	9	4	0
GULF MENHADEN	0	0	45	0
GULF PIPEFISH	1	0	0	0
GULF TOADFISH	0	0	1	0
HARDHEAD CATFISH	0	0	1	6
HOGCHOKER	0	0	0	1
INLAND SILVERSIDE	0	0	4	12
INSHORE LIZARDFISH	0	0	0	1
LINED SOLE	4	7	0	0
MULLET	1	0	0	0
NAKED GOBY	12	1	12	6
OYSTERCRACKER	0	0	0	1
PINK SHRIMP	1	0	3	0
SHARPTAIL GOBY	0	5	0	0
SHEEPSHEAD	1	0	3	0
SKILLETFISH	0	0	1	0
SNAPPING SHRIMP	31	24	4	0
SPECKLED WORM-EEL	1	1	2	0
SPECKLED TROUT	14	1	23	1
STONE CRAB	2	1	11	11
VIOLET GOBY	1	1	0	0
WHITE SHRIMP	236	130	96	40
Total Individuals	**2,064**	**836**	**1,775**	**486**
Total Number of Species	**21**	**19**	**20**	**16**

Table 3-7
Adjusted Mean Values of SPAL in CRD at Different Salinity Levels

Salinity	5 ppt	10 ppt	15 ppt	20 ppt	25 ppt
biomass (g/.25 m^2)	455.37	361.85	268.33	174.81	81.29

Notes:

g/.25 m^2 = grams per 0.25 of a square meter

CRD = Colorado River Delta

ppt = parts per thousand

SPAL = *Spartina alterniflora*

Table 5-1
Development of Salinity and Inflow Regression Relationships

	MBHE (Colorado)	BBEST (Lavaca)	This Study (Both)
Hydrodynamic Model	RMA	TxBLEND	TxBLEND
Period of Record	8 years, 6 months (July 1995 - December 2003)	20 years, 10 months (November 1986 - August 2006)	29 years, 2 months (November 1986 - December 2014)
Response Variable (Salinity)	Transect Daily Average	Point Monthly Average	Transect Daily Average
Predictor Variable (Inflow)	Cumulative 30-day antecedent inflow volume	Total Inflow in the Month Corresponding to the Average Salinity	Cumulative 30-day antecedent inflow volume
Inflow Time Series (Normal vs. Low Flow Separation)	Low defined as periods with flow less than long-term monthly average (171,600 ac-ft) for at least 20 months	N/A	Colorado - Low defined as periods with flow less than long-term monthly average (143,191 ac-ft) for at least 14 months Lavaca - Low defined as periods with flow less than long term monthly average (102,729 ac-ft) for at least 6 months
Regression Formula	$S_d = a * Ln(Q_{d-30}) + b$	$S_m = a * Ln(Q_m) + b$	$S_d = a * Ln(Q_{d-30}) + b$

Notes:

ac-ft = acre-feet

BBEST = Basin and Bay Expert Science Team

MBHE = Matagorda Bay Health Evaluation

RMA = USACE RMA model

S_d = Estimate of daily average salinity

S_m = Estimate of monthly average salinity

Q_{d-30} = 30 day cumulative antecedent inflow

Q_m = Monthly inflow

a,b = coefficients

Ln = natural logarithm

Table 5-2
Lavaca and Guadalupe Return Flows

Year	Lavaca (ac-ft/yr)	Guadalupe (ac-ft/yr)
1977	10,535	120,266
1978	5,065	127,852
1979	5,220	105,242
1980	2,262	96,993
1981	1,677	73,977
1982	1,277	74,975
1983	1,125	43,922
1984	1,100	55,771
1985	1,064	40,853
1986	853	35,314
1987	914	35,113
1988	762	22,398
1989	852	23,992
1990	797	25,168
1991	945	20,071
1992	853	16,777
1993	760	19,934
1994	884	25,875
1995	703	25,783
1996	581	25,788
1997	1,161	14,218
1998	1,600	13,431
1999	822	13,761
2000	764	26,402
2001	1,004	26,337
2002	947	26,245
2003	883	23,274
2004	1,251	27,033
2005	1,183	24,286
2006	1,037	25,028
2007	916	29,873
2008	794	25,911
2009	640	24,734
2010	74,818	103,401
2011	39,671	56,554
2012	69,637	74,220
2013	0	0
2014	0	0

Notes:
ac-ft/yr = acre-feet per year

Table 5-3
Colorado Blending of Low and Normal Regression Equations

Slope	-7.746	-7.394		-7.003	-7.36		-5.513	-6.85	
Intercept	103.642	99.358		97.123	100.25		83.376	96.848	
Salinity (ppt)	**Delta Low Eq.**	**Delta Normal Eq.**	**Delta**	**Shell Island/Tripod Low Eq.**	**Shell Island/Tripod Normal Eq.**	**Shell Island/Tripod**	**Mad Island Low Eq.**	**Mad Island Normal Eq.**	**Mad Island**
1	568,429	598,655	598,655	913,467	718,449	718,449	3,083,791	1,192,948	1,192,948
2	499,587	522,926	522,926	791,921	627,176	627,176	2,572,238	1,030,914	1,030,914
3	439,082	456,777	456,777	686,547	547,499	547,499	2,145,543	890,890	890,890
4	385,905	398,996	398,996	595,195	477,944	477,944	1,789,631	769,884	769,884
5	339,168	348,524	348,524	515,998	417,225	417,225	1,492,759	665,314	665,314
6	298,092	304,436	304,436	447,339	364,220	364,220	1,245,133	574,947	574,947
7	261,990	265,926	265,926	387,816	317,949	317,949	1,038,585	496,854	496,854
8	230,260	232,287	232,287	336,213	277,556	277,556	866,300	429,368	429,368
9	202,374	202,903	202,903	291,476	242,295	242,295	722,594	371,049	371,049
10	177,864	177,236	177,236	252,692	211,513	211,513	602,727	320,651	320,651
11	156,323	154,816	154,816	219,069	184,642	184,642	502,744	277,098	277,098
12	137,391	135,232	135,232	189,919	161,185	161,185	419,346	239,461	239,461
13	120,751	118,126	118,126	164,649	140,708	140,708	349,783	206,936	206,936
14	106,127	103,183	103,183	142,740	122,832	122,832	291,760	178,829	178,829
15	93,274	90,130	90,130	123,747	107,227	107,227	243,361	154,539	154,539
16	81,978	78,729	78,729	107,281	93,605	93,605	202,991	133,549	133,549
17	72,050	68,770	68,770	93,006	81,713	81,713	169,318	115,409	115,409
18	63,324	60,071	60,071	80,631	71,332	71,332	141,231	99,734	99,734
19	55,655	52,472	54,923	69,902	62,270	62,270	117,803	86,187	86,187
20	48,914	45,834	49,776	60,601	54,359	56,838	98,261	74,481	74,481
21	42,990	40,036	44,628	52,537	47,453	51,406	81,961	64,364	64,364
22	37,784	34,972	39,481	45,547	41,425	45,974	68,365	55,622	55,622
23	33,208	30,548	34,333	39,486	36,162	40,542	57,024	48,067	50,018
24	29,186	26,684	29,186	34,232	31,568	35,109	47,565	41,538	44,415
25	25,651	23,308	25,651	29,677	27,558	29,677	39,675	35,896	38,811
26	22,545	20,360	22,545	25,728	24,057	25,728	33,093	31,021	33,207
27	19,814	17,784	19,814	22,305	21,000	22,305	27,604	26,807	27,604
28	17,415	15,535	17,415	19,337	18,332	19,337	23,025	23,166	23,025
29	15,306	13,570	15,306	16,764	16,004	16,764	19,205	20,020	19,205
30	13,452	11,853	13,452	14,533	13,970	14,533	16,019	17,300	16,019
31	11,823	10,354	11,823	12,600	12,196	12,600	13,362	14,951	13,362
32	10,391	9,044	10,391	10,923	10,646	10,923	11,145	12,920	11,145

Table 5-3
Colorado Blending of Low and Normal Regression Equations

Notes:
All values are in acre-feet per month
eq. = equation
ppt = parts per thousand

Table 5-4
Lavaca Bay Blending of Low and Normal Regression Equations

Slope	-3.944	-4.332		-3.597	-4.389		-2.587	-3.821		-2.024	-3.262	
Intercept	53.655	56.261		52.683	58.991		47.153	57.748		44.712	55.554	
Salinity (ppt)	**North Low Eq.**	**North Normal Eq.**	**North**	**Rt 35 Low Eq.**	**Rt 35 Normal Eq.**	**Rt 35**	**Gallinipper Low Eq.**	**Gallinipper Normal Eq.**	**Gallinipper**	**Indian Point Low Eq.**	**Indian Point Normal Eq.**	**Indian Point**
1	628,504	346,384	346,384	1,738,719	547,910	547,910	56,051,860	2,821,365	2,821,365	2,385,014,548	18,361,312	18,361,312
2	487,740	274,988	274,988	1,316,704	436,266	436,266	38,080,286	2,171,652	2,171,652	1,455,331,564	13,513,033	13,513,033
3	378,503	218,308	218,308	997,118	347,370	347,370	25,870,831	1,671,557	1,671,557	888,040,688	9,944,935	9,944,935
4	293,731	173,310	173,310	755,102	276,589	276,589	17,576,020	1,286,625	1,286,625	541,880,822	7,318,989	7,318,989
5	227,945	137,588	137,588	571,826	220,230	220,230	11,940,726	990,337	990,337	330,654,698	5,386,420	5,386,420
6	176,893	109,228	109,228	433,035	175,355	175,355	8,112,242	762,279	762,279	201,764,899	3,964,143	3,964,143
7	137,275	86,714	86,714	327,930	139,624	139,624	5,511,262	586,739	586,739	123,116,576	2,917,417	2,917,417
8	106,530	68,841	68,841	248,336	111,174	111,174	3,744,219	451,623	451,623	75,125,512	2,147,077	2,147,077
9	82,671	54,652	54,652	188,061	88,520	88,520	2,543,732	347,622	347,622	45,841,452	1,580,144	1,580,144
10	64,155	43,387	43,387	142,415	70,483	70,483	1,728,151	267,570	267,570	27,972,371	1,162,909	1,162,909
11	49,787	34,444	34,444	107,849	56,121	56,121	1,174,064	205,953	205,953	17,068,690	855,845	855,845
12	38,636	27,344	27,344	81,672	44,686	44,686	797,631	158,526	158,526	10,415,284	629,860	629,860
13	29,983	21,708	24,011	61,849	35,580	35,580	541,891	122,020	122,020	6,355,387	463,546	463,546
14	23,268	17,234	20,678	46,837	28,330	28,330	368,148	93,921	93,921	3,878,046	341,148	341,148
15	18,057	13,682	17,345	35,469	22,558	24,997	250,111	72,292	72,292	2,366,377	251,068	251,068
16	14,012	10,862	14,012	26,860	17,961	21,664	169,919	55,645	55,645	1,443,959	184,774	184,774
17	10,874	8,623	10,874	20,341	14,301	18,331	115,439	42,831	42,831	881,101	135,985	135,985
18	8,439	6,845	8,439	15,404	11,387	14,998	78,427	32,967	32,967	537,646	100,078	100,078
19	6,549	5,434	6,549	11,665	9,067	11,665	53,281	25,376	25,376	328,071	73,653	73,653
20	5,082	4,314	5,082	8,834	7,219	8,834	36,198	19,532	21,869	200,188	54,205	54,205
21	3,944	3,425	3,944	6,690	5,748	6,690	24,592	15,034	18,363	122,154	39,892	39,892
22	3,061	2,719	3,061	5,066	4,577	5,066	16,707	11,572	14,857	74,538	29,359	29,359
23	2,375	2,159	2,375	3,836	3,644	3,836	11,350	8,907	11,350	45,483	21,607	24,602
24	1,843	1,714	1,843	2,905	2,902	2,905	7,711	6,856	7,711	27,754	15,901	19,846
25	1,430	1,360	1,430	2,200	2,311	2,200	5,239	5,277	5,239	16,935	11,703	15,090
26	1,110	1,080	1,110	1,666	1,840	1,666	3,559	4,062	3,559	10,334	8,613	10,334
27	861	857	861	1,262	1,465	1,262	2,418	3,127	2,418	6,306	6,338	6,306
28	668	681	668	955	1,166	955	1,643	2,407	1,643	3,848	4,665	3,848
29	519	540	519	724	929	724	1,116	1,852	1,116	2,348	3,433	2,348
30	403	429	403	548	739	548	758	1,426	758	1,433	2,527	1,433
31	312	341	312	415	589	415	515	1,097	515	874	1,859	874
32	242	270	242	314	469	314	350	845	350	533	1,368	533

Notes:
All values are in acre-feet per month
eq. = equation
ppt = parts per thousand
Rt 35 = Route 35

Table 5-5
Colorado Inflows and Resulting Salinities in the Design Areas

Inflow Criteria	Flow (ac-ft / 30 days)	Salinity (ppt) over Design Area		
		Delta	Shell Island/Tripod	Mad Island
Level 4	90,000	15	16.3	18.7
Level 3	50,000	20	21.3	23.3
Level 2	29,000	24.1	25.2	26.8
Level 1	20,000	26.9	27.8	28.8
Threshold	12,000	30.9	31.4	31.6

Notes:
ac-ft = acre-feet
ppt = parts per thousand

Table 5-6
Lavaca/Garcitas Inflows and Resulting Salinities in the Design Area

Inflow Criteria	Flow (ac-ft / 30 days)	Salinity (ppt) over Design Area			
		North	**Rt. 35 Bridge**	**Gallinipper**	**Indian Point**
Base Wet	54,200	9	11.2	16.1	20
Base Average	29,400	11.7	13.9	18.5	22
Base Dry	15,100	14.6	16.8	21.6	25
Subsistence	1,400	25.1	26.7	28.5	30.1

Notes:
ac-ft = acre-feet
ppt = parts per thousand

Table 5-7
Colorado Annual Total and Seasonal Distribution of Freshwater Regime Components

Inflow Criteria	Flow (ac-ft / 30 days)	Annualized	Spring (38%)	Fall (27%)	Intervening (35%)
Level 4	90,000	1,080,000	410,400	291,600	378,000
Level 3	50,000	600,000	228,000	162,000	210,000
Level 2	29,000	348,000	132,240	93,960	121,800
Level 1	20,000	240,000	91,200	64,800	84,000

Note:

ac-ft = acre-feet

Table 5-8
Lavaca/Garcitas Annual Total and Seasonal Distribution of Freshwater Regime Components

Inflow Criteria	Flow (ac-ft / 30 days)	Annualized	Spring (45%)	Fall (32%)	Intervening (23%)
Base Wet	54,200	650,400	247,152	175,608	227,640
Base Average	29,400	352,800	134,064	95,256	123,480
Base Dry	15,100	181,200	68,856	48,924	63,420
Subsistence	1,400	16,800	6,384	4,536	5,880

Note:
ac-ft = acre-feet

Table 5-9
Colorado Changes in Inflow Targets Resulting from Modeling and Analysis Updates

Inflow Criteria	MBHE Study Design Inflow	Annualized	Seasonal Spring (38%)	Seasonal Fall (27%)	Seasonal Intervening (35%)
Level 4	95,000	1,140,000	433,200	307,800	399,000
Level 3	54,000	648,000	246,240	174,960	226,800
Level 2	37,000	444,000	168,720	119,880	155,400
Level 1	25,000	300,000	114,000	81,000	105,000

Inflow Criteria	TxBLEND 8.5 year Design Inflow	Annualized	Seasonal Spring (38%)	Seasonal Fall (27%)	Seasonal Intervening (35%)
Level 4	92,000	1,104,000	419,520	298,080	386,400
Level 3	51,000	612,000	232,560	165,240	214,200
Level 2	33,000	396,000	150,480	106,920	138,600
Level 1	24,000	288,000	109,440	77,760	100,800

Inflow Criteria	TxBLEND 1986-2014 Design Inflow	Annualized	Seasonal Spring (38%)	Seasonal Fall (27%)	Seasonal Intervening (35%)
Level 4	90,000	1,080,000	410,400	291,600	378,000
Level 3	50,000	600,000	228,000	162,000	210,000
Level 2	29,000	348,000	132,240	93,960	121,800
Level 1	20,000	240,000	91,200	64,800	84,000

Notes:

All values are in acre-feet.

MBHE = Matagorda Bay Health Evaluation

Table 5-10
Lavaca/Garcitas Changes in Inflow Targets Resulting From Modeling and Analysis Updates

Inflow Criteria	BBEST Study Design Inflow	Annualized	Seasonal Spring (45%)	Seasonal Fall (32%)	Intervening (23%)
Base Wet	41,400	496,800	223,560	158,976	114,264
Base Average	23,700	284,400	127,980	91,008	65,412
Base Dry	10,200	122,400	55,080	39,168	28,152
Subsistence	2,500	30,000	13,500	9,600	6,900

Inflow Criteria	TxBLEND 1986-2014 Design Inflow	Annualized	Seasonal Spring (45%)	Seasonal Fall (32%)	Intervening (23%)
Base Wet	54,200	650,400	292,680	208,128	149,592
Base Average	29,400	352,800	158,760	112,896	81,144
Base Dry	15,100	181,200	81,540	57,984	41,676
Subsistence	1,400	16,800	7,560	5,376	3,864

Notes:

All values are in acre-feet.

BBEST = Basin and Bay Expert Science Team

Table 5-11
Historical Return Flows by Subwatershed

Year	Caney	Ikaustin	Boggy	Oyster	Colo	Tres	Turtle	Caran	Keller	Cox	Lavdelt	Gar	Choc	Powder	Guadalupe	Hynes
1977	3,899	3,593	1,527	6,198	0	762	2,845	5,620	1,071	4,477	10,535	2,872	3,235	3,943	120, 266	0
1978	6,034	5,616	1,681	6,903	0	825	2,748	6,545	1,247	2,470	5,065	3,269	1,886	5,567	127,852	0
1979	7,697	6,084	1,773	6,383	0	854	2,965	6,621	1,261	1,763	5,220	3,727	3,601	3,609	105,242	0
1980	8,401	6,598	1,926	7,620	0	1,798	3,325	7,340	1,398	1,708	2,262	3,973	3,873	4,324	96,993	0
1981	8,956	7,002	1,926	8,056	0	2,501	2,595	7,212	1,374	2,041	1,677	4,003	3,259	3,359	73,977	0
1982	9,981	6,941	1,742	7,045	0	2,652	2,774	6,031	1,149	1,669	1,277	2,933	2,466	3,721	74,975	0
1983	9,699	4,745	1,038	5,063	0	2,196	1,679	3,876	738	1,669	1,125	1,926	2,284	2,533	43,922	0
1984	9,342	6,417	1,528	7,198	0	3,599	2,536	5,363	1,022	1,498	1,100	2,658	2,229	3,539	55,771	0
1985	7,360	5,417	1,223	5,513	0	2,846	2,042	4,362	831	2,767	1,064	2,168	2,195	2,517	40,853	0
1986	6,803	5,175	1,009	6,141	0	3,086	2,171	4,210	802	3,044	853	1,864	2,410	3,332	35,314	0
1987	6,719	6,528	1,560	5,226	0	2,627	2,410	3,824	728	3,589	914	1,926	2,097	3,053	35,113	0
1988	7,037	5,362	1,070	3,272	977	2,783	2,079	5,085	969	2,349	762	3,424	2,077	2,416	22,398	0
1989	6,513	5,538	1,070	2,660	885	2,326	1,922	4,701	895	2,343	852	3,057	2,101	2,080	23,992	0
1990	5,561	5,789	1,437	3,088	1,006	2,416	2,078	4,803	915	2,128	797	3,149	2,100	2,140	25,168	0
1991	5,772	6,392	1,560	3,088	1,068	2,436	2,229	4,572	871	2,129	945	2,997	2,103	2,202	20,071	0
1992	5,032	6,504	1,712	3,118	945	2,078	2,198	4,701	895	2,104	853	3,180	1,983	1,774	16,777	0
1993	4,774	3,854	1,528	2,721	914	1,588	1,707	3,725	709	1,161	760	2,599	1,826	1,468	19,934	0
1994	6,009	6,034	2,109	3,394	1,221	1,774	2,228	4,495	856	6,546	884	3,180	1,952	1,988	25,875	0
1995	6,645	6,217	1,649	3,211	365	672	6,666	8,810	1,678	7,949	703	5,045	3,662	3,211	25,783	0
1996	6,691	5,818	1,375	3,211	366	919	5,321	8,630	1,644	8,114	581	4,801	5,617	1,957	25,788	0
1997	7,895	6,211	1,221	3,975	365	1,284	5,015	7,372	1,404	7,696	1,161	4,373	12,623	3,333	14,218	0
1998	7,045	6,927	1,221	3,699	365	1,465	3,333	7,397	1,409	7,448	1,600	3,975	12,890	3,211	13,431	0
1999	6,965	6,516	1,008	3,058	365	1,285	1,682	6,422	1,223	7,784	822	3,454	12,112	1,988	13,761	0
2000	10,825	6,382	826	4,128	2,564	2,202	2,813	14,149	2,695	25,072	764	856	1,130	1,070	26,402	0
2001	11,007	5,526	2,770	3,333	8,984	1,774	2,536	14,330	2,729	26,726	1,004	885	1,372	886	26,337	0
2002	10,763	5,584	4,838	3,241	6,588	1,774	2,567	13,475	2,567	23,561	947	764	1,524	886	26,245	0
2003	10,603	5,488	3,983	3,945	6,424	2,080	3,115	13,448	2,561	18,812	883	550	2,042	1,926	23,274	0
2004	13,254	5,628	2,244	3,486	6,711	1,926	2,964	18,000	3,429	27,996	1,251	275	1,887	2,110	27,033	0
2005	11,497	5,096	2,000	2,966	5,909	1,621	2,442	14,412	2,745	23,742	1,183	642	1,736	2,447	24,286	0
2006	7,995	4,807	976	2,447	6,312	1,285	2,047	10,244	1,951	28,207	1,037	214	1,947	2,080	25,028	0
2007	5,367	5,146	1,102	2,202	6,433	1,193	1,955	6,370	1,213	25,660	916	0	2,103	1,468	29,873	0
2008	248	2,655	367	0	3,837	0	428	52	10	16,590	794	0	1,312	0	25,911	0
2009	62	2,650	673	0	4,834	0	518	51	10	17,621	640	0	1,368	0	24,734	0
2010	882	2,824	0	4,151	6,528	0	93	26	5	0	74,818	0	10,061	0	103,401	0
2011	807	2,754	0	3,612	3,806	0	0	0	0	30	39,671	0	7,788	0	56,554	0
2012	381	2,098	0	3,431	5,660	31	0	0	0	0	69,637	0	7,499	0	74,220	0
2013	183	0	0	0	0	0	0	0	0	0	0	0	7,840	0	0	0
2014	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0

Notes:

All values are in acre-feet per year

Colo = Colorado River

Choc = Chocolate

Gar = Garcitas

Lavdelt = Lavaca River Delta

Ikaustin = Lake Austin

Tres = Tres Palacios

Table 6-1
Colorado River Achievement Frequencies

Inflow Criteria	POR Occurrence
Level 4	24%
Level 3	50%
Level 2	82%
Level 1	89%
Threshold	100%

Note:
POR = period of record

Table 6-2
Lavaca River and Garcitas Creek Achievement Frequencies

Inflow Criteria	POR Occurrence
Level 4	32%
Level 3	42%
Level 2	66%
Level 1	100%

Note
POR = period of record

FIGURES

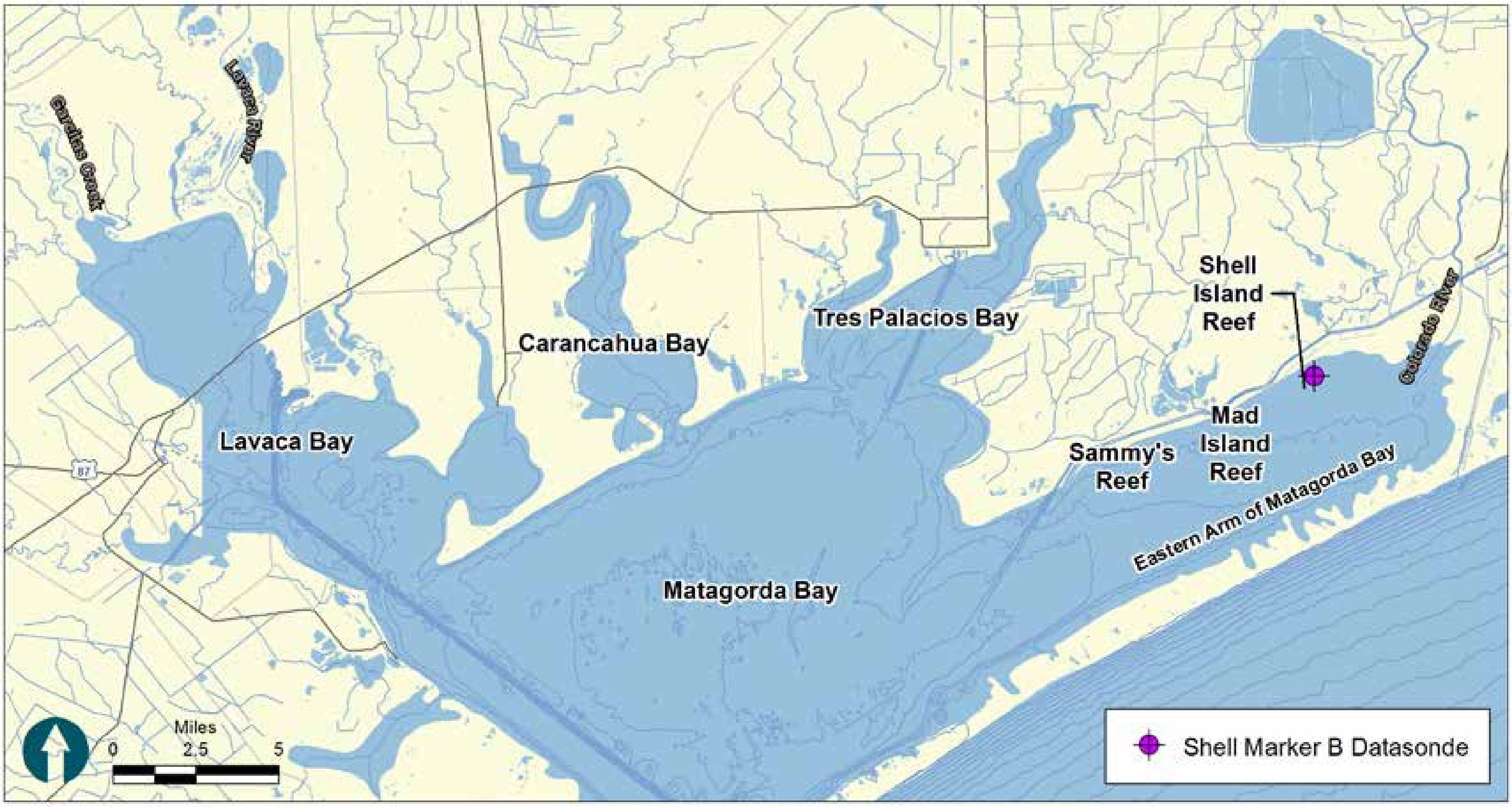

Figure 1-1
The Matagorda Lavaca Bay System
Freshwater Inflow Standards and Ecological Response
TWDB

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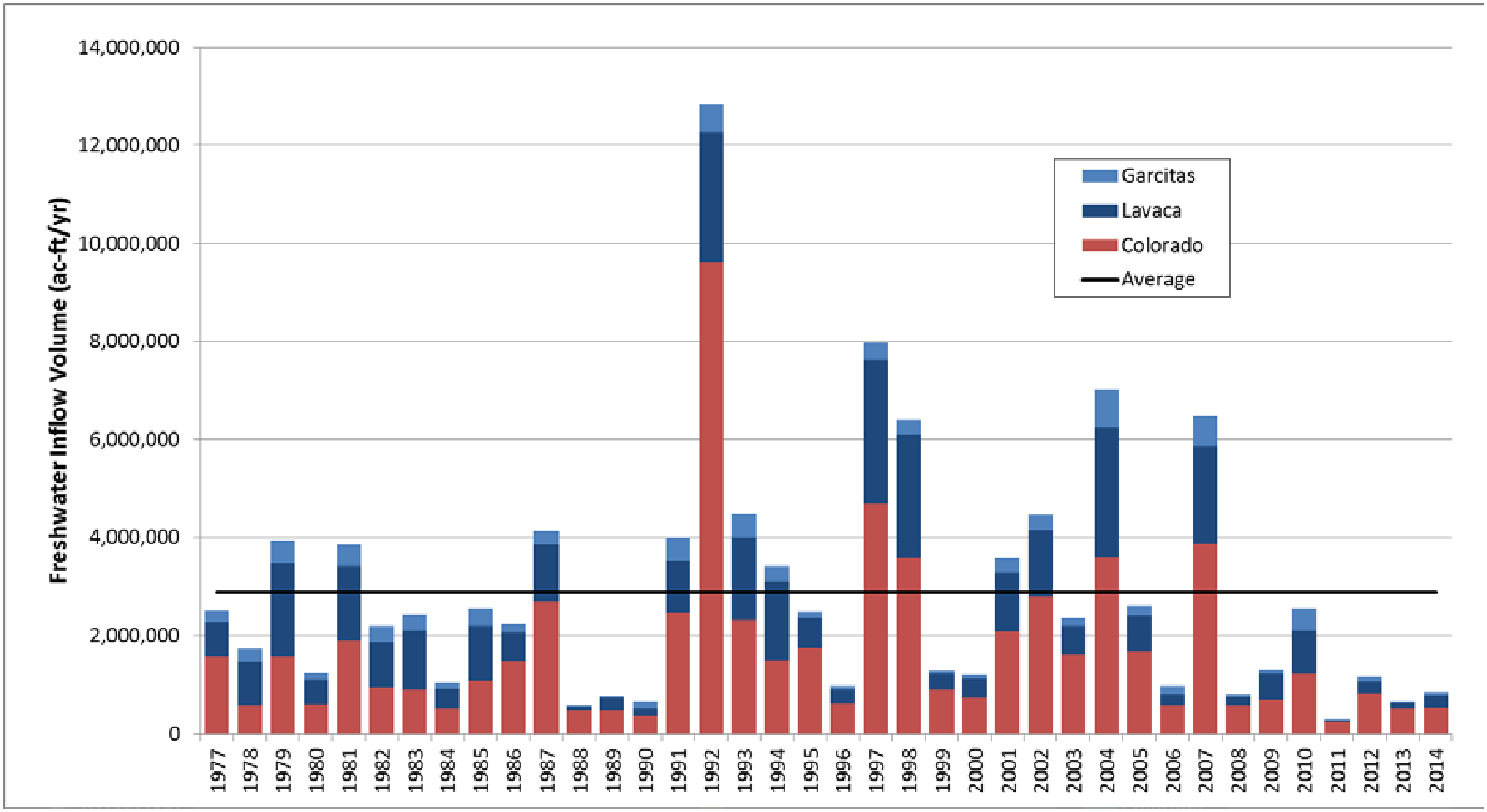

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Figure 1-2
Annual Inflows Corresponding to the Locations of TCEQ Freshwater Inflow Standards
Freshwater Inflow Standards and Ecological Response
TWDB

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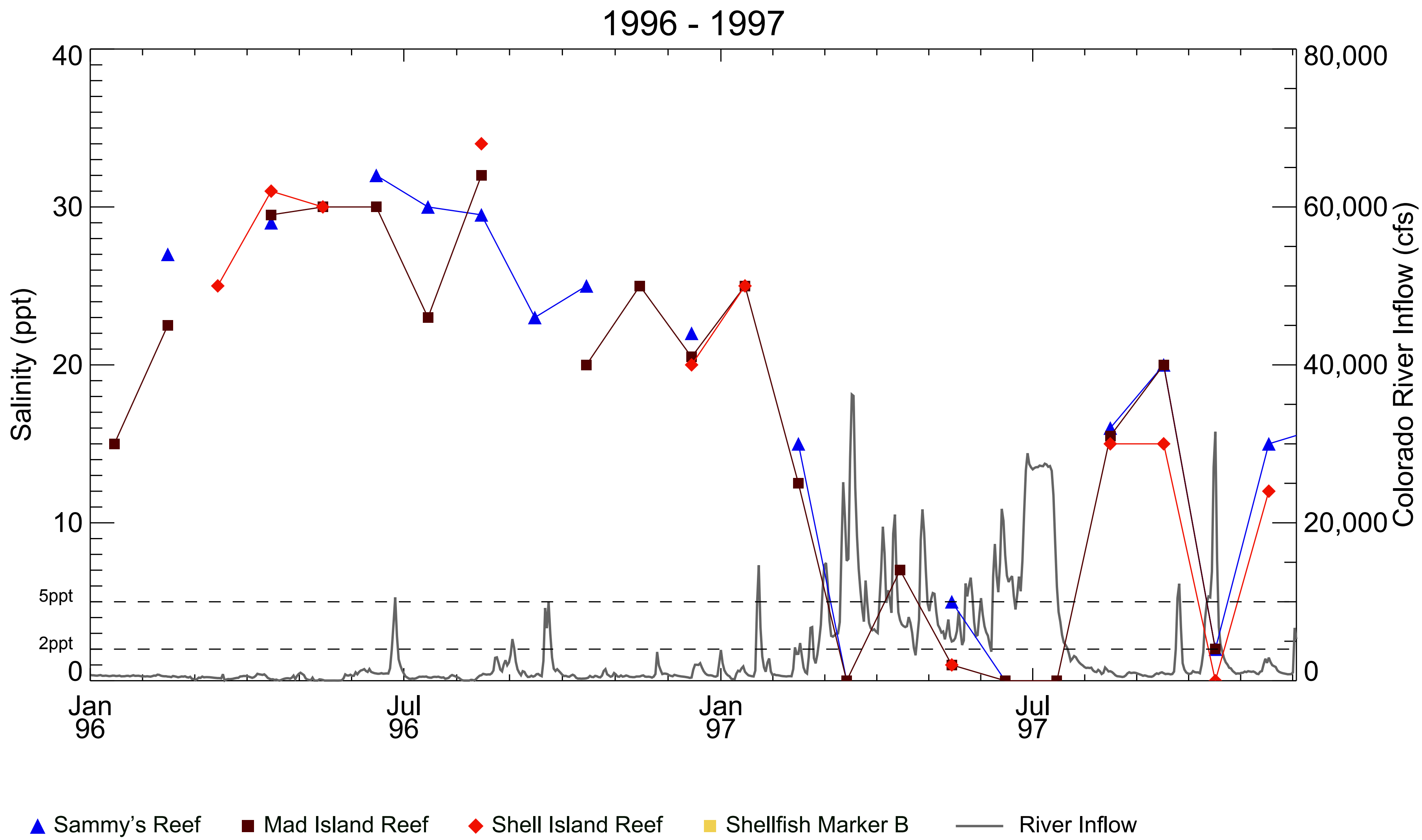

Figure 1-3a

Salinity and Inflow Time Series for the Eastern Arm of Matagorda Bay

Monthly average salinities calculated from TPWD and OysterSentinel data are plotted at 15th of each month.
Sonde salinity data downloaded from waterquality.lcra.org are plotted at dates where salinity is measured.

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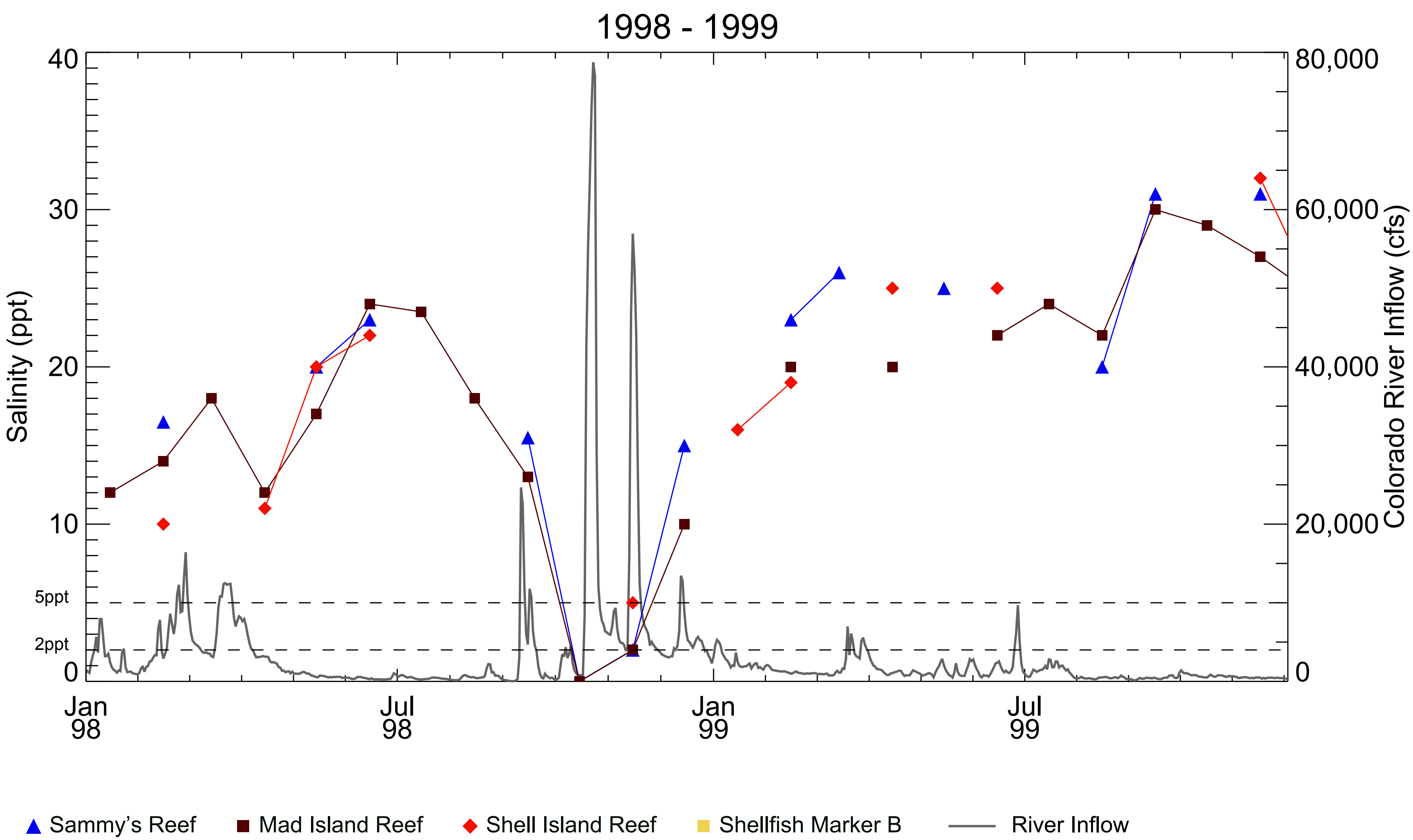

Figure 1-3b

Salinity and Inflow Time Series for the Eastern Arm of Matagorda Bay

Monthly average salinities calculated from TPWD and OysterSentinel data are plotted at 15th of each month.
Sonde salinity data downloaded from waterquality.lcra.org are plotted at dates where salinity is measured.

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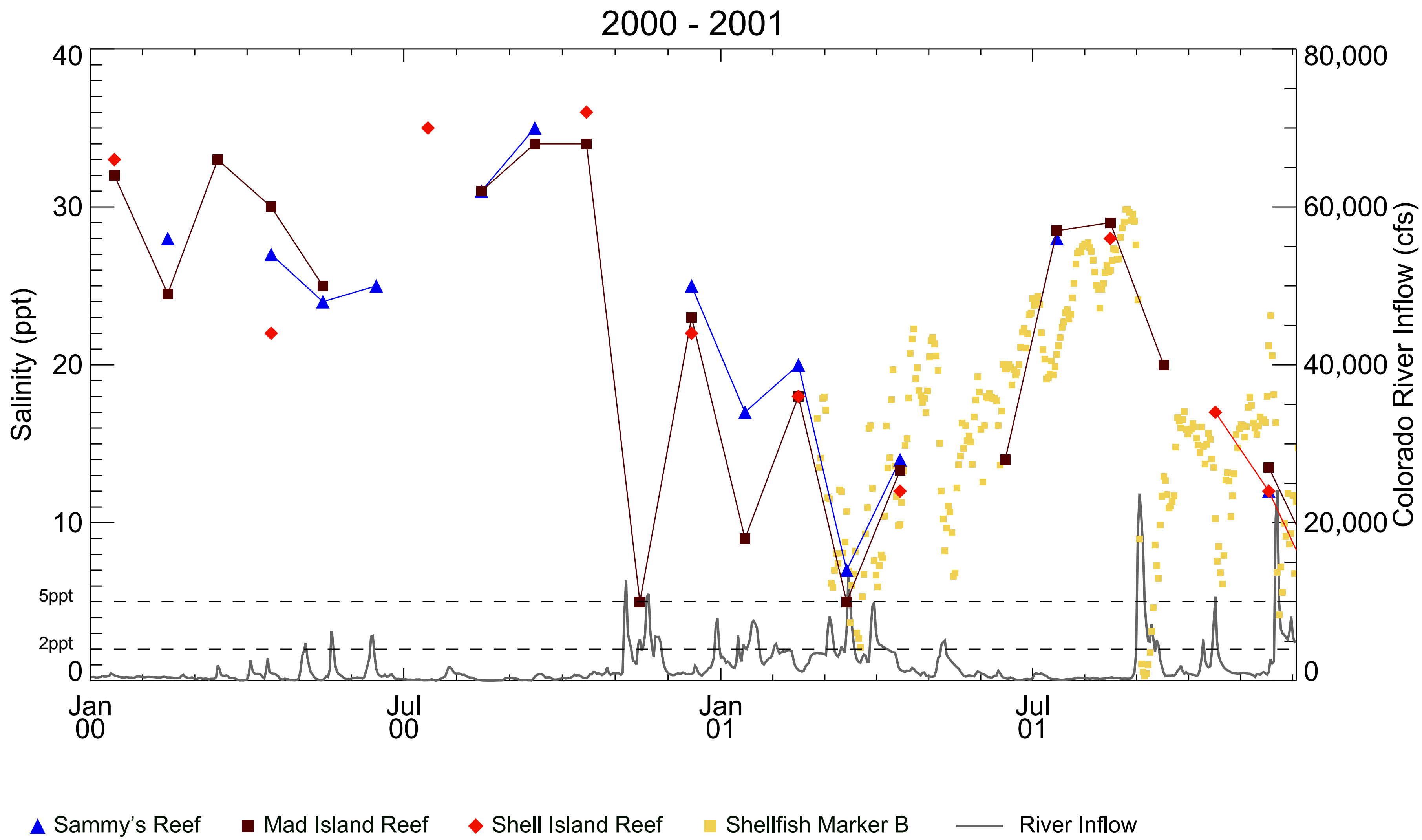

Figure 1-3c

Salinity and Inflow Time Series for the Eastern Arm of Matagorda Bay

Monthly average salinities calculated from TPWD and OysterSentinel data are plotted at 15th of each month.
Sonde salinity data downloaded from waterquality.lcra.org are plotted at dates where salinity is measured.

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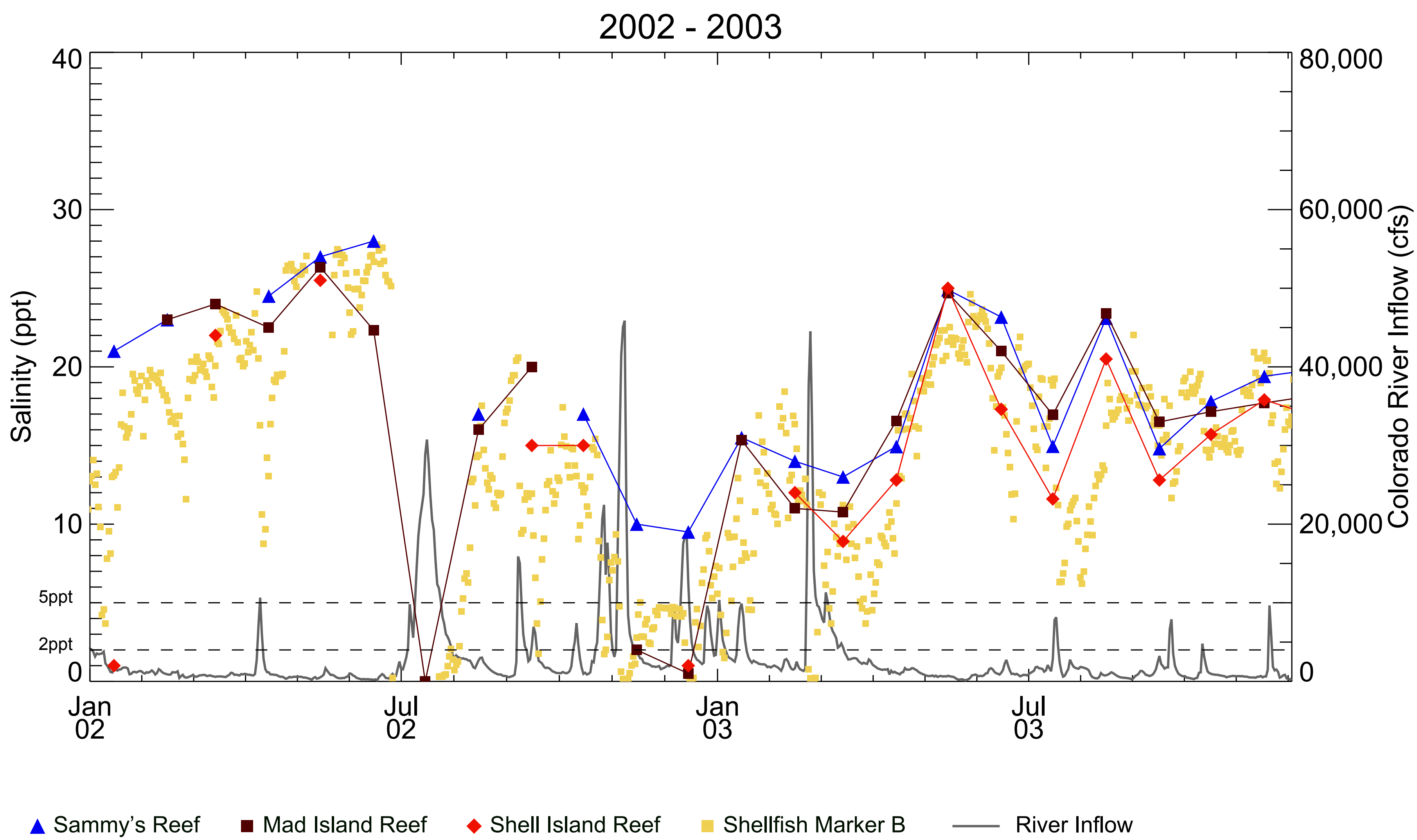

Figure 1-3d

Salinity and Inflow Time Series for the Eastern Arm of Matagorda Bay

Monthly average salinities calculated from TPWD and OysterSentinel data are plotted at 15th of each month.
Sonde salinity data downloaded from waterquality.lcra.org are plotted at dates where salinity is measured.

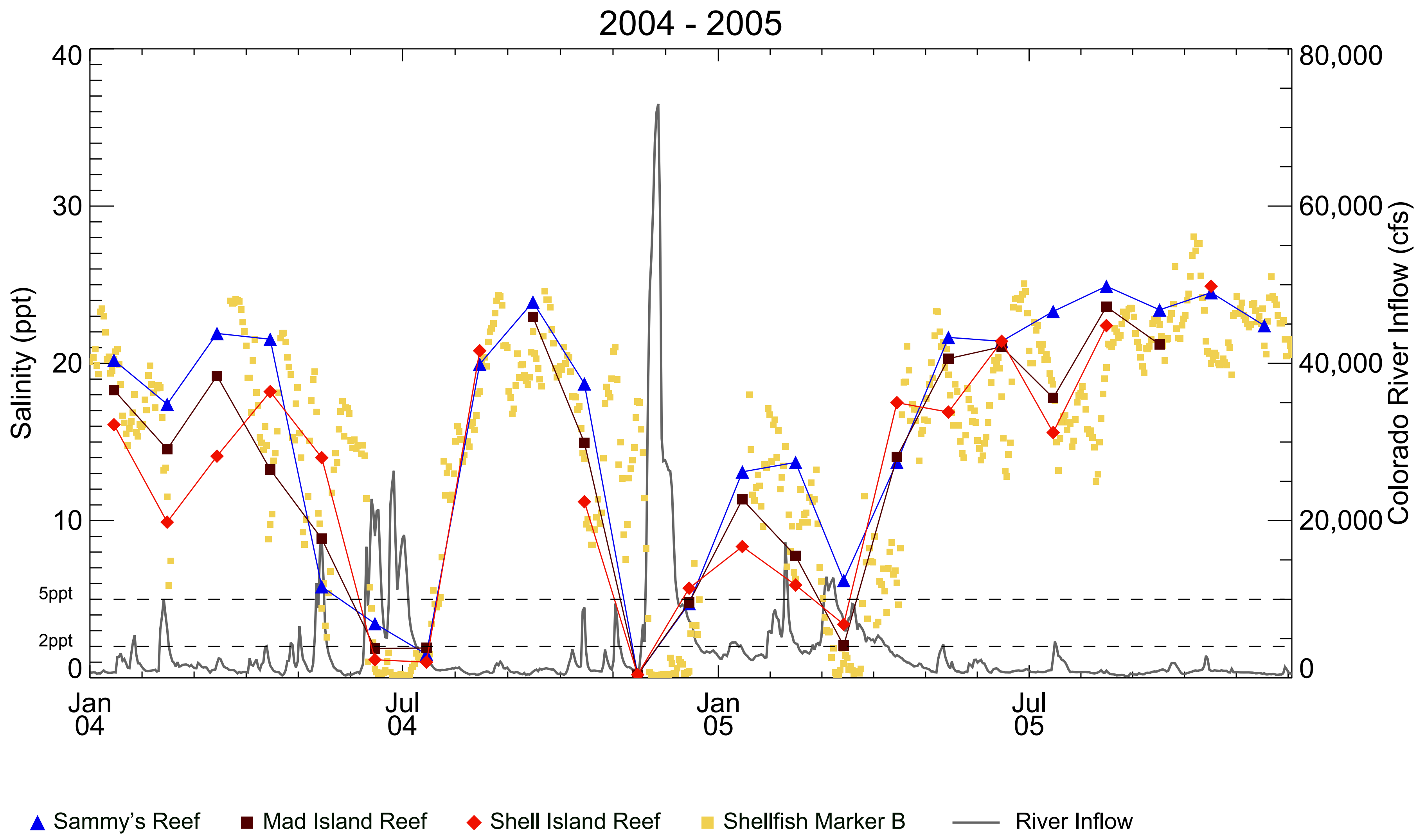

Figure 1-3e

Salinity and Inflow Time Series for the Eastern Arm of Matagorda Bay

Monthly average salinities calculated from TPWD and OysterSentinel data are plotted at 15th of each month.
Sonde salinity data downloaded from waterquality.lcra.org are plotted at dates where salinity is measured.

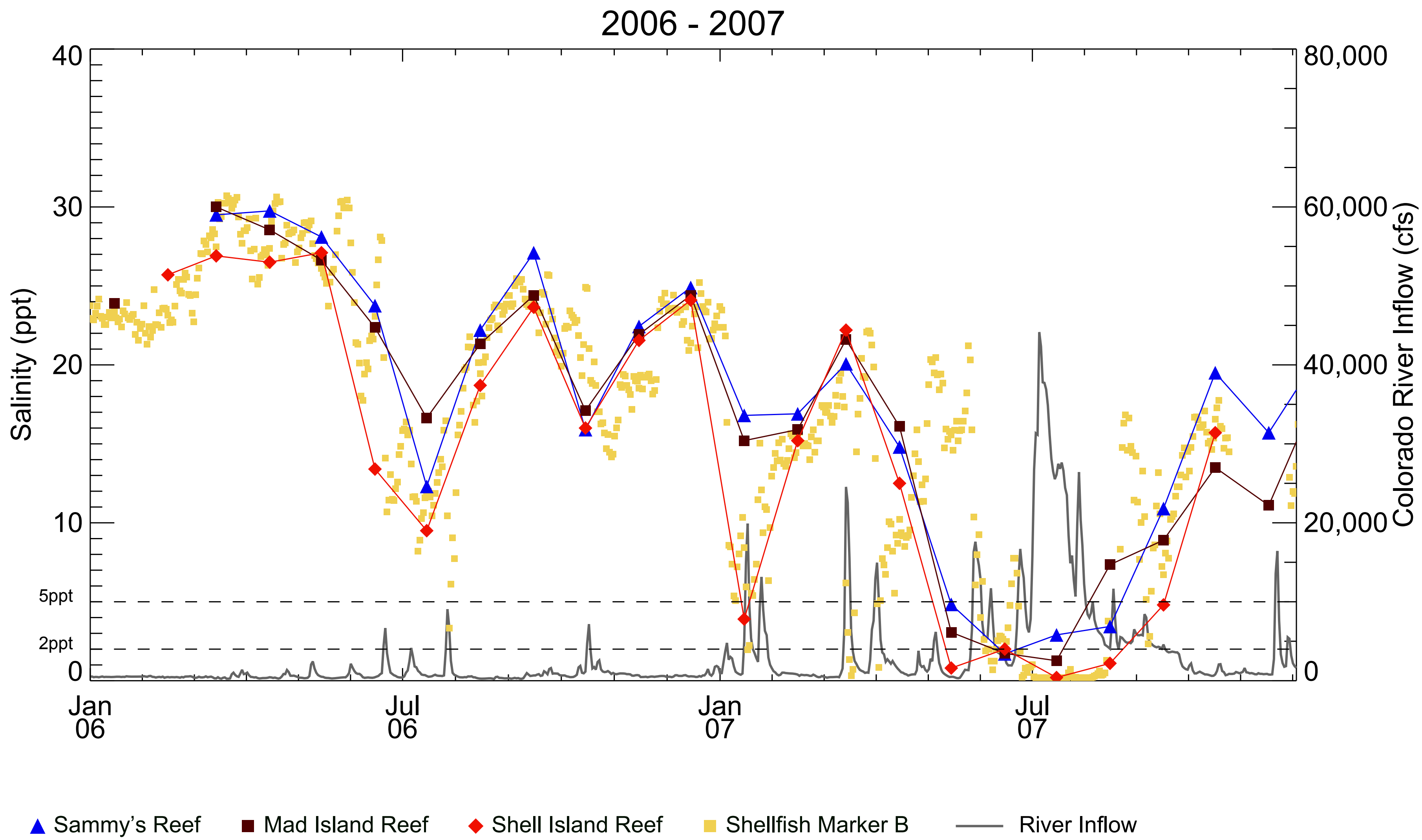

Figure 1-3f

Salinity and Inflow Time Series for the Eastern Arm of Matagorda Bay

Monthly average salinities calculated from TPWD and OysterSentinel data are plotted at 15th of each month.
Sonde salinity data downloaded from waterquality.lcra.org are plotted at dates where salinity is measured.

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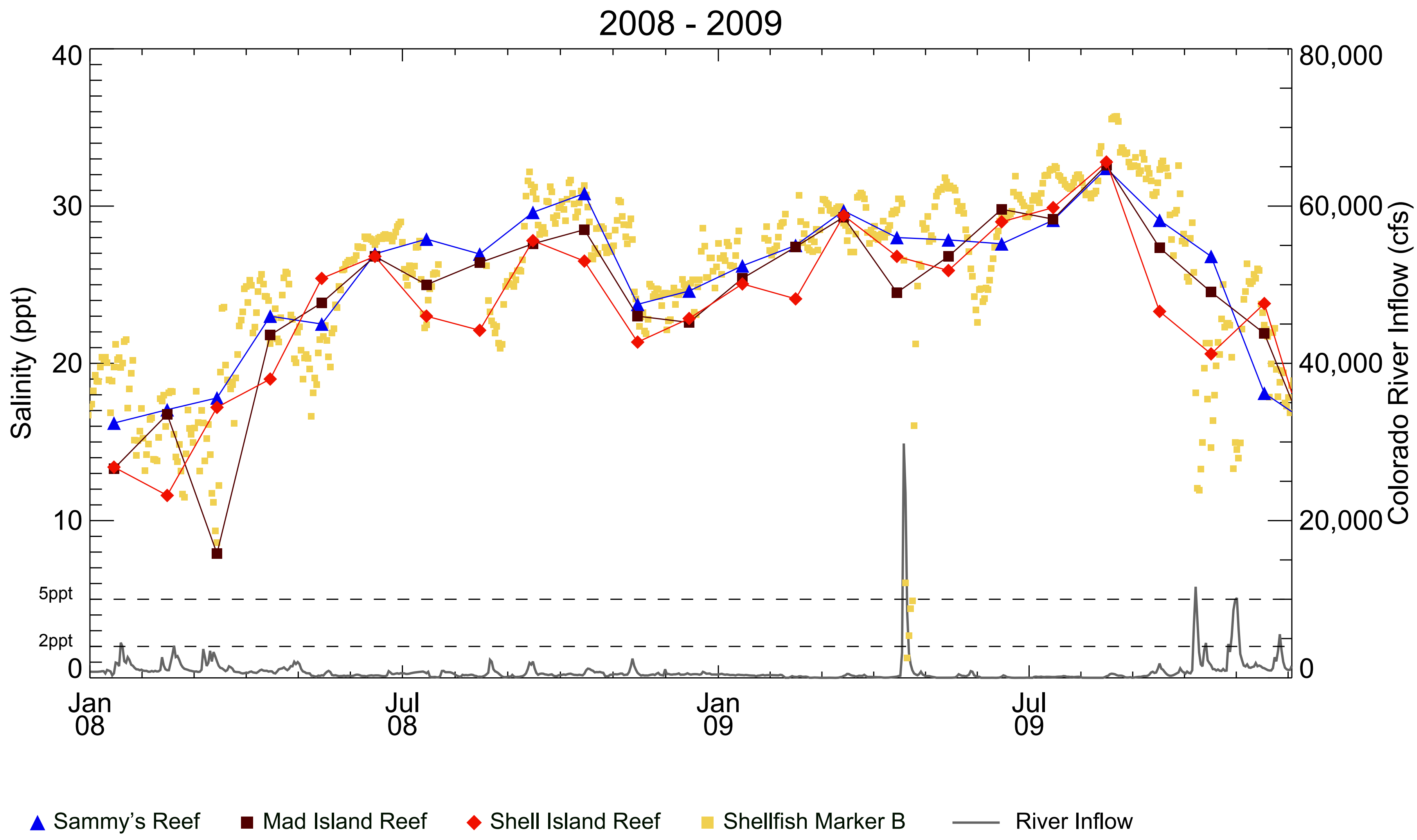

Figure 1-3g

Salinity and Inflow Time Series for the Eastern Arm of Matagorda Bay

Monthly average salinities calculated from TPWD and OysterSentinel data are plotted at 15th of each month.
Sonde salinity data downloaded from waterquality.lcra.org are plotted at dates where salinity is measured.

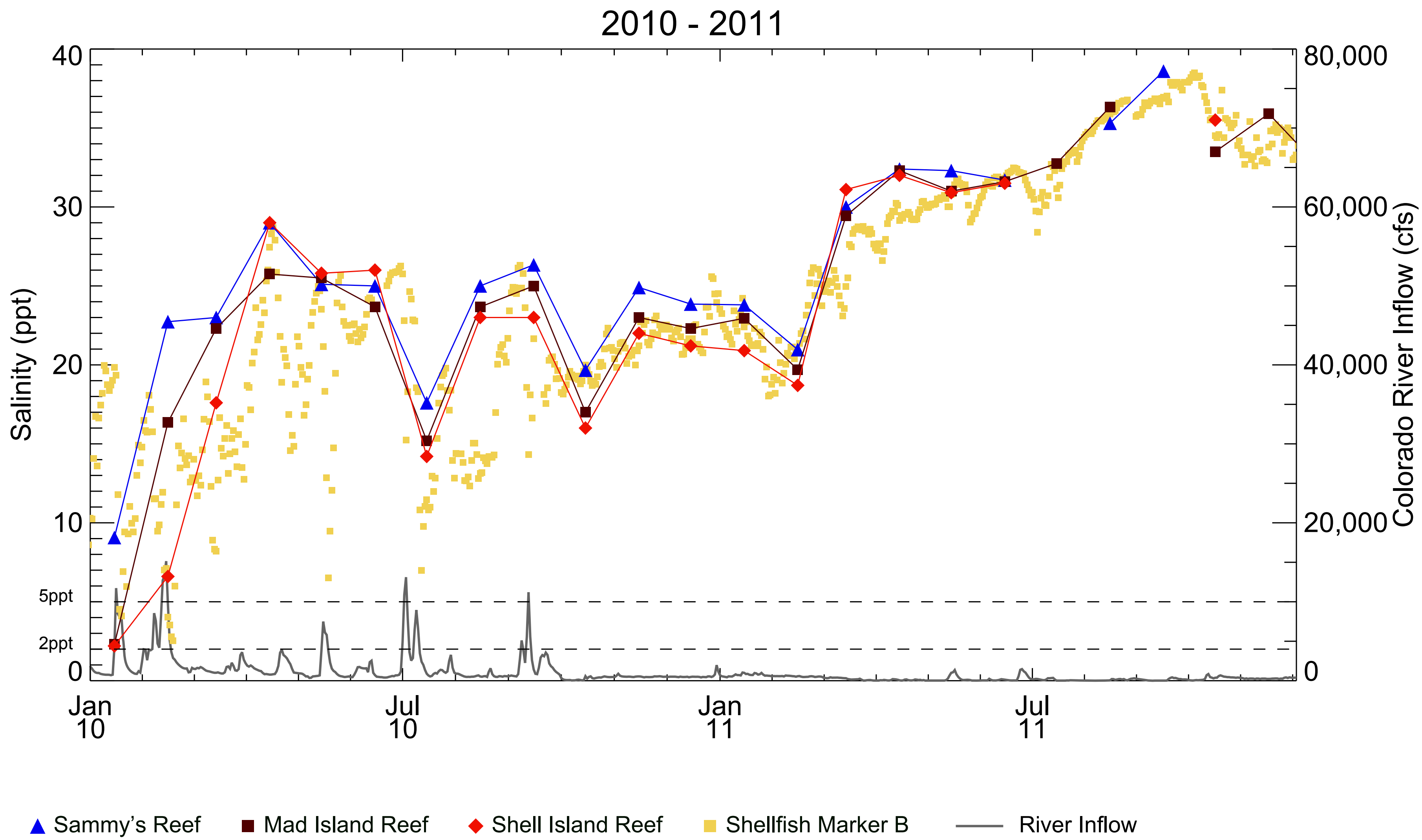

Figure 1-3h

Salinity and Inflow Time Series for the Eastern Arm of Matagorda Bay

Monthly average salinities calculated from TPWD and OysterSentinel data are plotted at 15th of each month. Sonde salinity data downloaded from waterquality.lcra.org are plotted at dates where salinity is measured.

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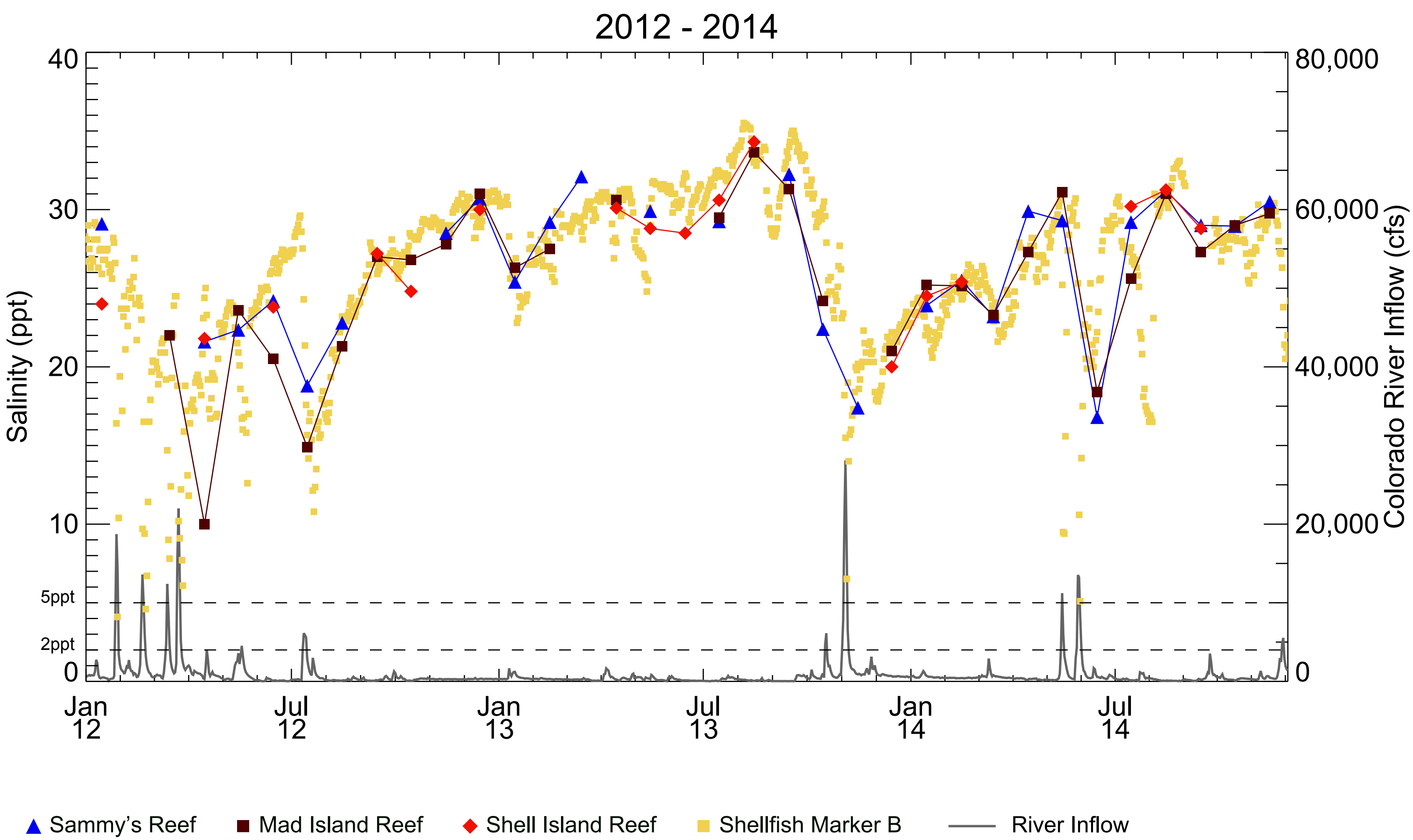
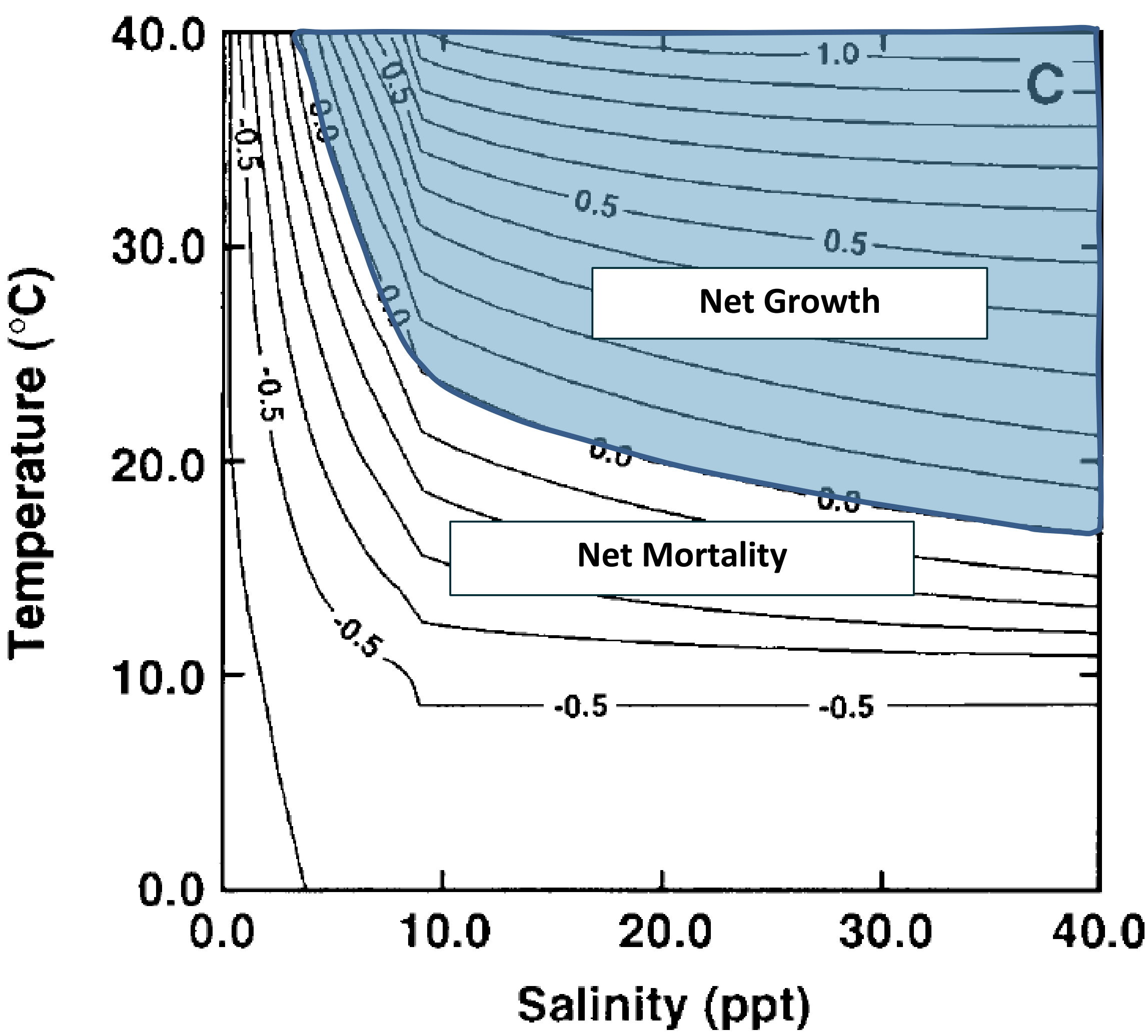


Figure 1-3i

Salinity and Inflow Time Series for the Eastern Arm of Matagorda Bay

Monthly average salinities calculated from TPWD and OysterSentinel data are plotted at 15th of each month. Sonde salinity data downloaded from waterquality.lcra.org are plotted at dates where salinity is measured.



Note: Adapted from Hoffmann et al. 1995; growth in units of day^{-1}

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Figure 2-1
Dermo Net Growth as a Function of Temperature and Salinity
Freshwater Inflow Standards and Ecological Response
TWDB

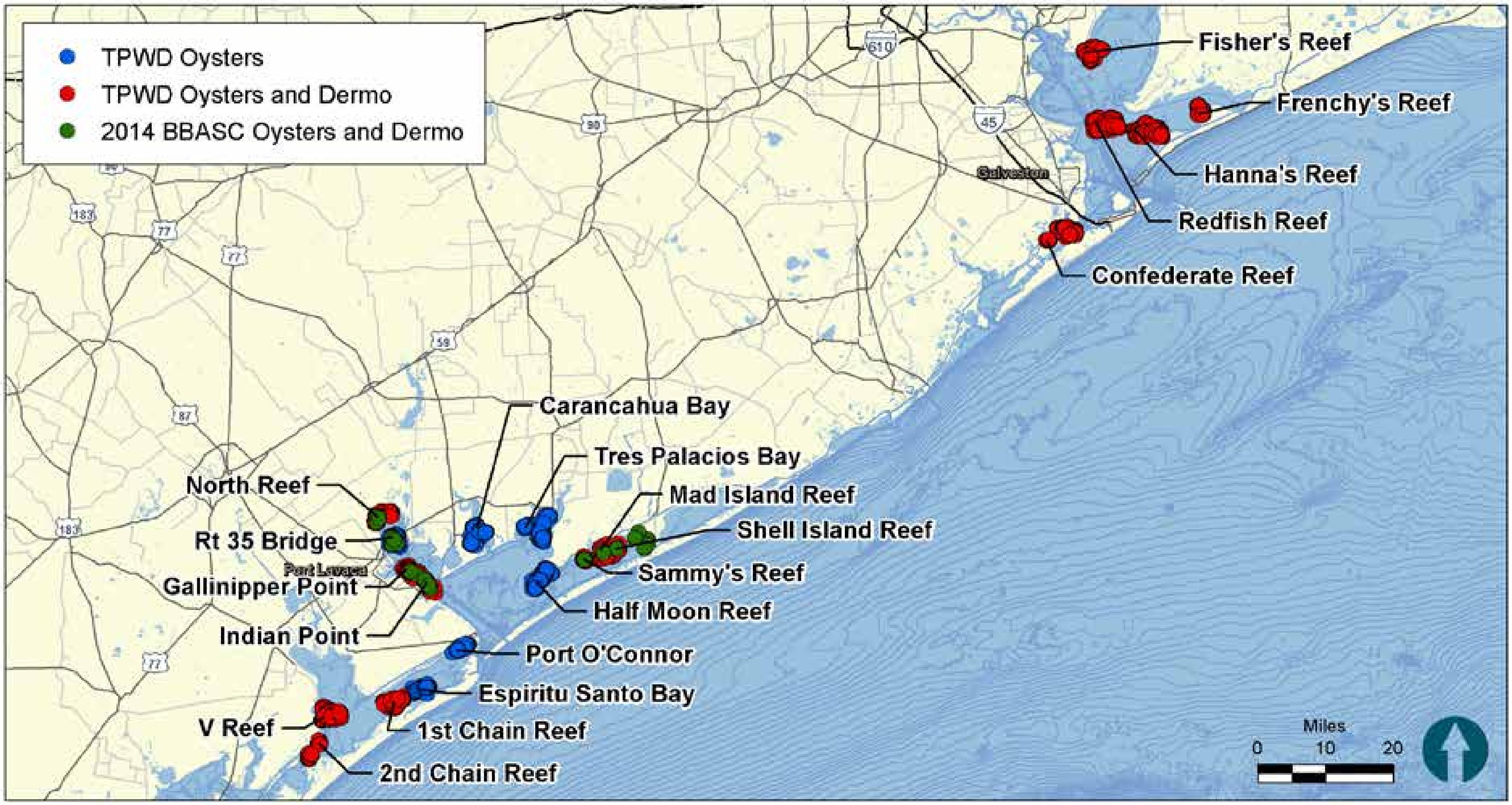

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Figure 2-2
Reef Locations in San Antonio, Matagorda, Lavaca, and Galveston Bays
Freshwater Inflow Standards and Ecological Response
TWDB

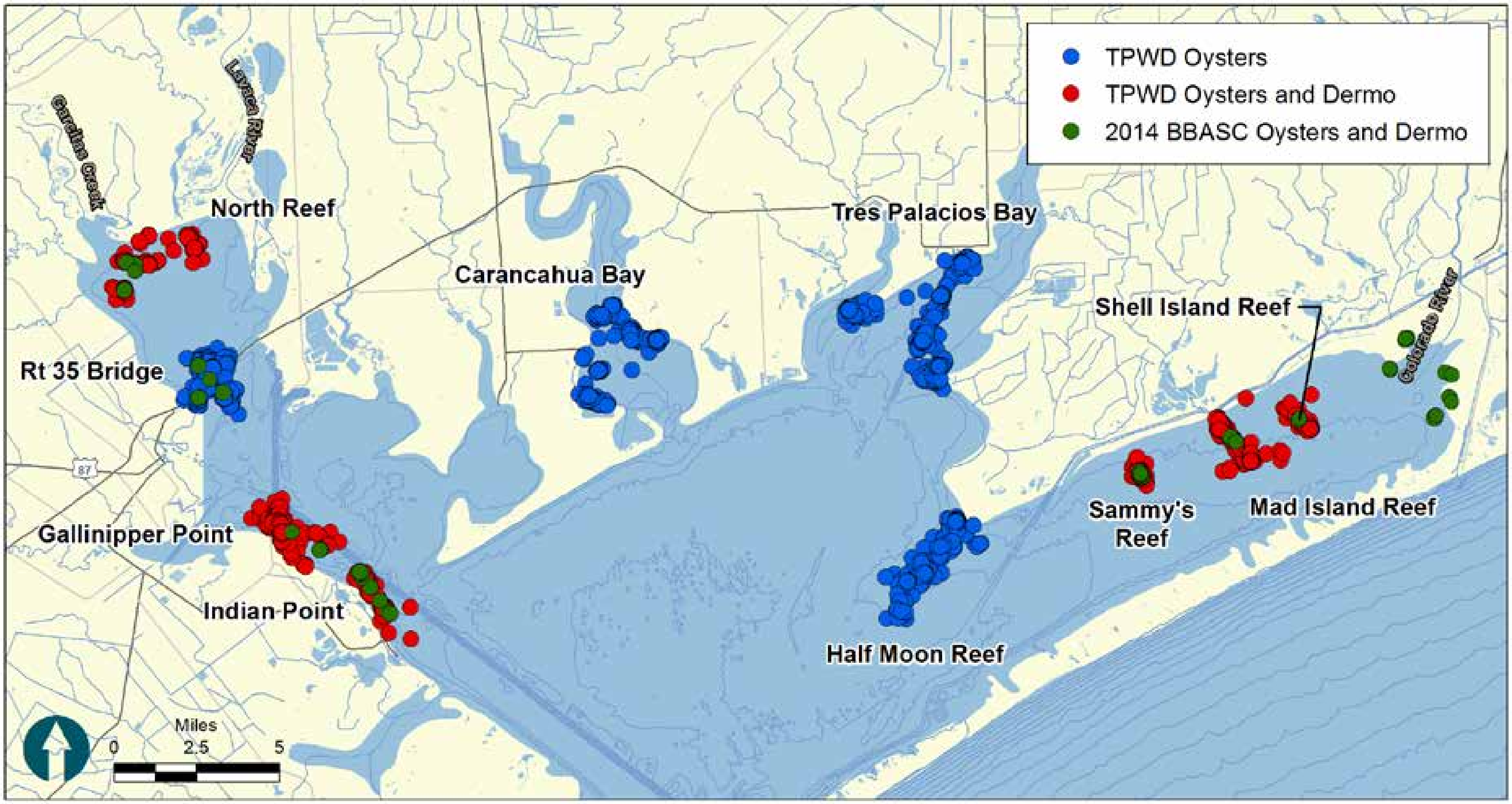

Figure 2-3
Reef Locations in the Matagorda and Lavaca Bay System
Freshwater Inflow Standards and Ecological Response
TWDB

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Figure 2-4
2014 Oyster Sampling Locations in Lavaca Bay
Freshwater Inflow Standards and Ecological Response
TWDB

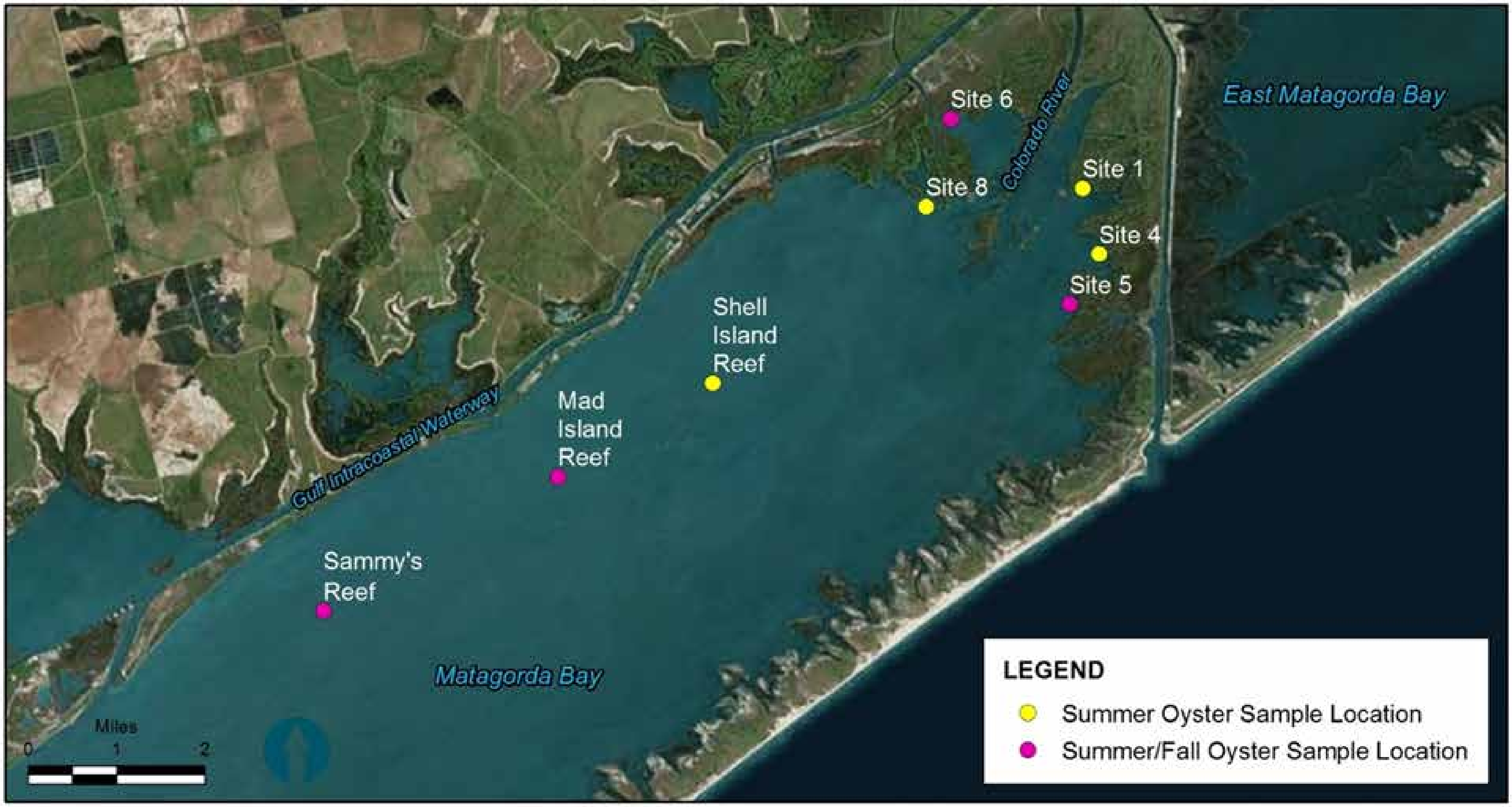

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Figure 2-5
2014 Oyster Sampling Locations in Matagorda Bay
Freshwater Inflow Standards and Ecological Response
TWDB

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Figure 2-6
Oyster Dredge with Live Oysters Collected for Dermo Analysis
Freshwater Inflow Standards and Ecological Response
TWDB

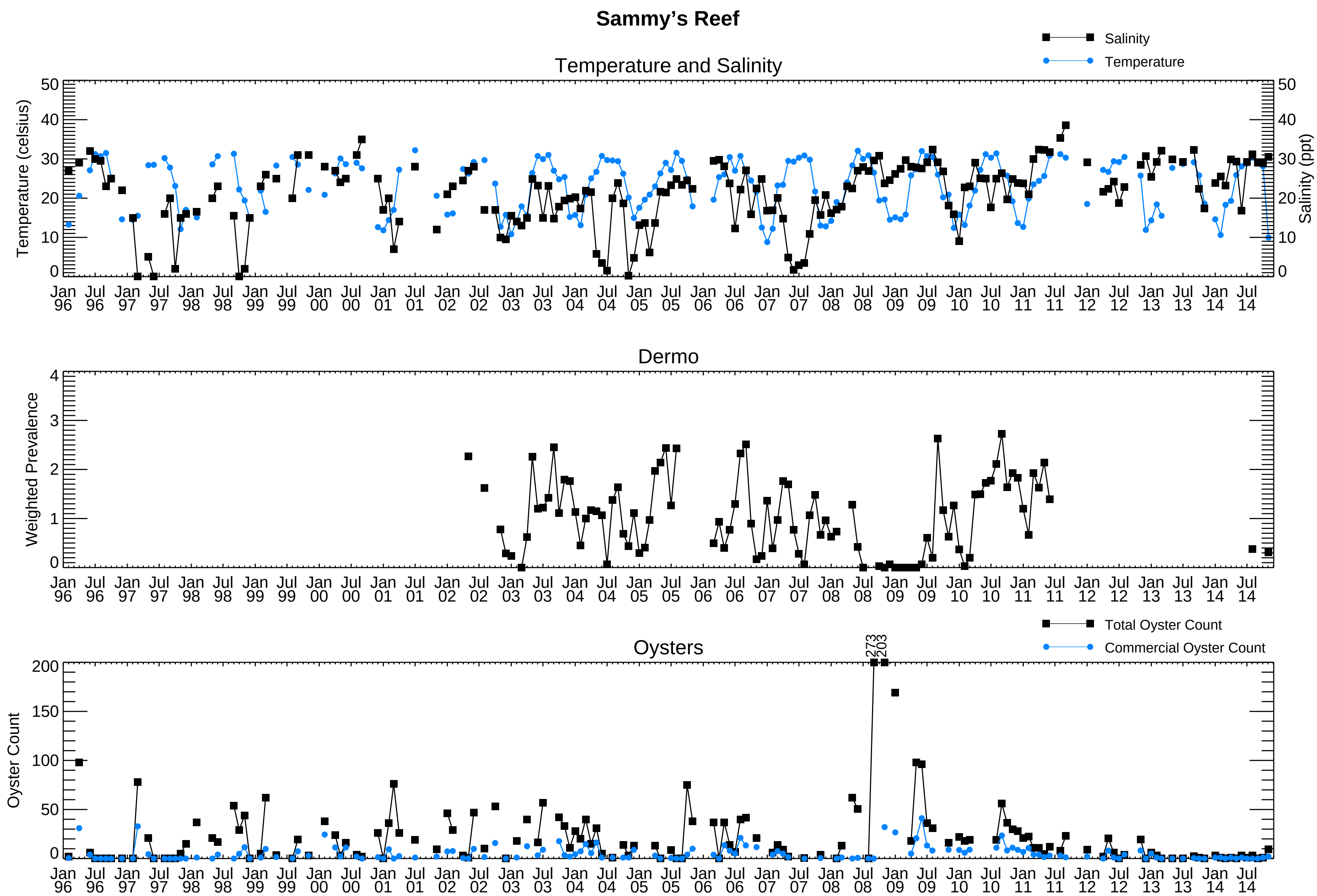

Figure 2-7a

Time Series of Monthly Temperature, Salinity, Dermo and Oyster Results for Sammy's Reef

Data source: TPWD; OysterSentinel.

Freshwater Inflow Standards and Ecological Response
TWDB

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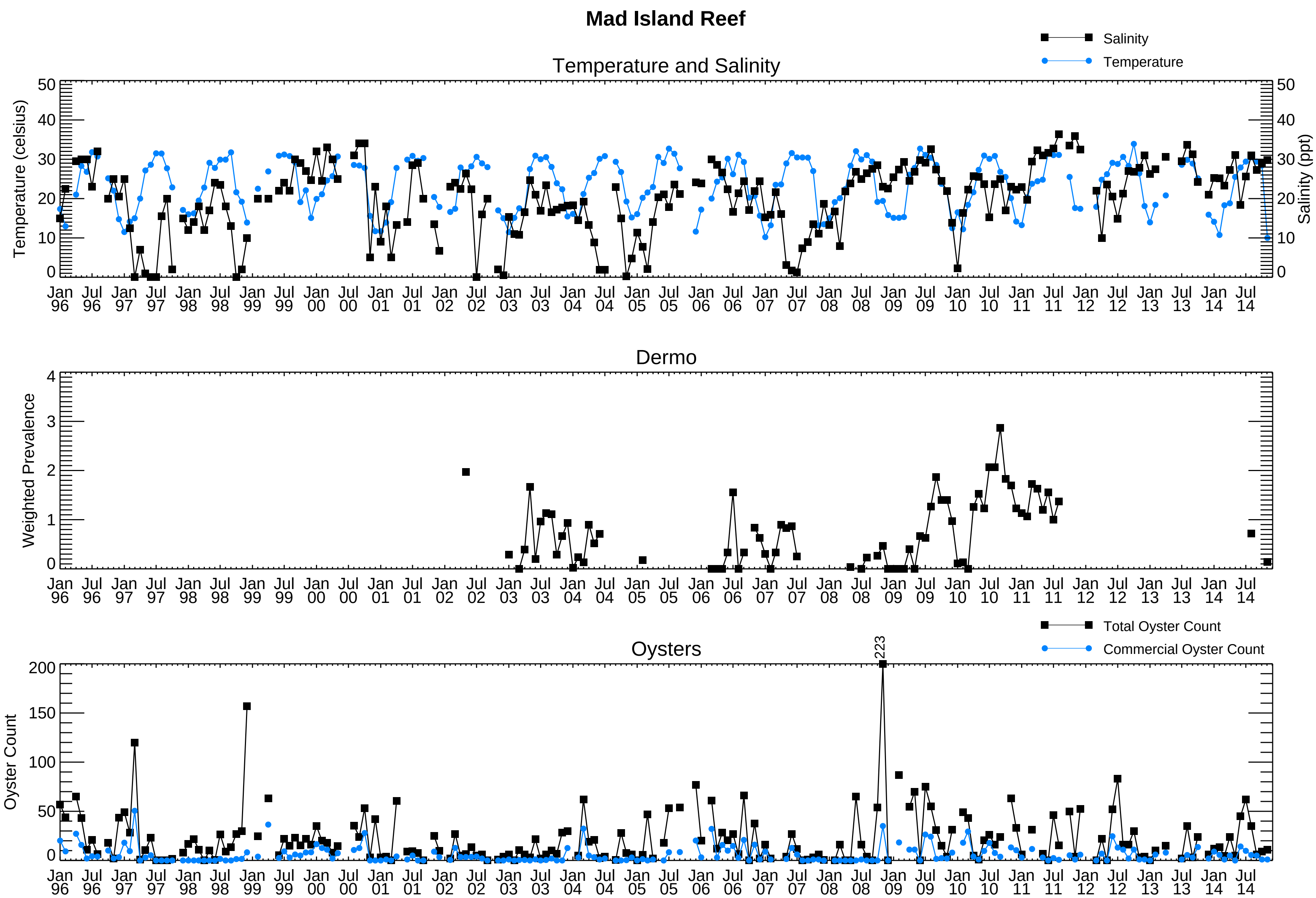

Figure 2-7b

Time Series of Monthly Temperature, Salinity, Dermo and Oyster Results for Mad Island Reef

Data source: TPWD; OysterSentinel.

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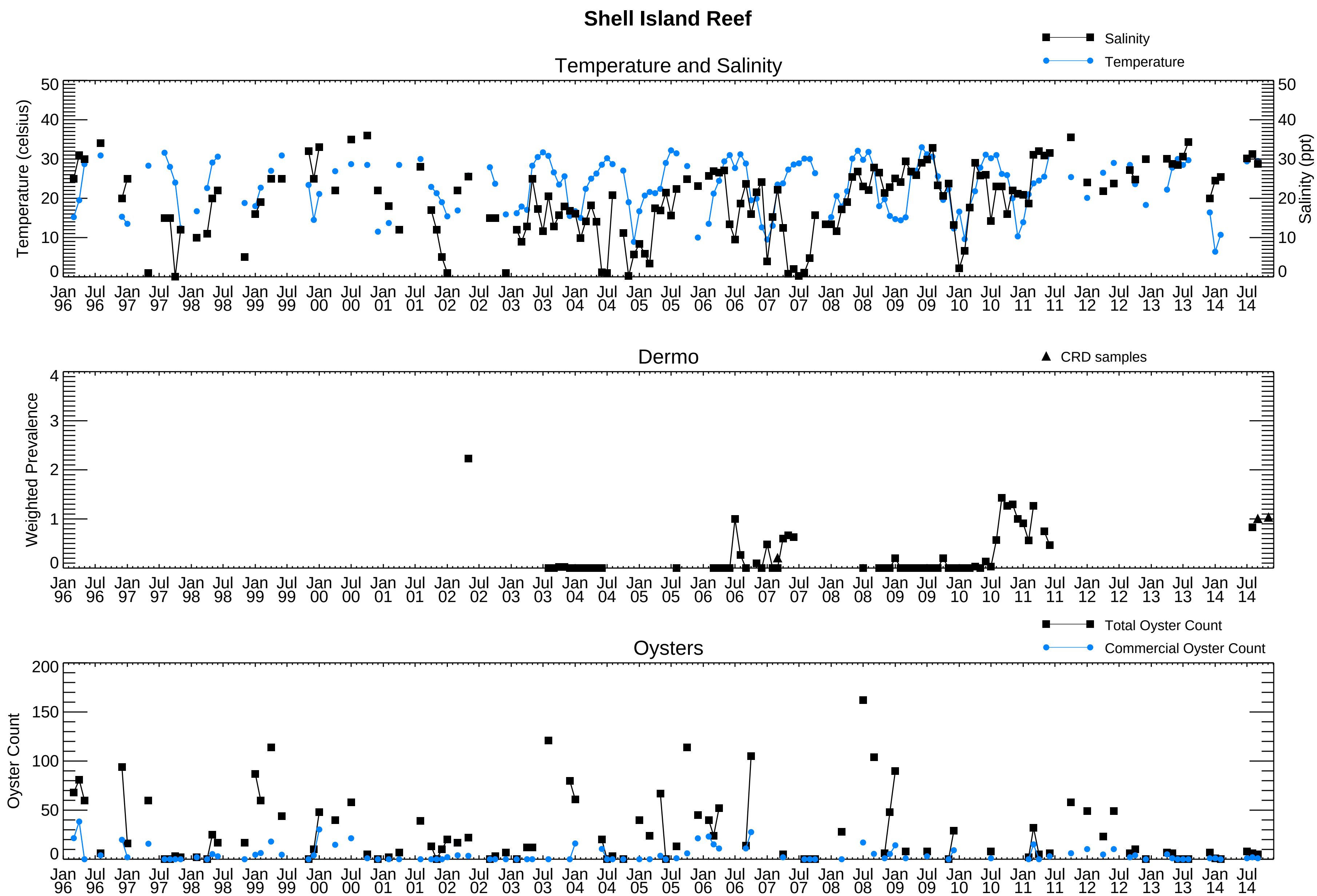

Figure 2-7c

Time Series of Monthly Temperature, Salinity, Dermo and Oyster Results for Shell Island Reef

Data source: TPWD; OysterSentinel.

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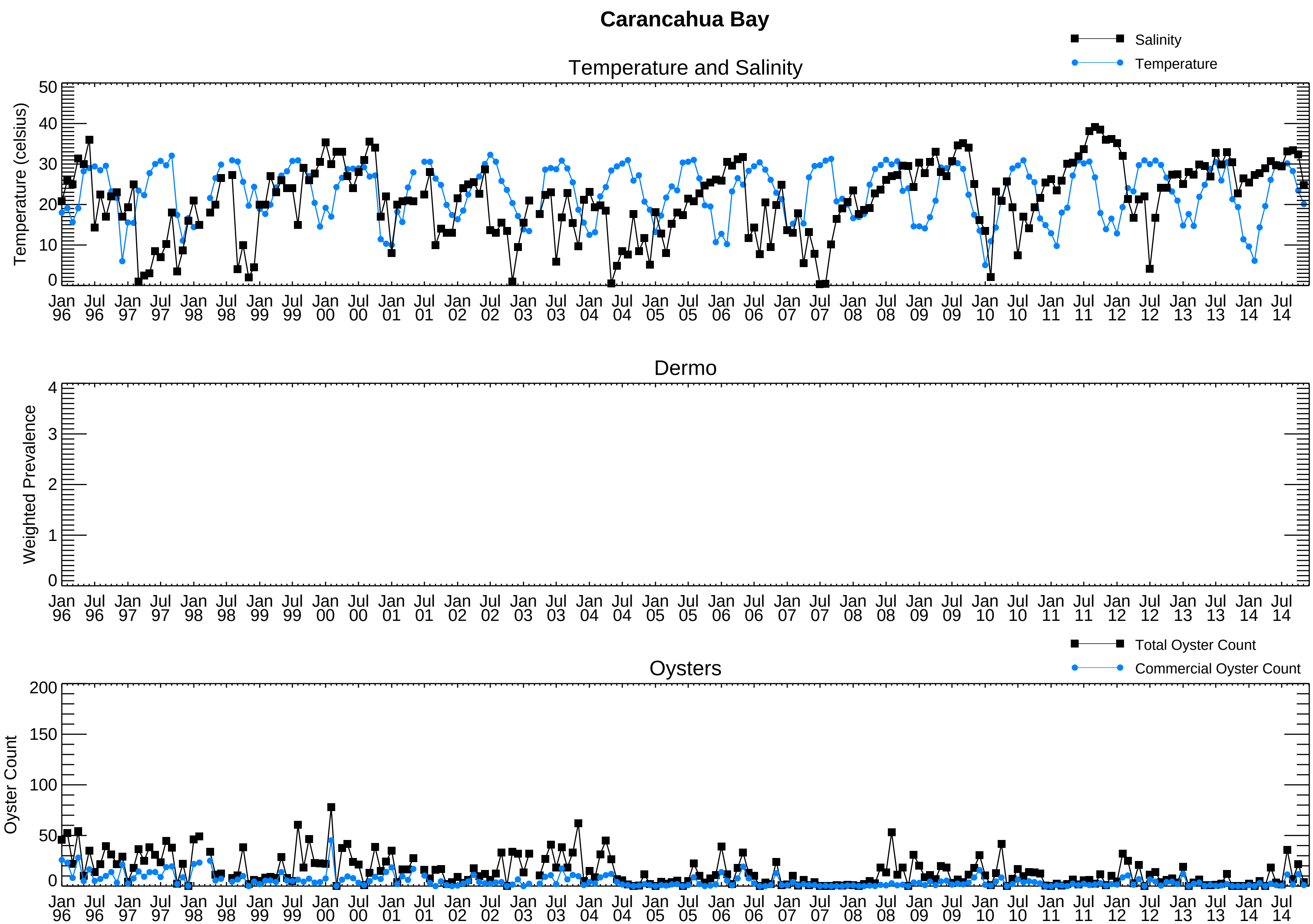

Figure 2-7d

Time Series of Monthly Temperature, Salinity, Dermo and Oyster Results for Carancahua Bay

Data source: TPWD; OysterSentinel.

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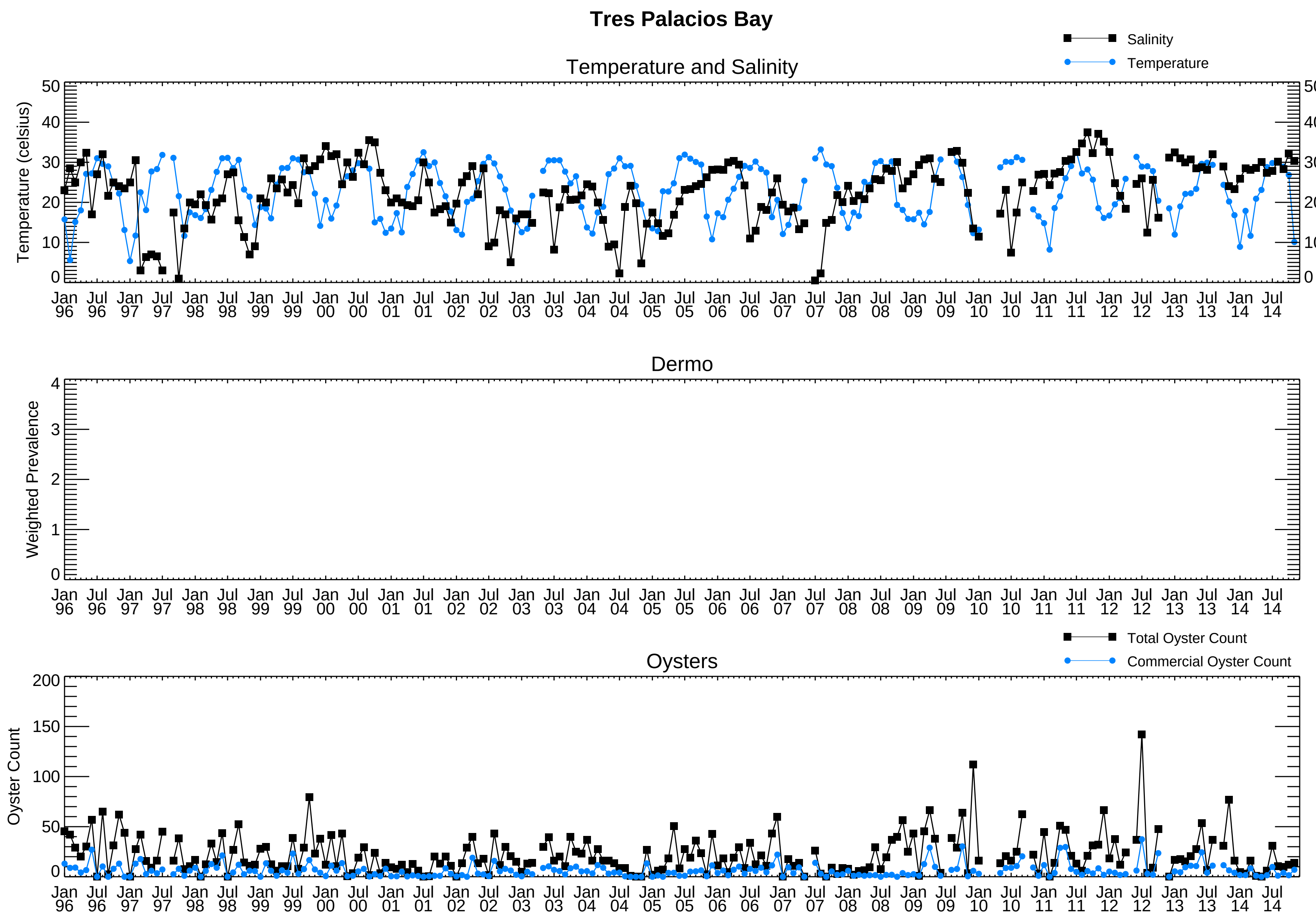

Figure 2-7e

Time Series of Monthly Temperature, Salinity, Dermo and Oyster Results for Tres Palacios Bay

Data source: TPWD; OysterSentinel.

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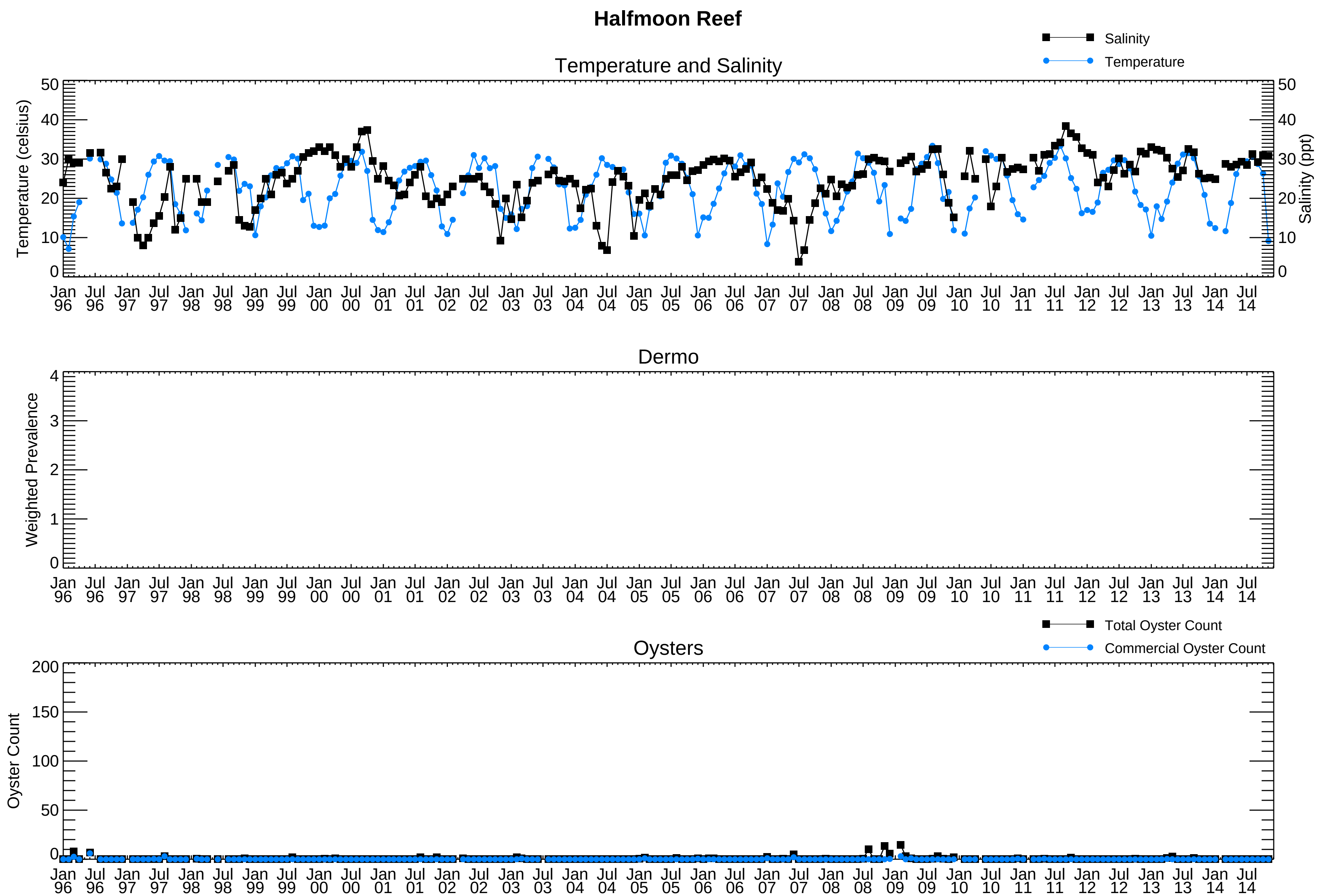

Figure 2-7f

Time Series of Monthly Temperature, Salinity, Dermo and Oyster Results for Halfmoon Reef

Data source: TPWD; OysterSentinel.

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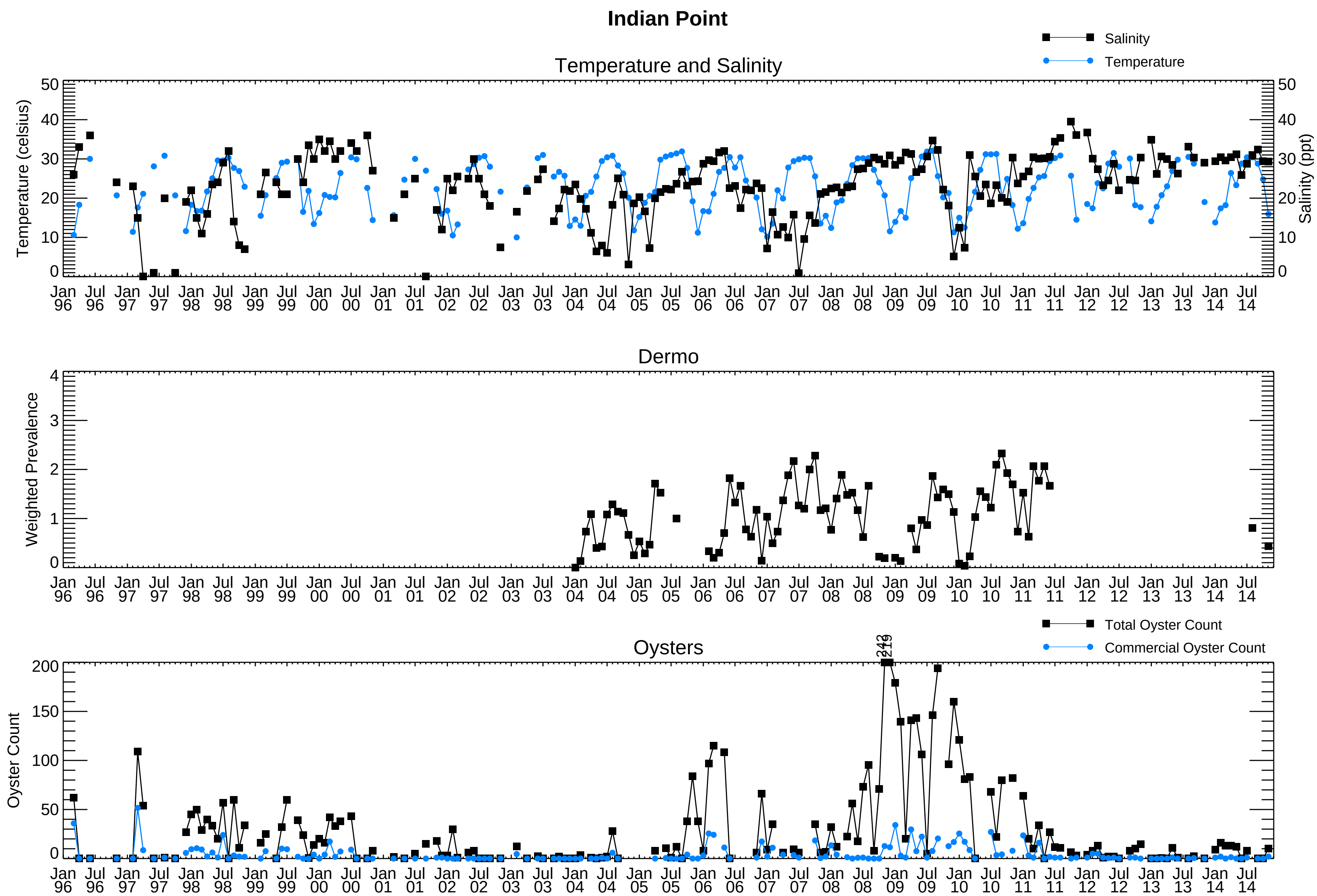

Figure 2-7g

Time Series of Monthly Temperature, Salinity, Dermo and Oyster Results for Indian Point

Data source: TPWD; OysterSentinel.

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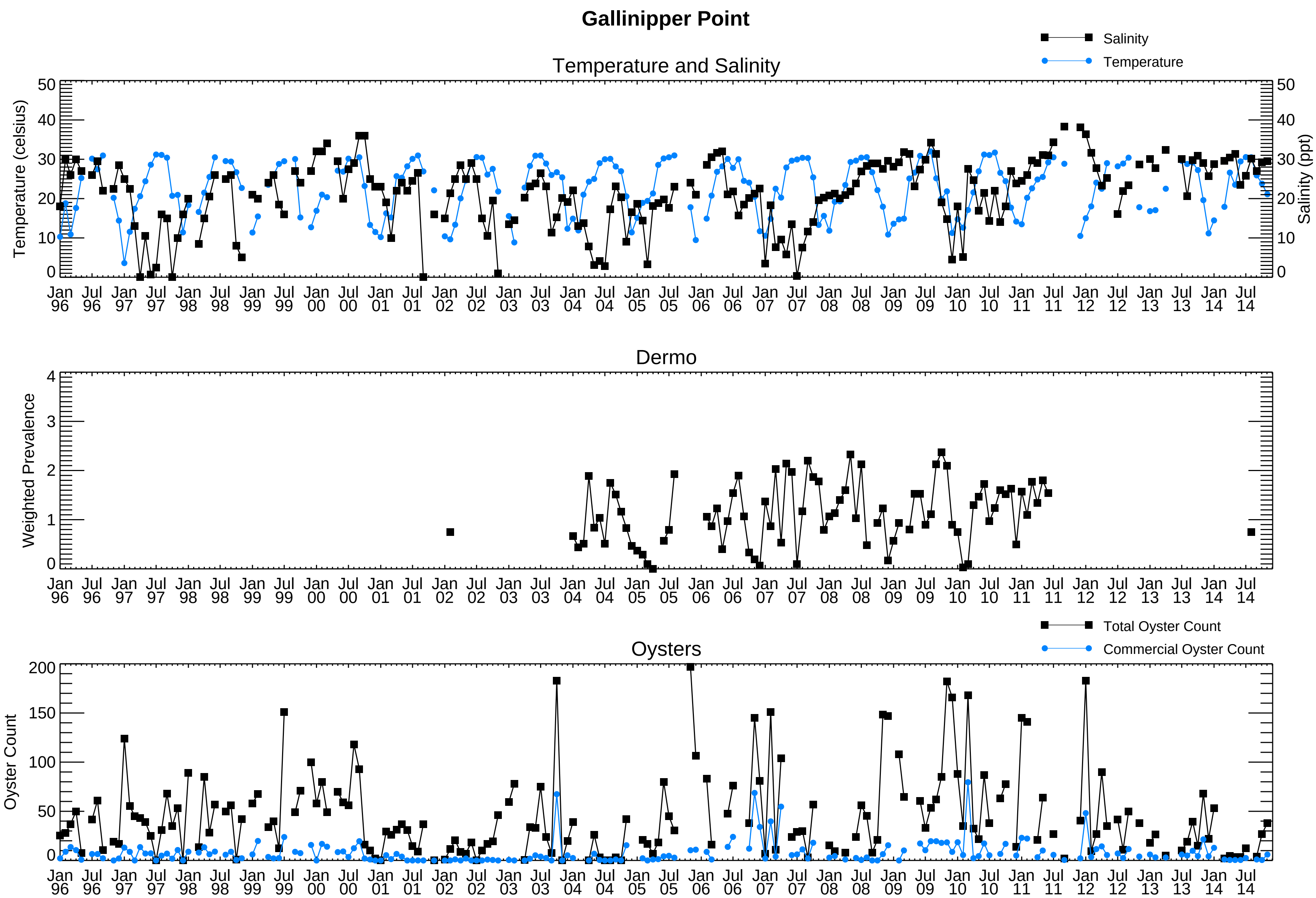

Figure 2-7h

Time Series of Monthly Temperature, Salinity, Dermo and Oyster Results for Gallinipper Point

Data source: TPWD; OysterSentinel.

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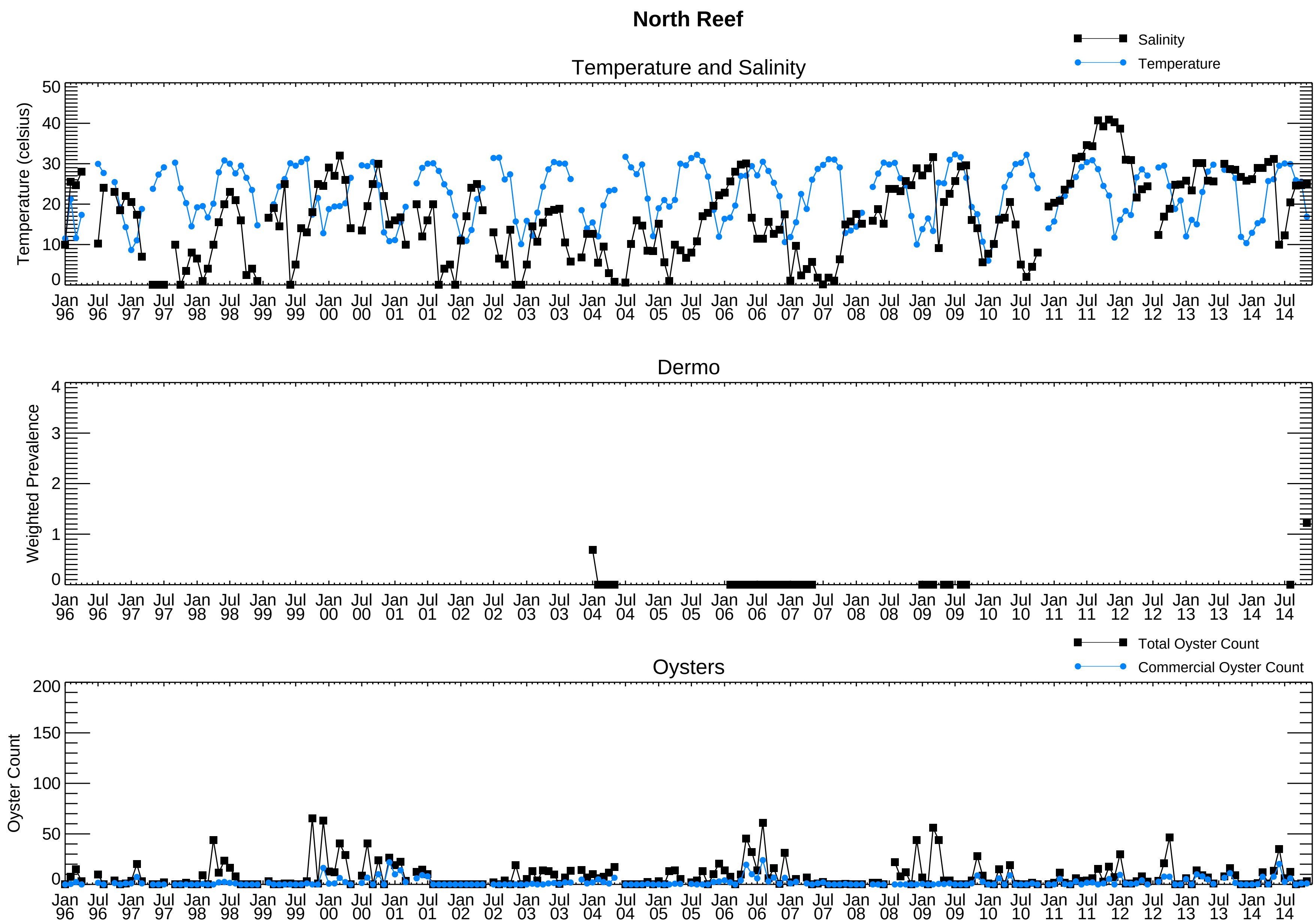

Figure 2-7i

Time Series of Monthly Temperature, Salinity, Dermo and Oyster Results for North Reef

Data source: TPWD; OysterSentinel.

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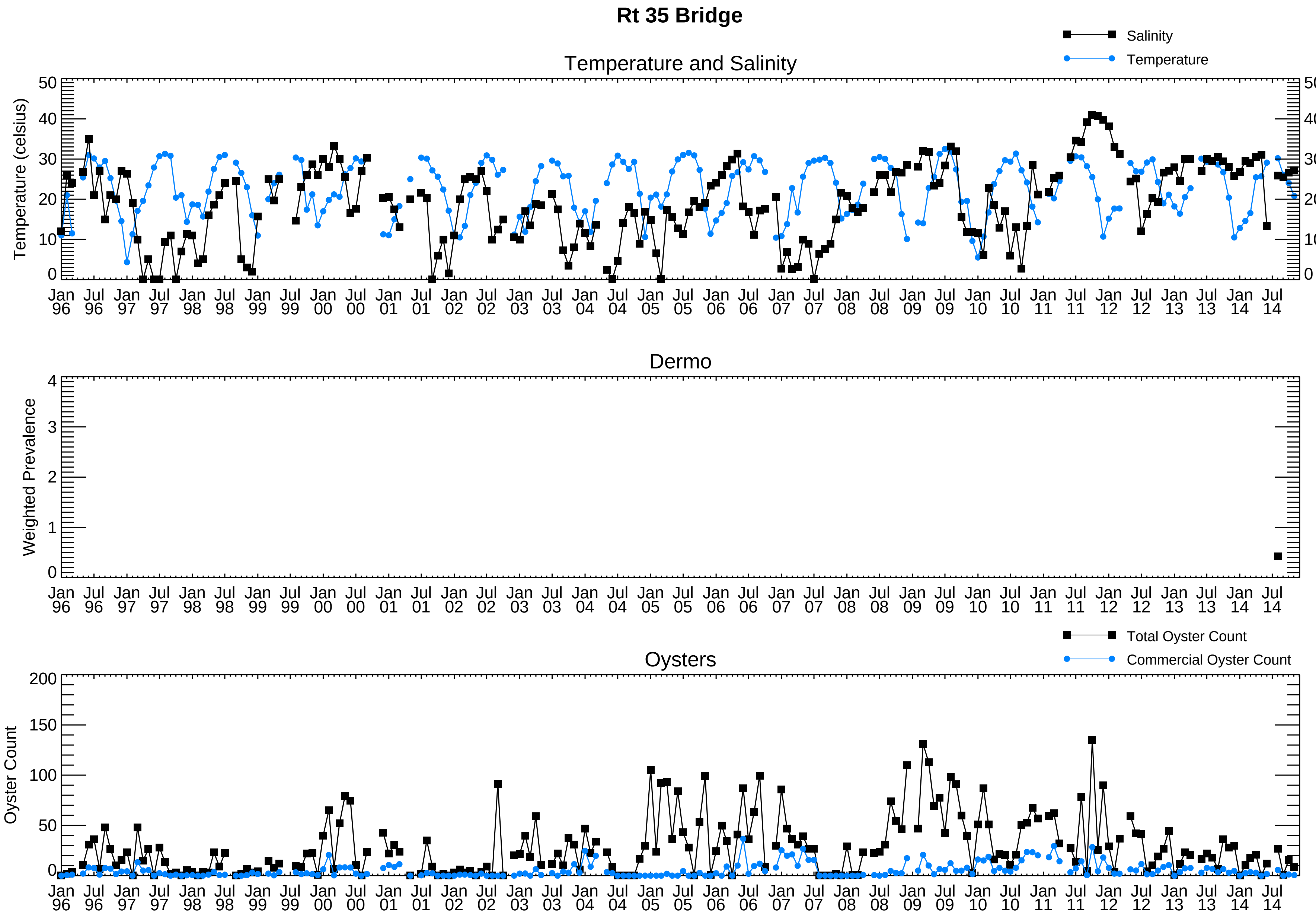

Figure 2-7j

Time Series of Monthly Temperature, Salinity, Dermo and Oyster Results for Rt 35 Bridge

Data source: TPWD; OysterSentinel.

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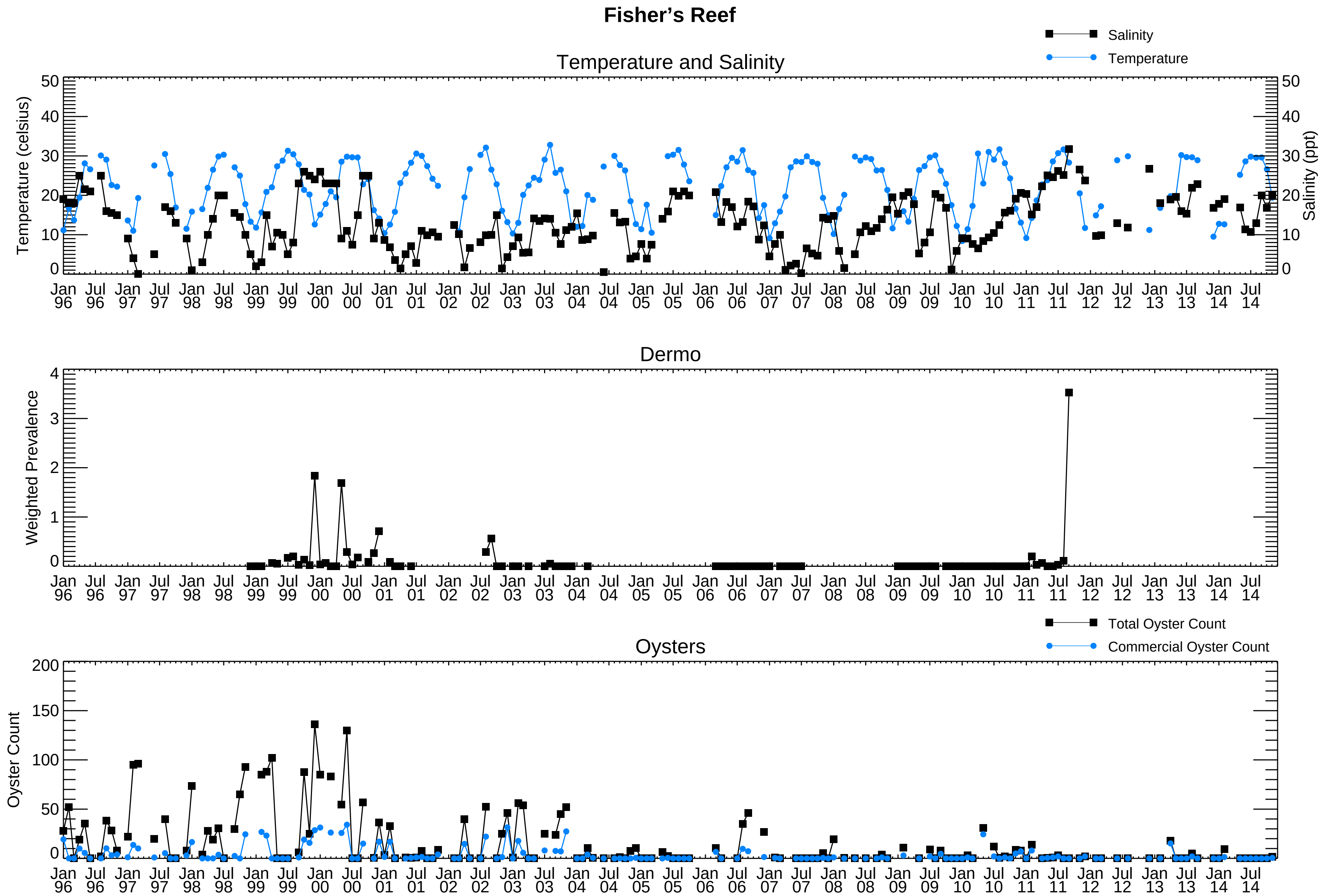

Figure 2-7k

Time Series of Monthly Temperature, Salinity, Dermo and Oyster Results for Fisher's Reef

Data source: TPWD; OysterSentinel.

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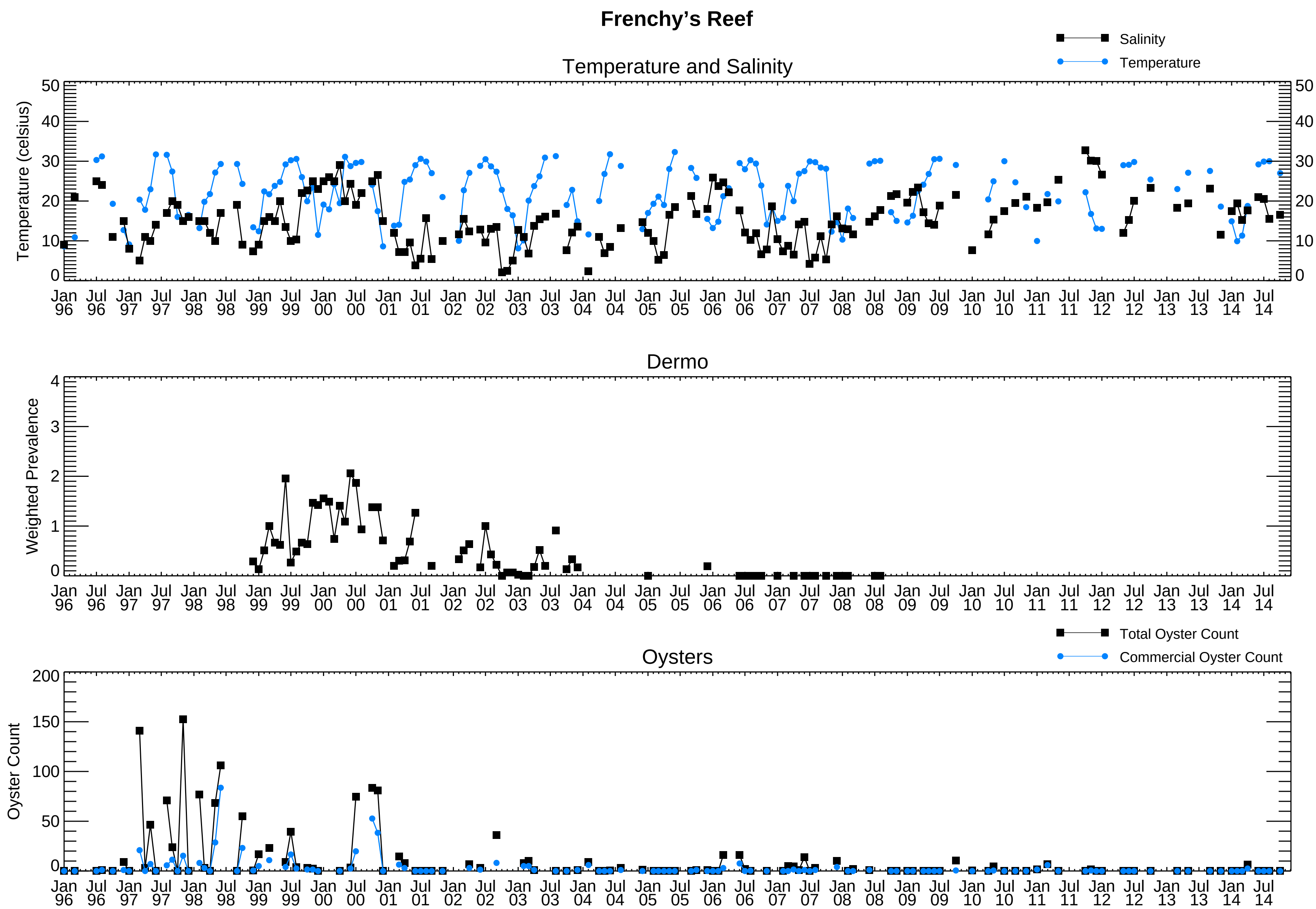

Figure 2-7l

Time Series of Monthly Temperature, Salinity, Dermo and Oyster Results for Frenchy's Reef

Data source: TPWD; OysterSentinel.

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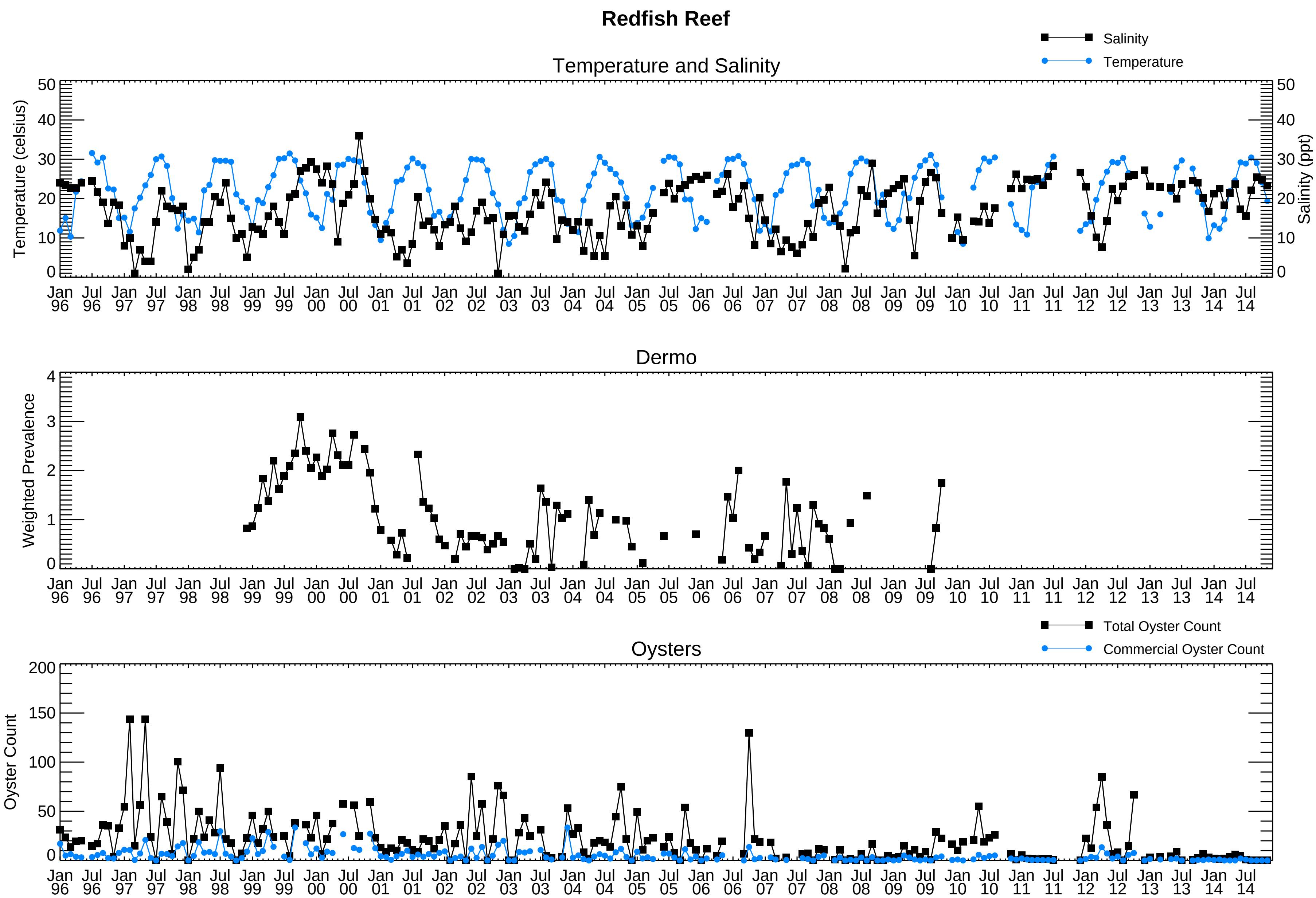

Figure 2-7m

Time Series of Monthly Temperature, Salinity, Dermo and Oyster Results for Redfish Reef

Data source: TPWD; OysterSentinel.

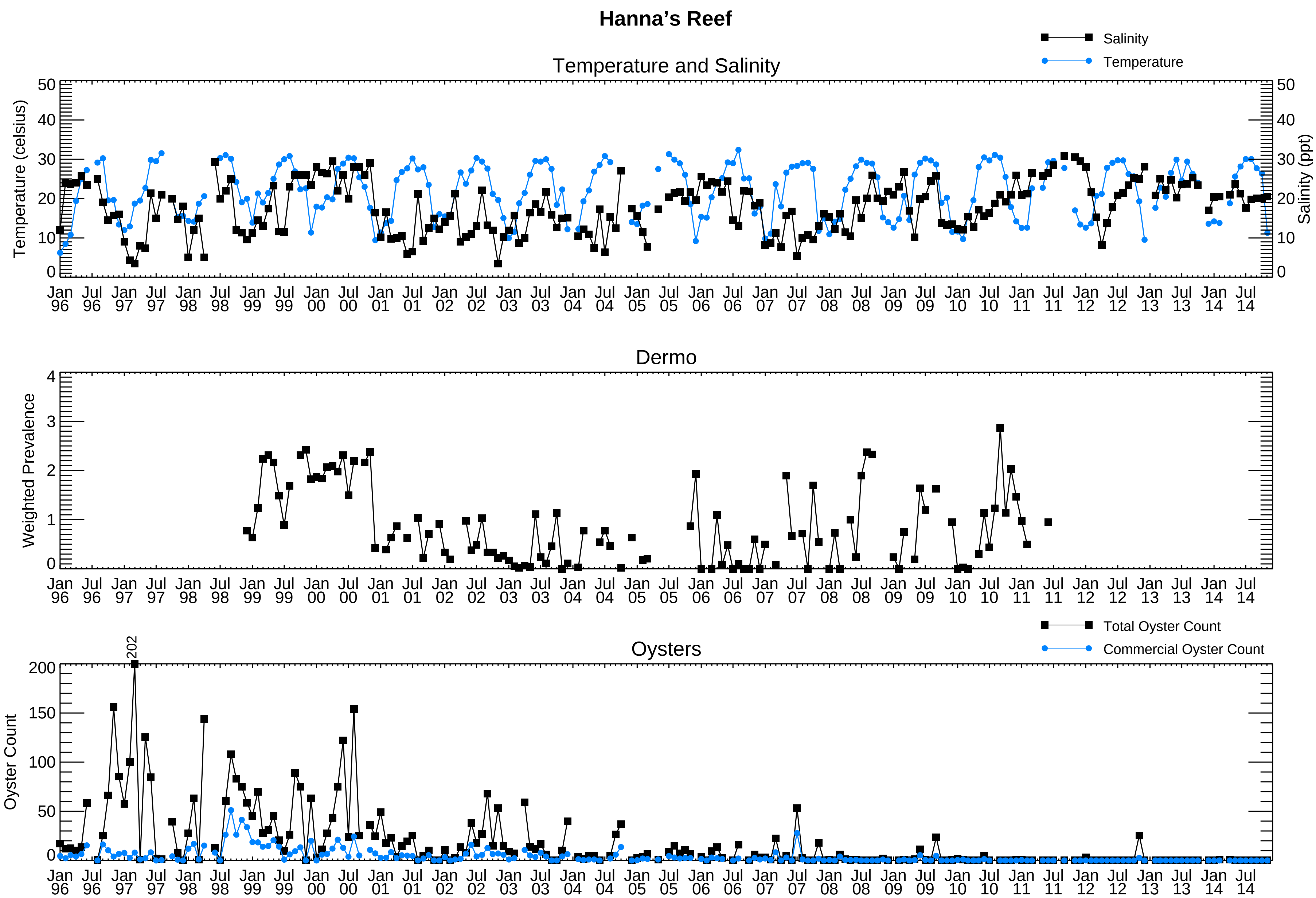

Figure 2-7n

Time Series of Monthly Temperature, Salinity, Dermo and Oyster Results for Hanna's Reef

Data source: TPWD; OysterSentinel.

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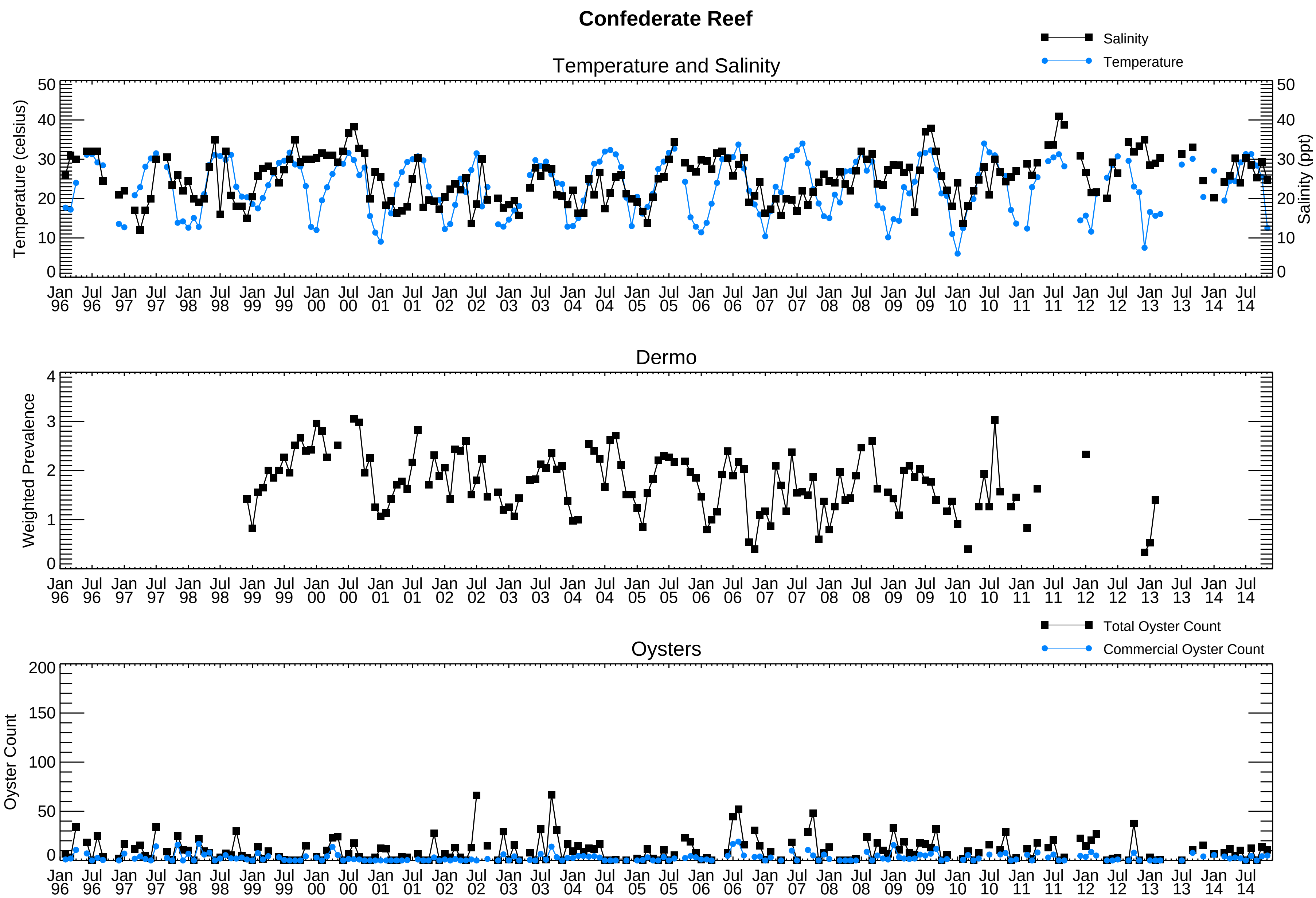

Figure 2-7o

Time Series of Monthly Temperature, Salinity, Dermo and Oyster Results for Confederate Reef

Data source: TPWD; OysterSentinel.

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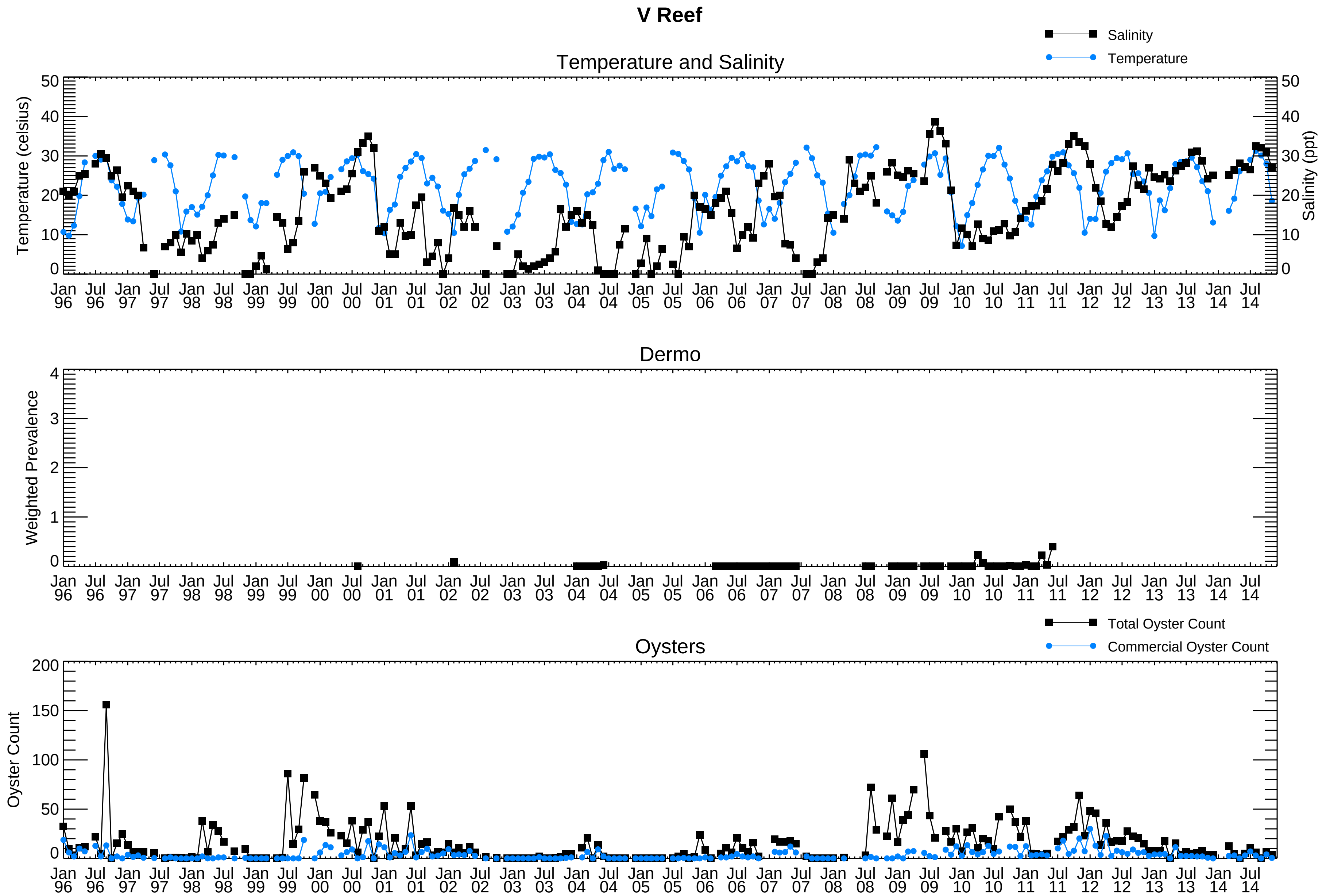

Figure 2-7p

Time Series of Monthly Temperature, Salinity, Dermo and Oyster Results for V Reef

Data source: TPWD; OysterSentinel.

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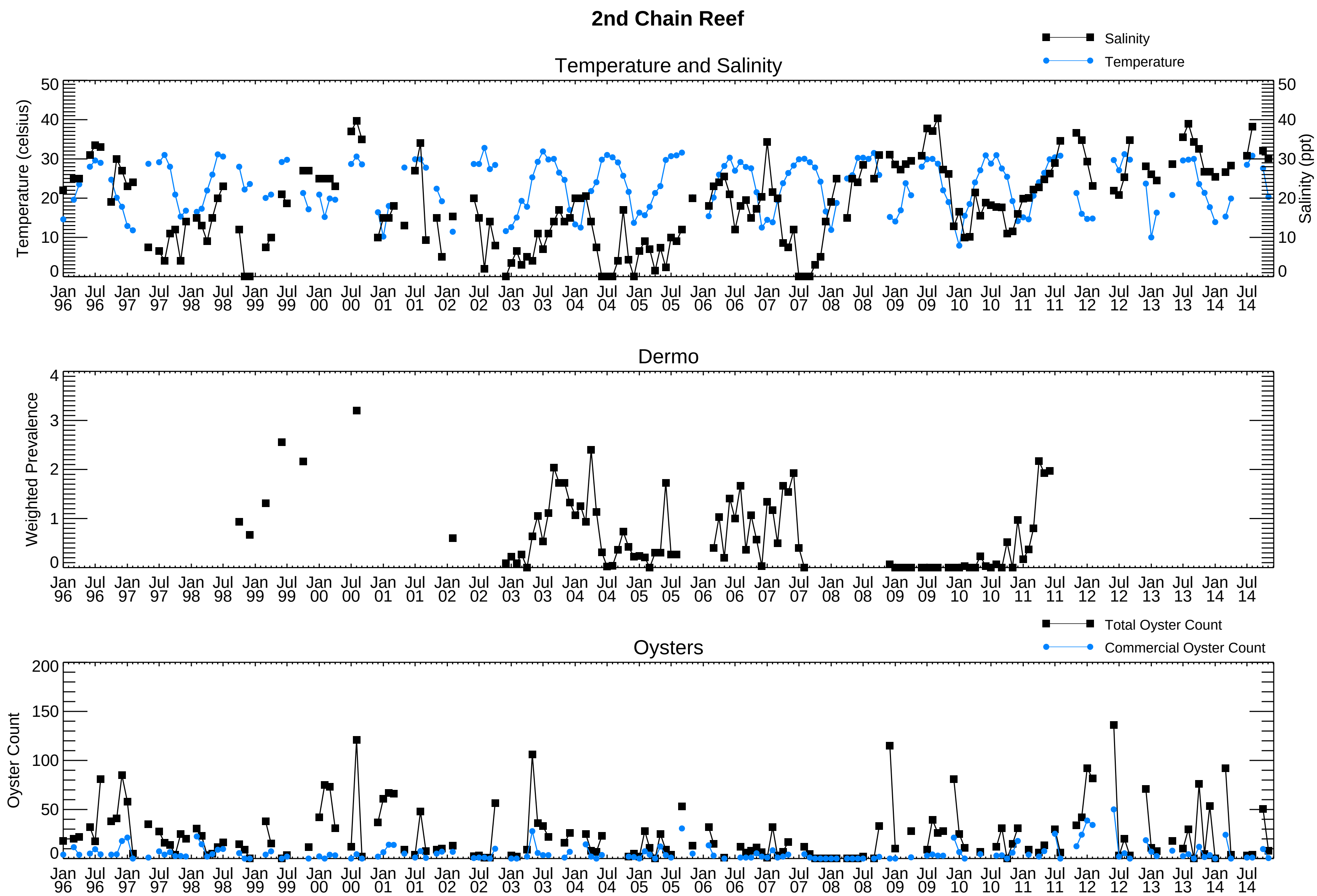

Figure 2-7q

Time Series of Monthly Temperature, Salinity, Dermo and Oyster Results for 2nd Chain Reef

Data source: TPWD; OysterSentinel.

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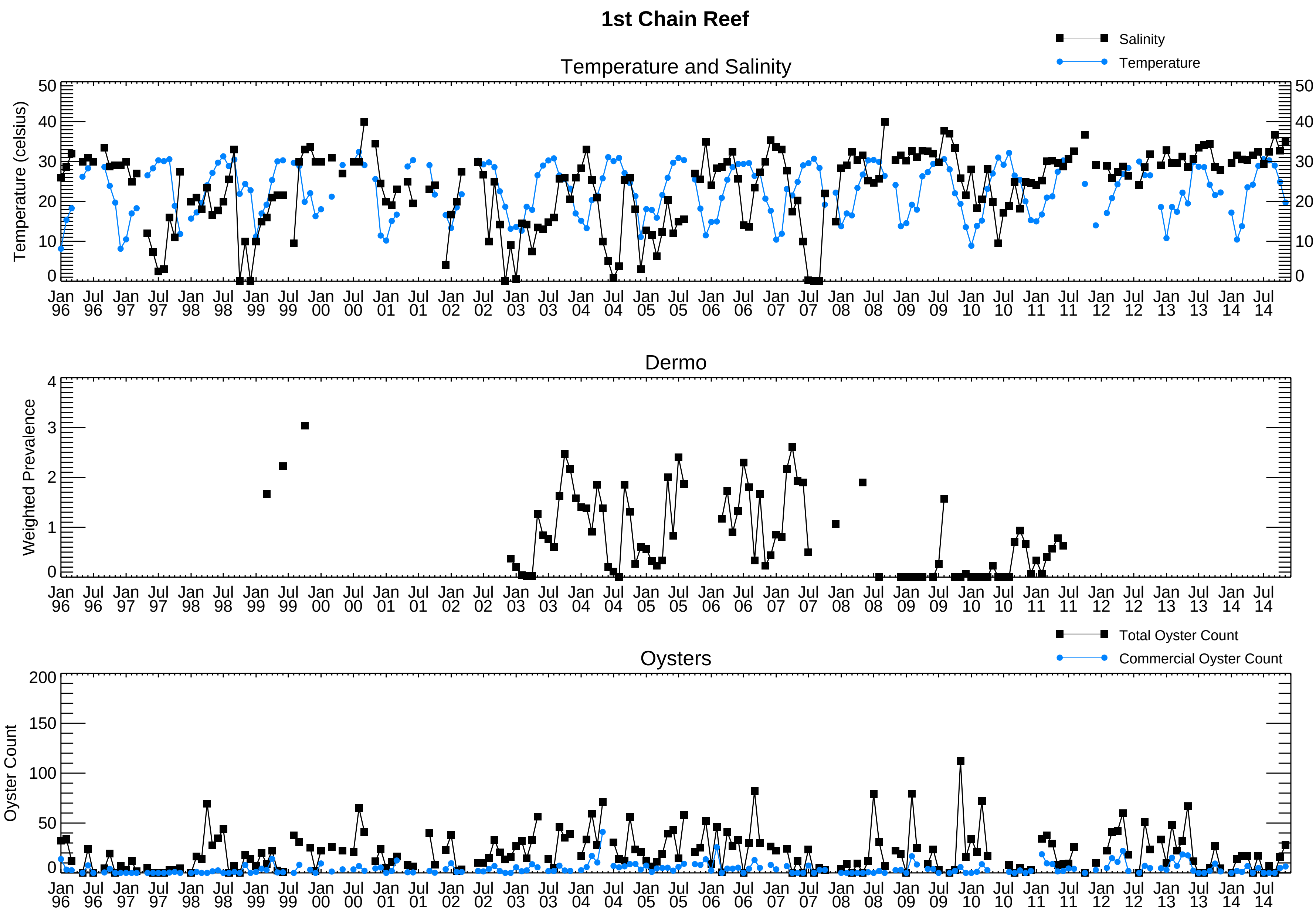

Figure 2-7r

Time Series of Monthly Temperature, Salinity, Dermo and Oyster Results for 1st Chain Reef

Data source: TPWD; OysterSentinel.

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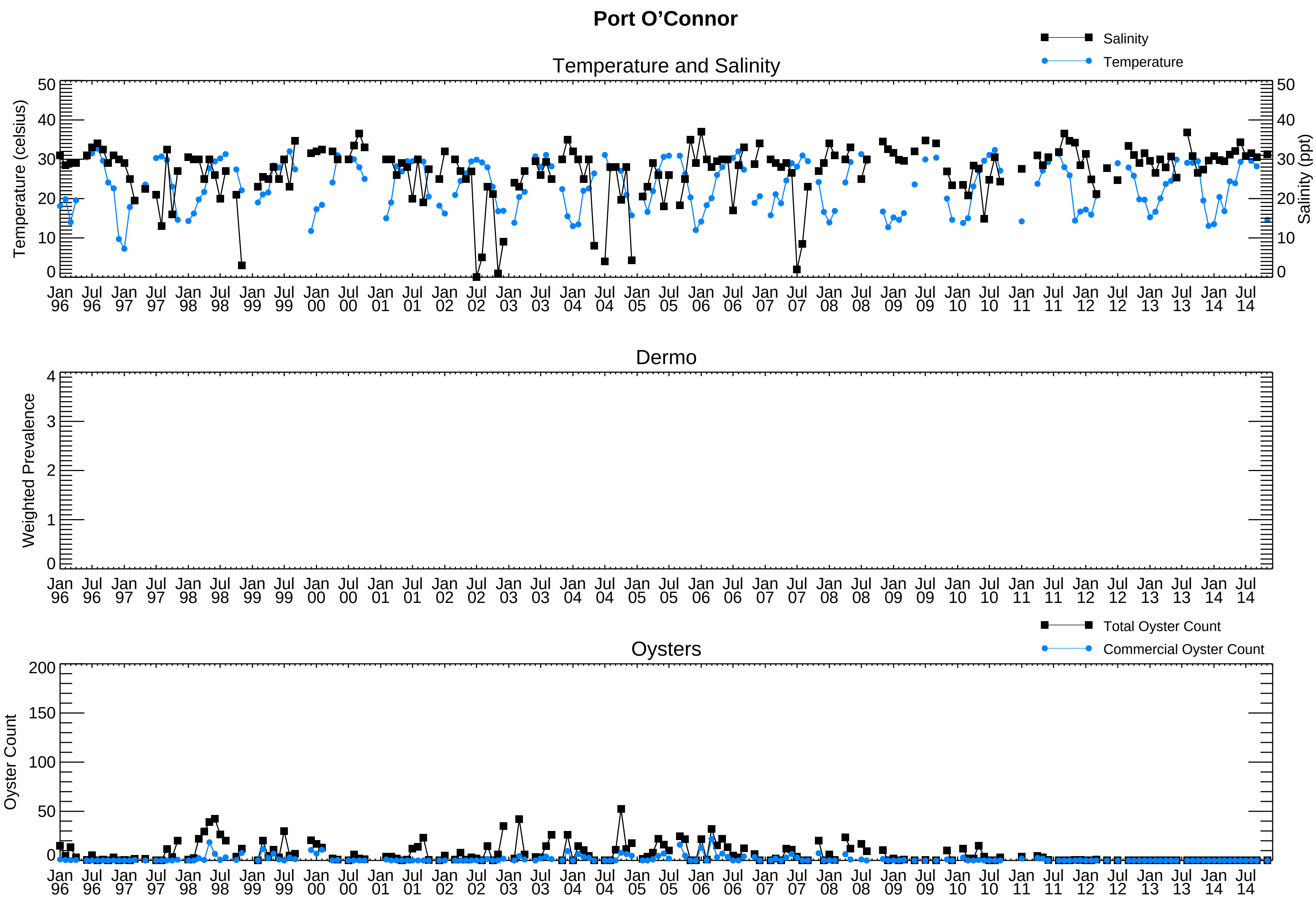

Figure 2-7s

Time Series of Monthly Temperature, Salinity, Dermo and Oyster Results for Port O'Connor

Data source: TPWD; OysterSentinel.

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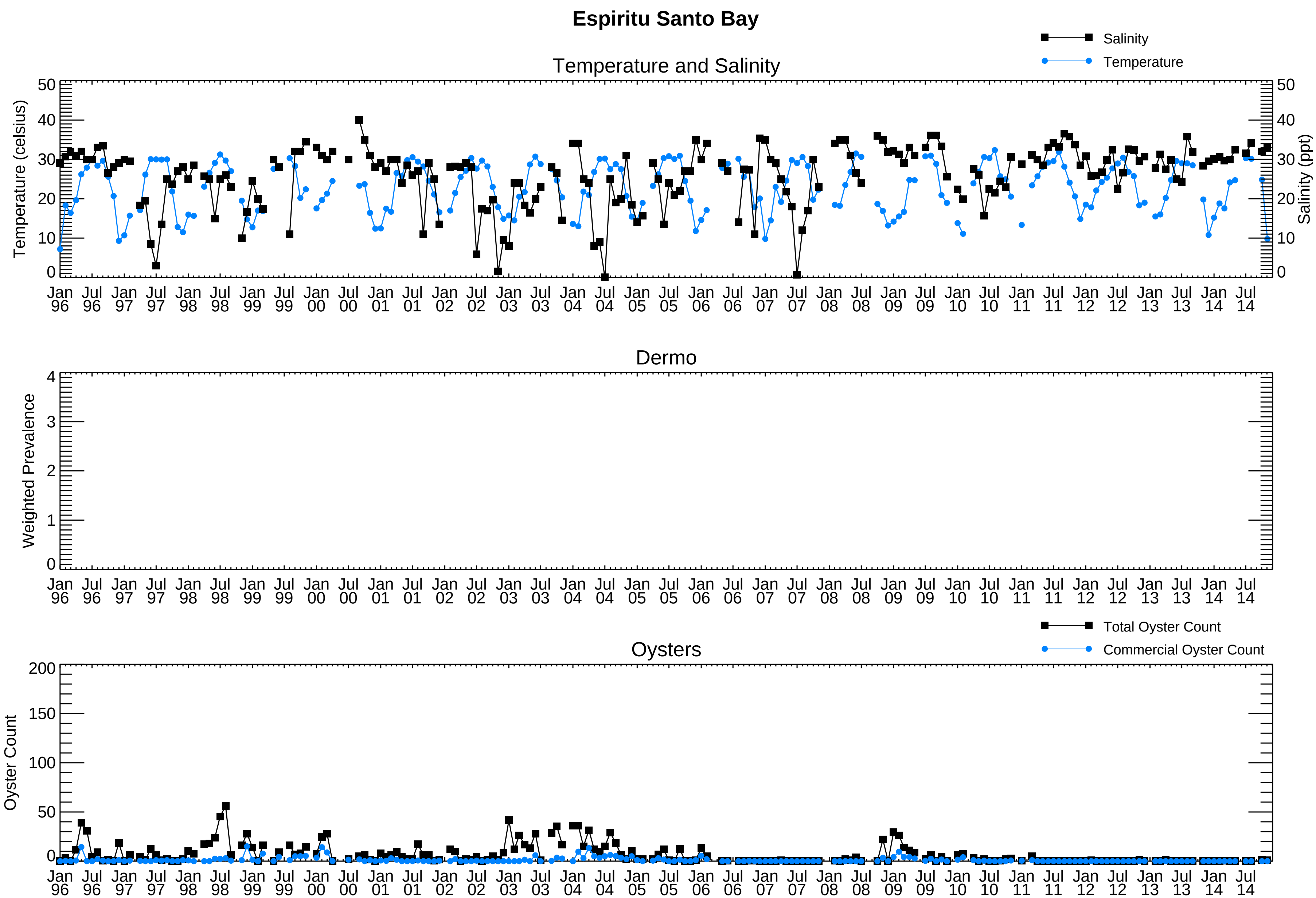

Figure 2-7t

Time Series of Monthly Temperature, Salinity, Dermo and Oyster Results for Espiritu Santo Bay

Data source: TPWD; OysterSentinel.

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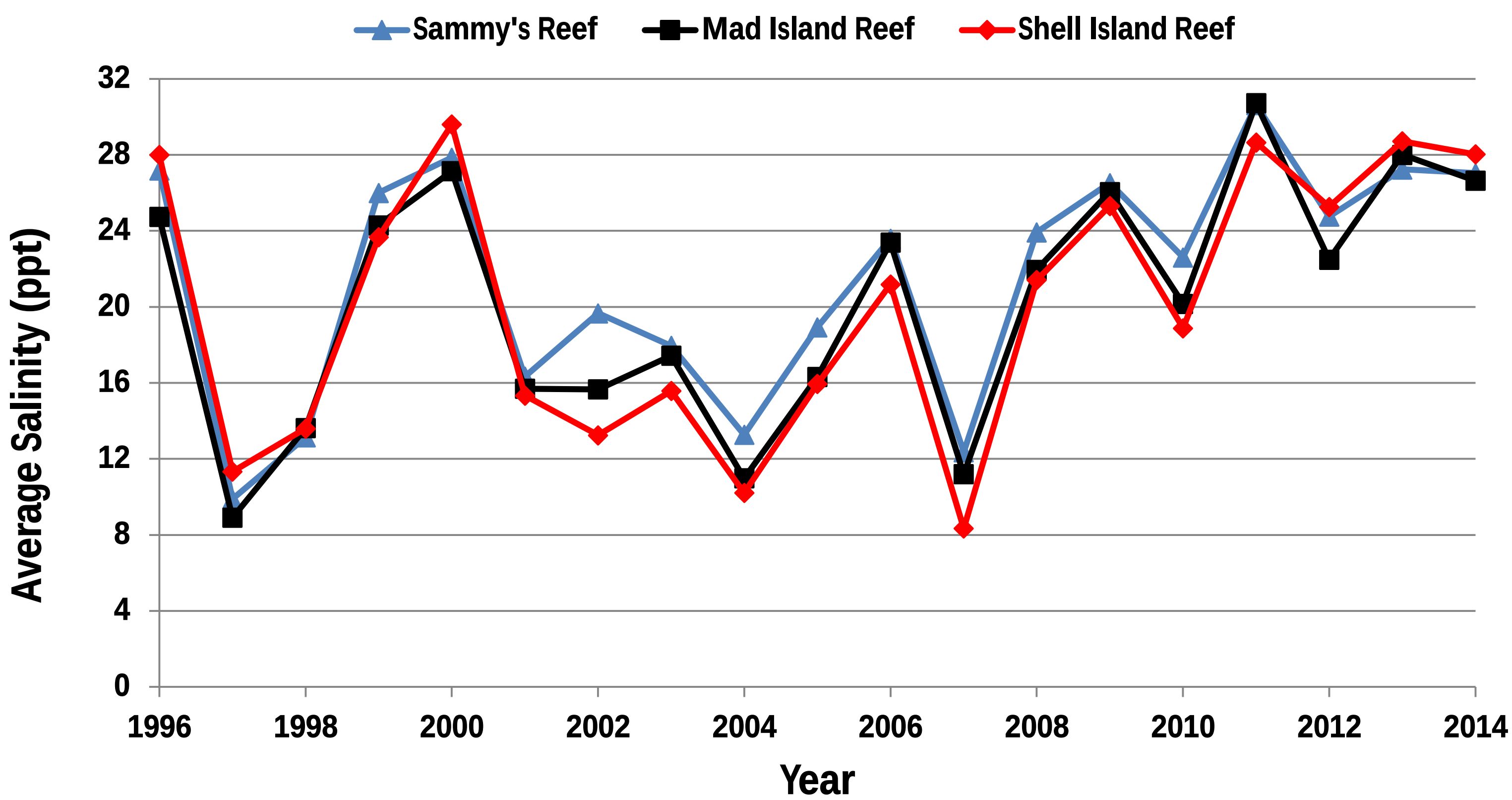

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Figure 2-8
Average Annual Salinities for Three Reefs in the Eastern Arm of Matagorda Bay
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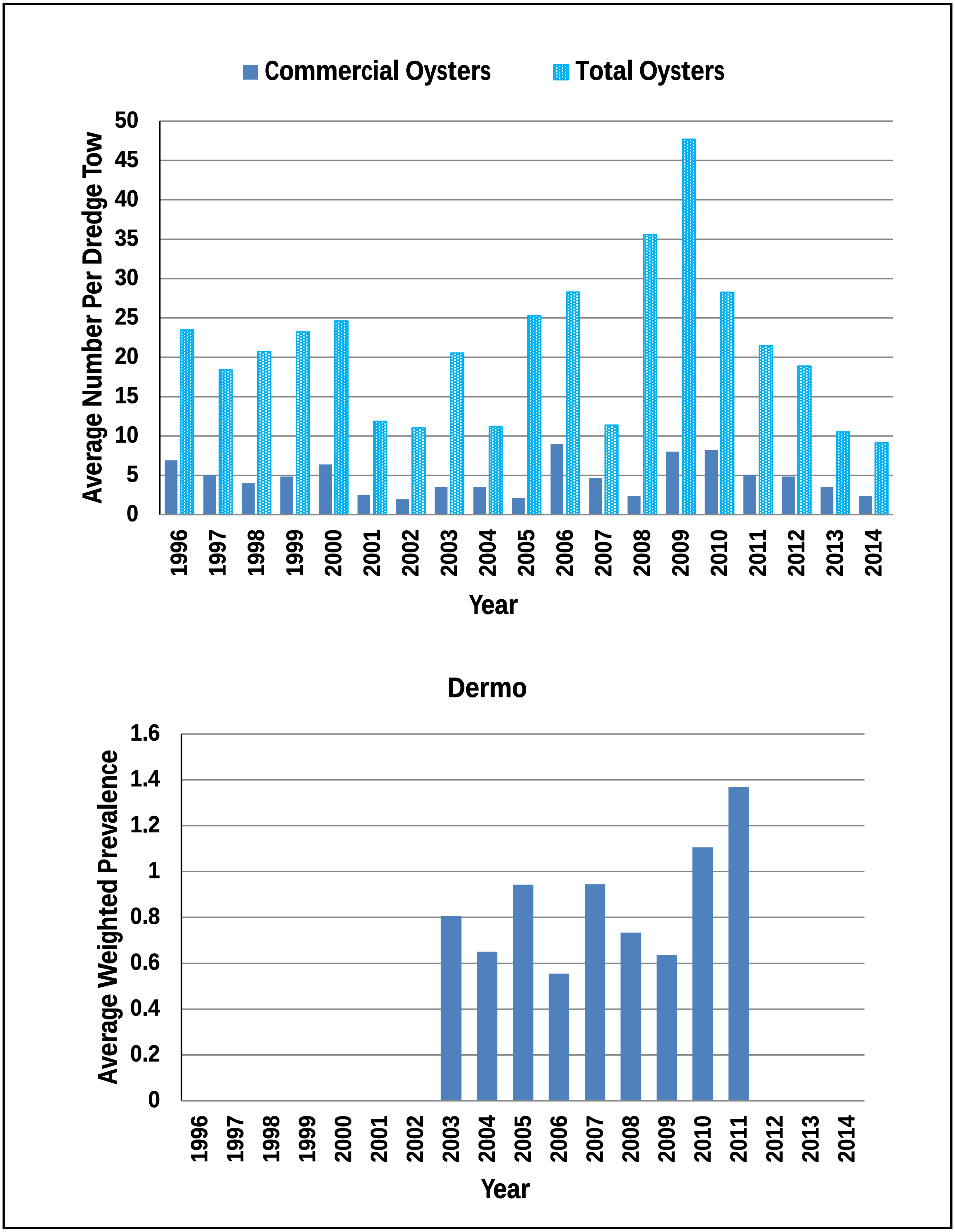

Figure 2-9
Matagorda and Lavaca Whole Bay Annual Trends in Oysters and Dermo
Freshwater Inflow Standards and Ecological Response
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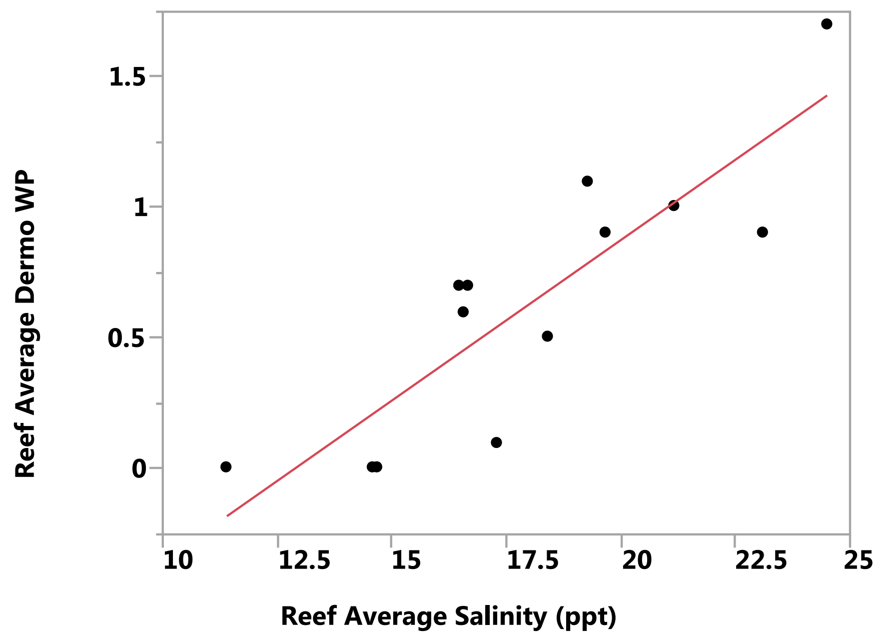
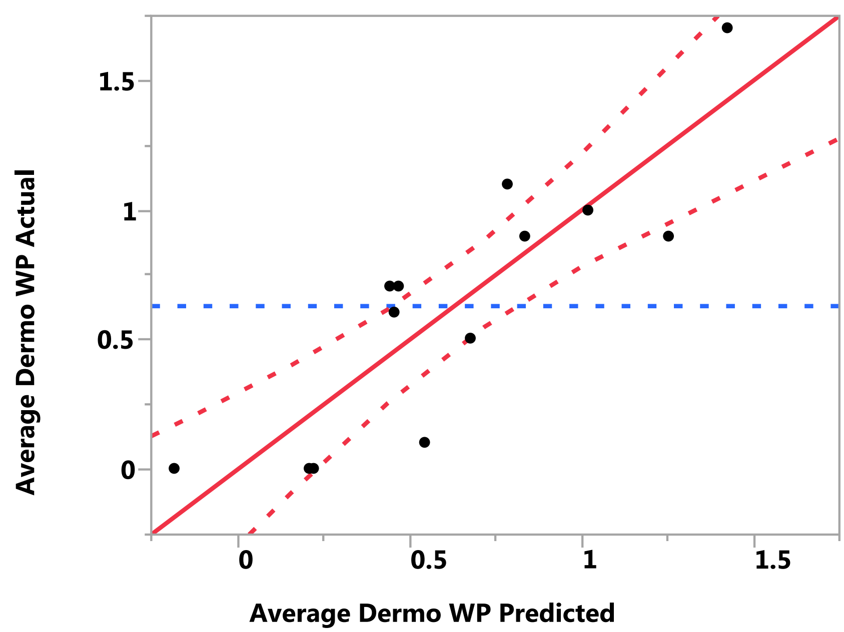
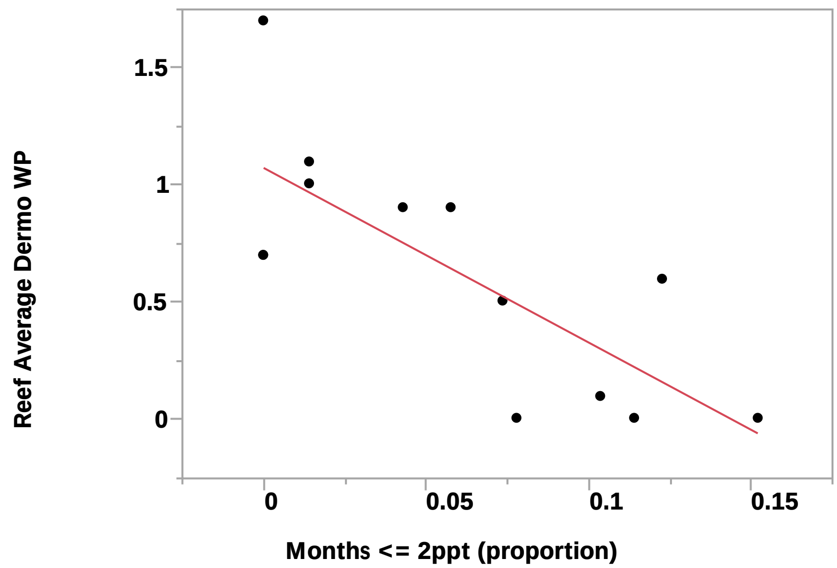
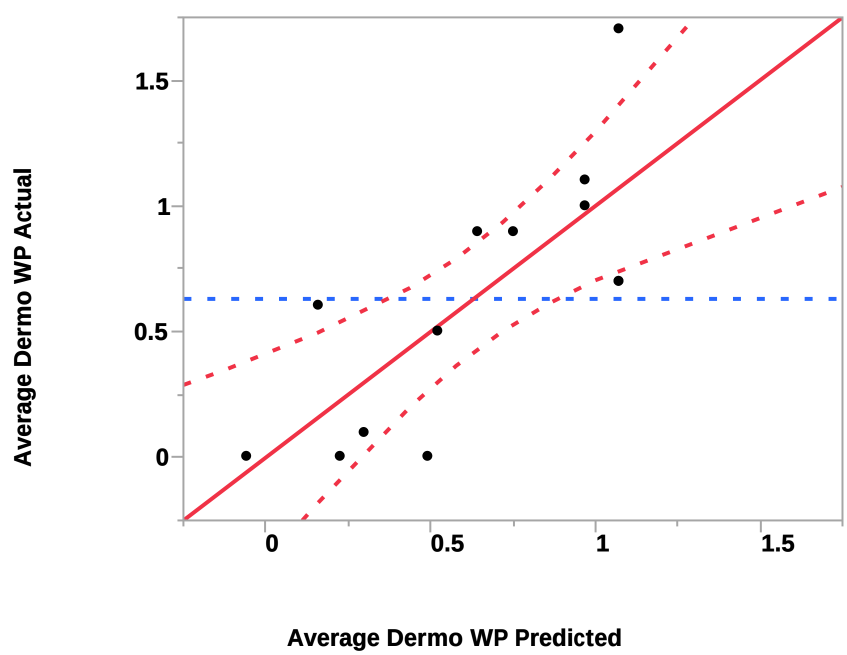




Notes:
Top: Linear Regression of WP vs. Salinity
Bottom: Predicted WP from the Regression vs. Actual WP

Figure 2-10
Relationship Between Average Reef Dermo WP and Average Reef Salinity





Notes:
Top: Linear Regression of WP vs. Salinity ≤ 2
Bottom: Predicted WP from the Regression vs. Actual WP

Figure 2-11
Relationship Between Average Reef Dermo WP and Average Reef Proportion of Months with Salinity ≤ 2 ppt
Freshwater Inflow Standards and Ecological Response
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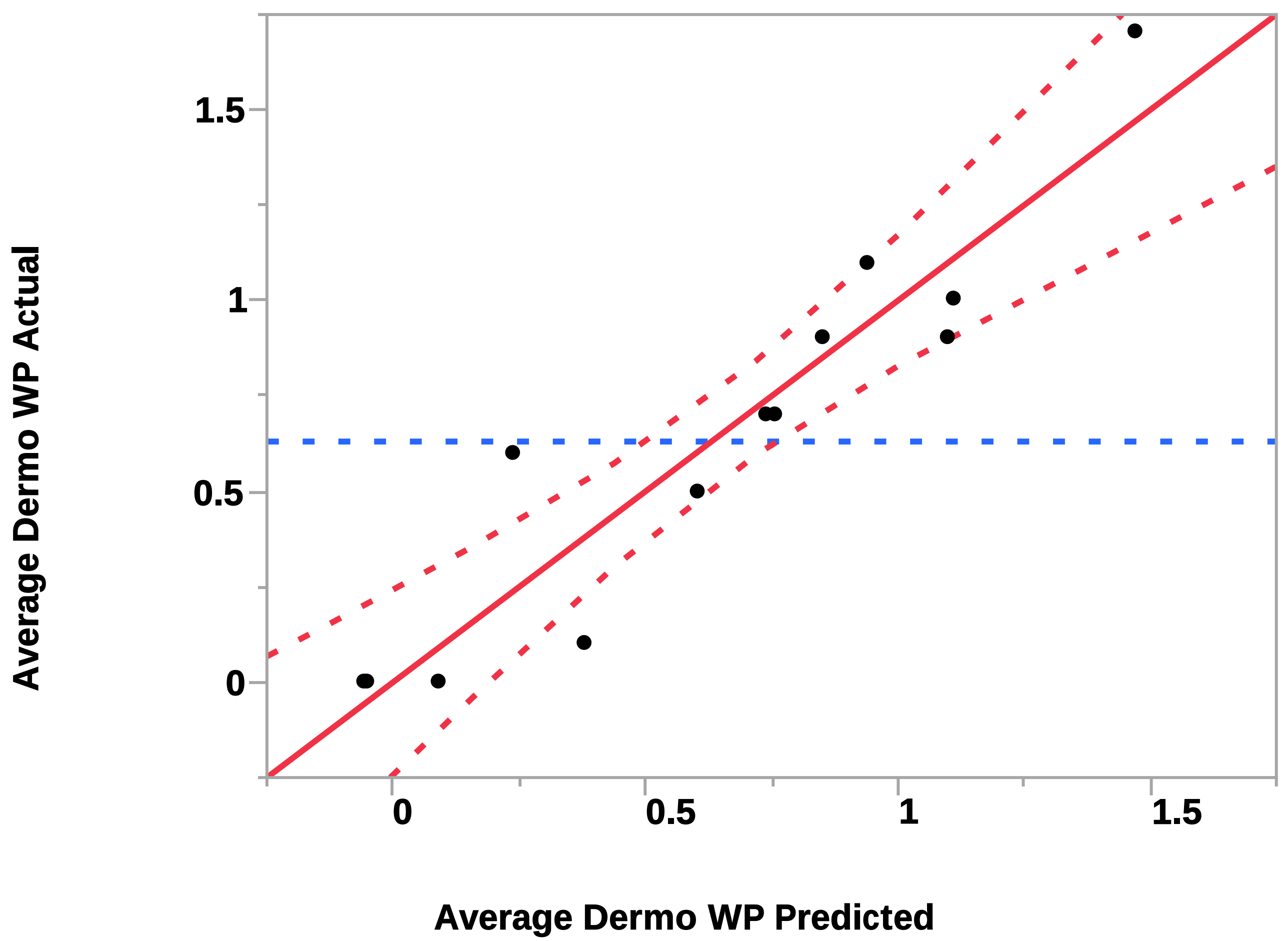

Figure 2-12
Predicted vs. Actual Average Reef Dermo WP for the Multiple Regression of Dermo vs. Average Reef Salinity and Average Reef Proportion of Months with Salinity ≤ 2 ppt
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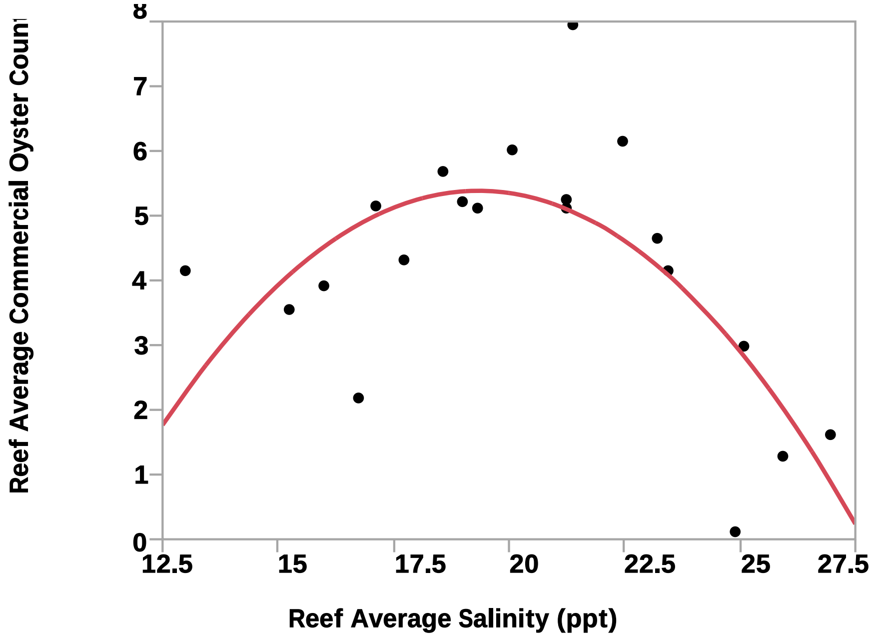

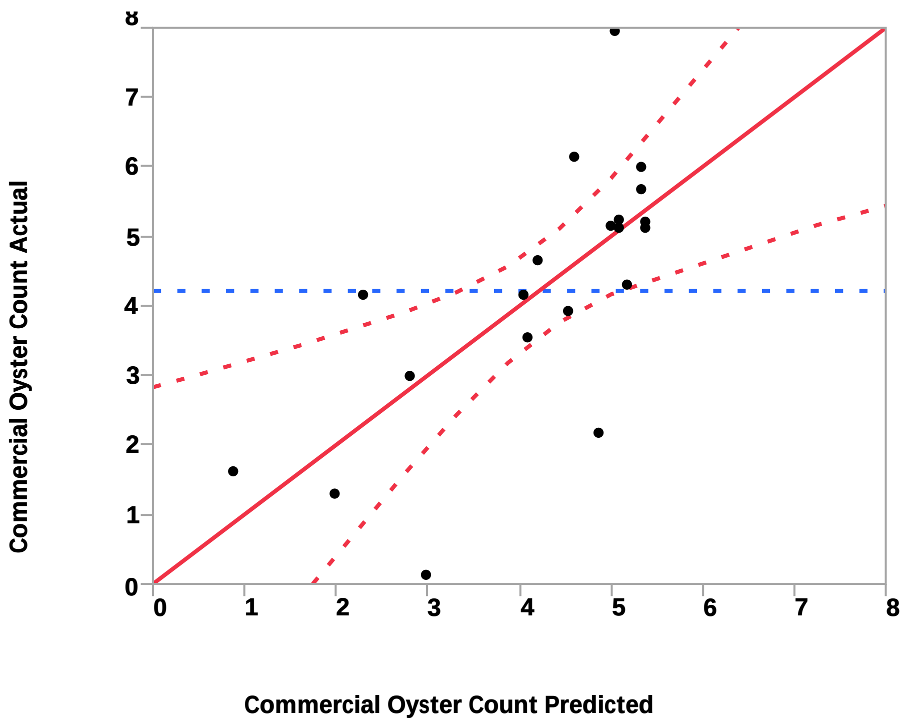

Notes:
Top: Regression of Oyster Count vs. Salinity
Bottom: Predicted Oyster Count from the Regression vs. Actual Oyster Count

Figure 2-13
Relationship Between Average Reef Oyster Count and Average Reef Salinity
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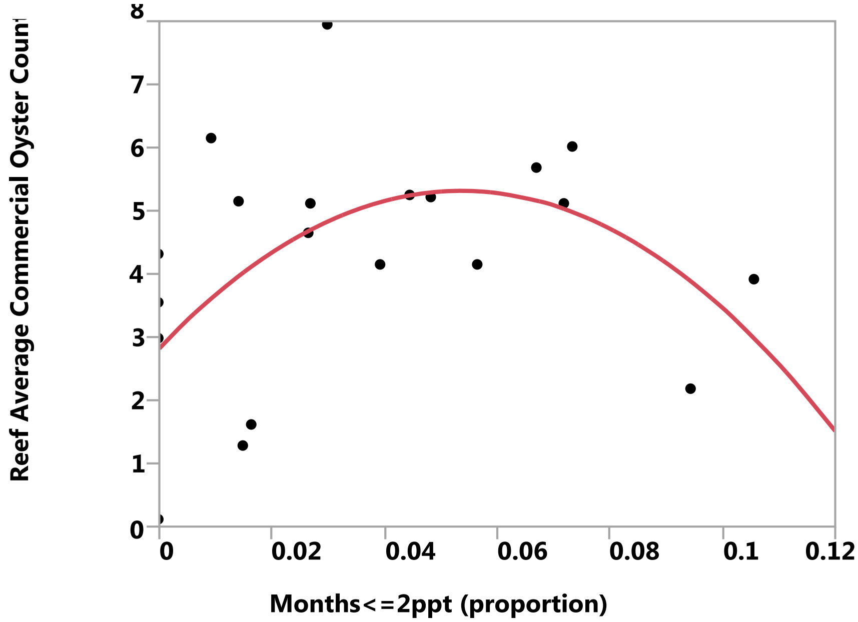

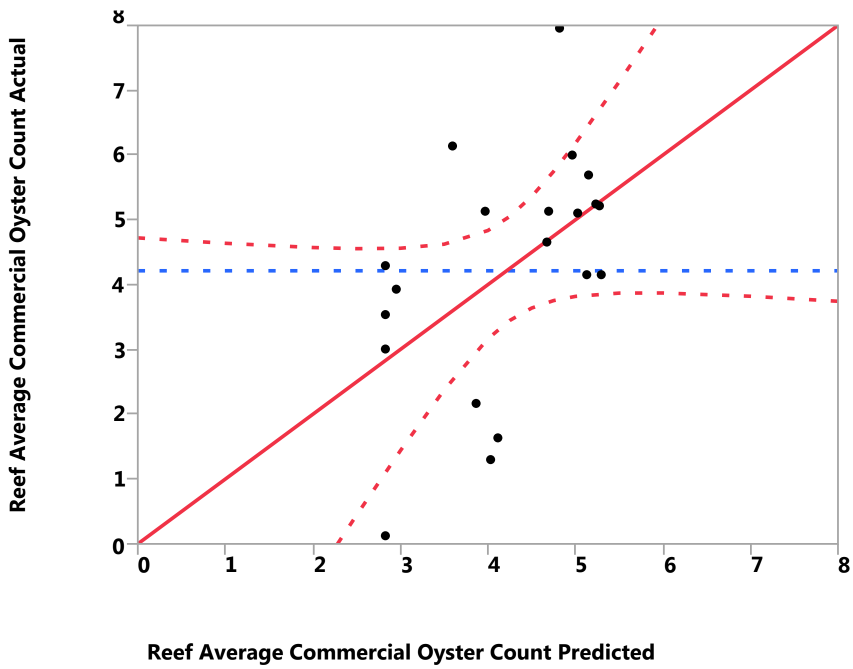

Notes:
Top: Regression of Oyster Count vs. Salinity ≤ 2
Bottom: Predicted Oyster Count from the Regression vs. Actual Oyster Count

Figure 2-14
Relationship Between Average Reef Dermo Oyster Count and Average Reef Proportion of Months with Salinity ≤ 2 ppt
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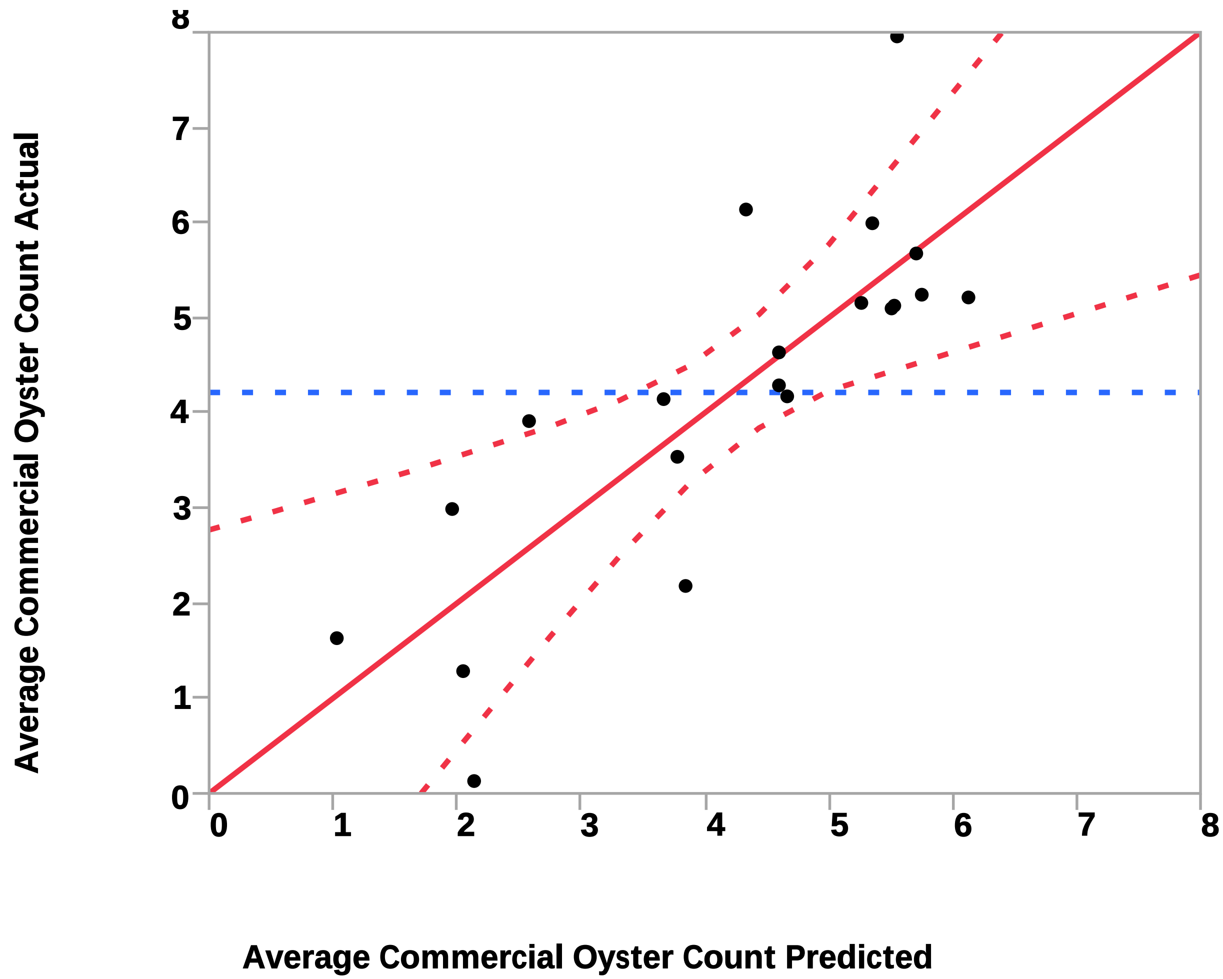

Figure 2-15

Predicted vs. Actual Average Reef Oyster Count for the Multiple Regression of Oyster Count vs. Average Reef Salinity and Average Reef Proportion of Months with Salinity ≤ 2

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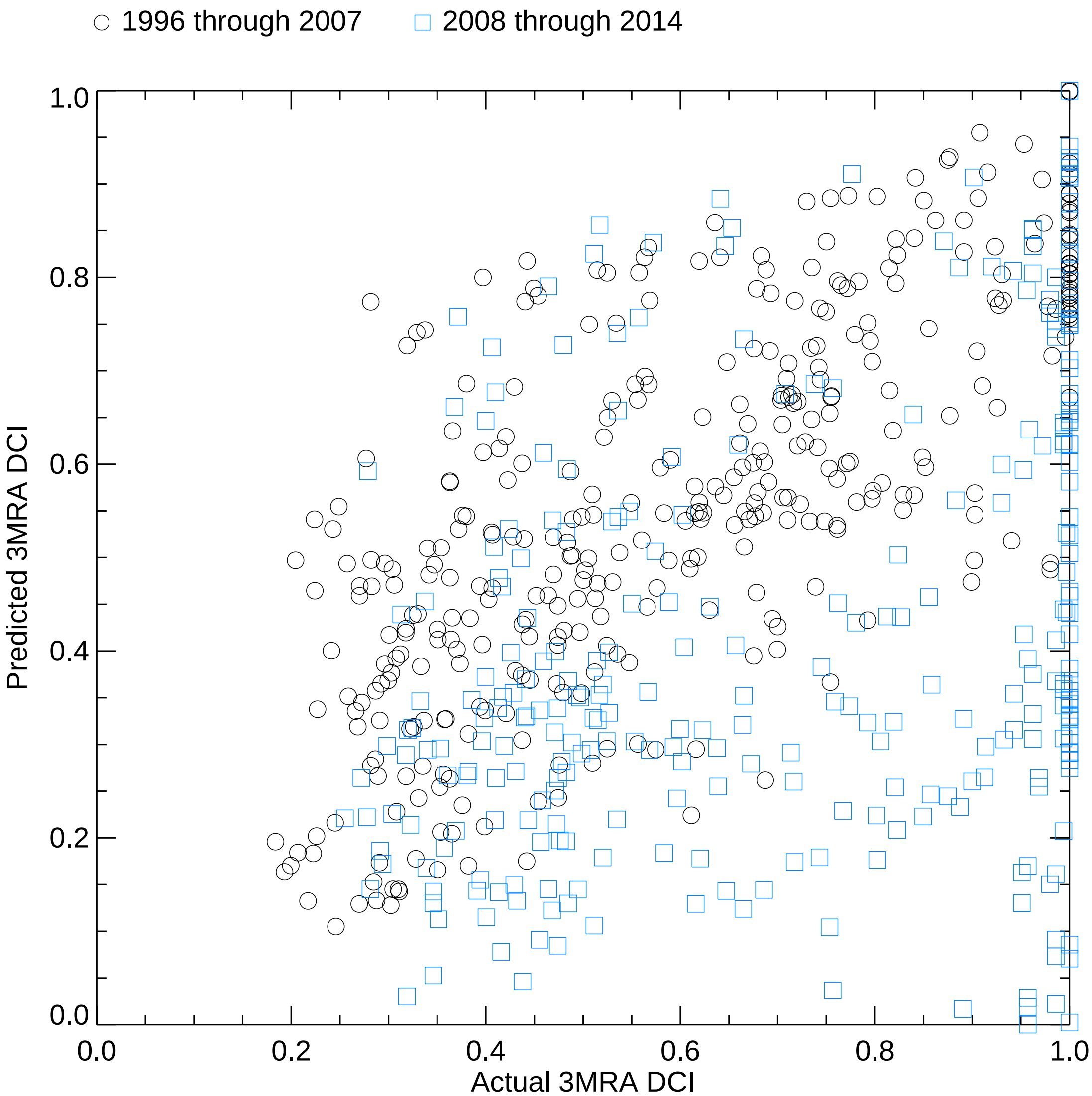

Figure 2-16

New and Old Data Results for Predicted vs. Actual 3MRA DCI for the MBHE 2008 Model

Data source: TPWD; OysterSentinel.

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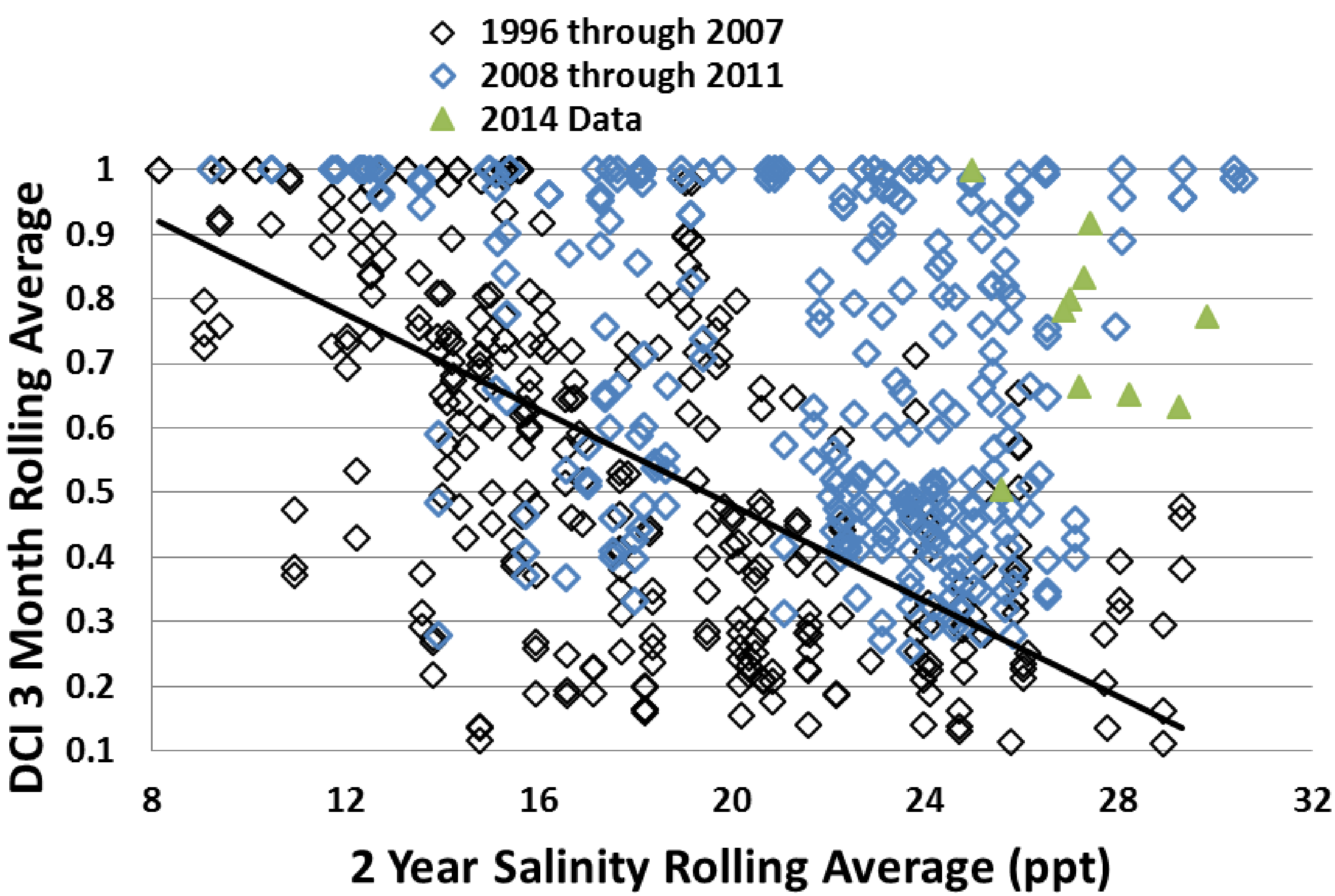

Note: Regression trend line is shown for 1996 through 2007 data only.

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Figure 2-17
Comparison of New and Old Data TPWD Data and 2014 Field Data for DCI vs. 2-Year Rolling Salinity Average
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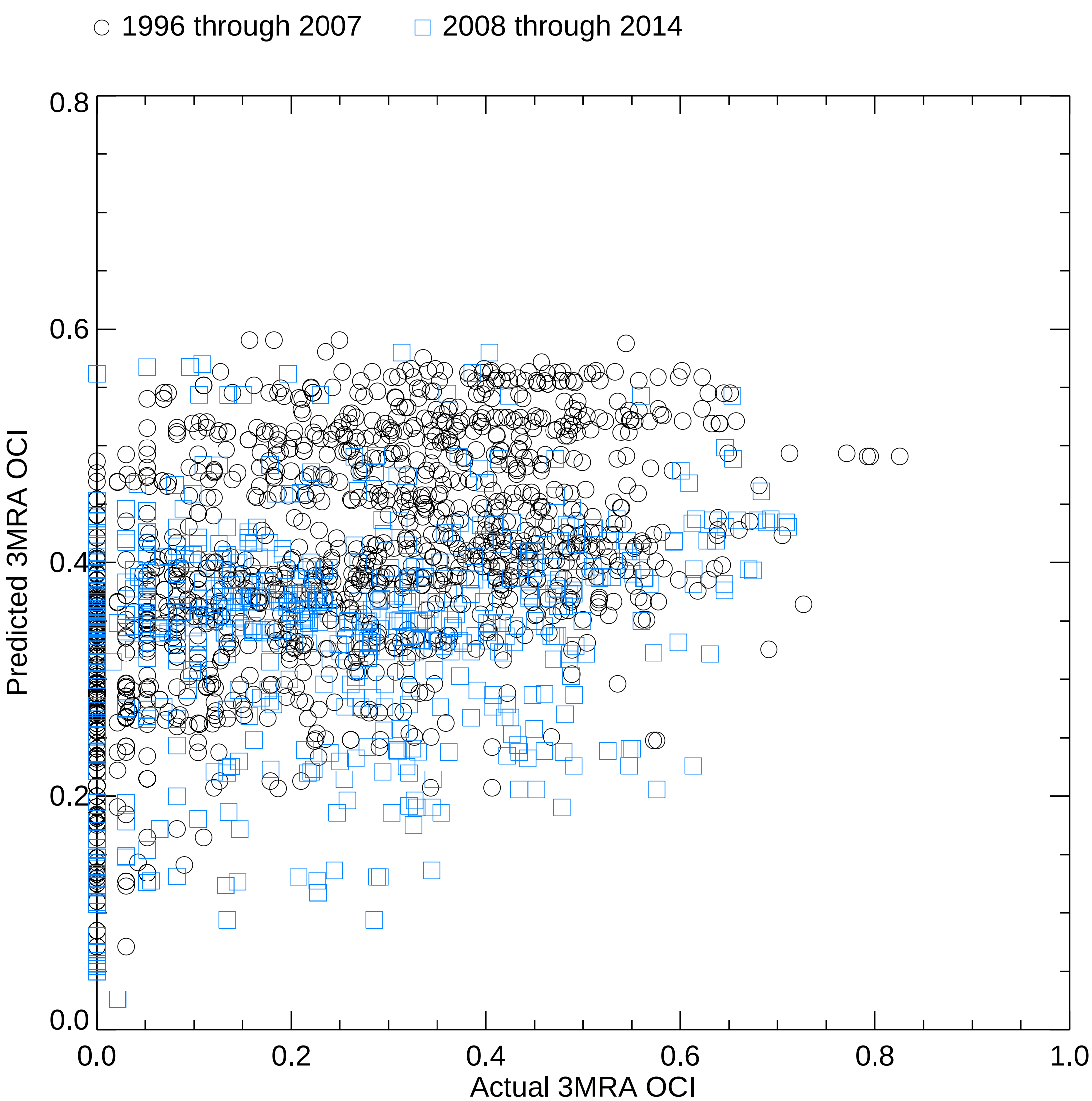

Figure 2-18

New and Old Data Results for Predicted vs. Actual 3MRA OCI for the MBHE 2008 Model

Data source: TPWD; OysterSentinel.

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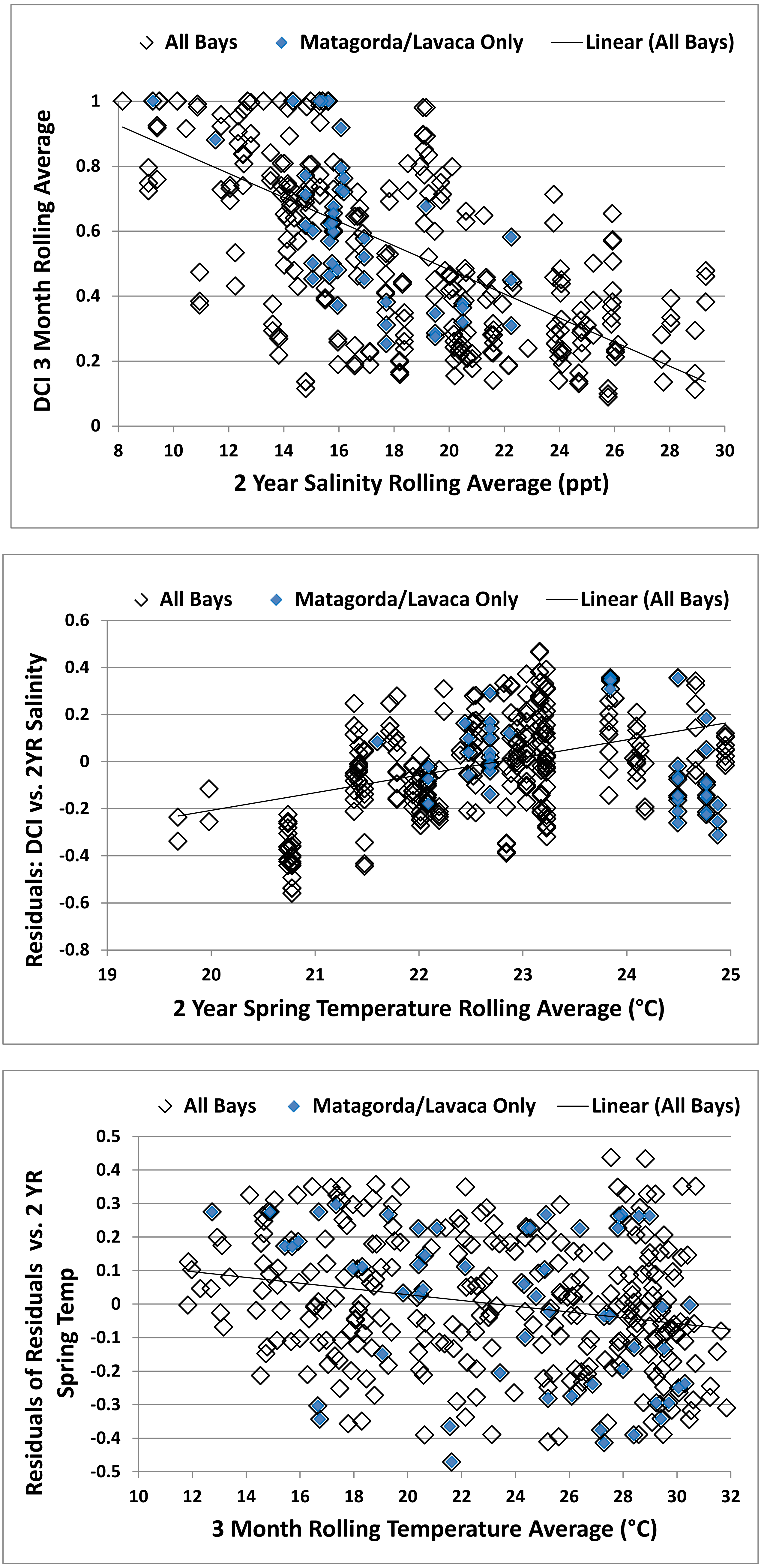

Notes:
The MBHE 2008 suitability criterion multiple regression model for 3MRA DCI is shown in this figure
Top: Linear relationship between 3MRA DCI and the most predictive salinity term: two-year salinity rolling average
Middle: Residuals from the top panel regression vs. two-year spring temperature average
Bottom: Residuals from the middle panel regression vs. 3MRA temperature

Figure 2-19
MBHE 2008 Suitability Criterion Multiple Regression Model for 3MRA DCI
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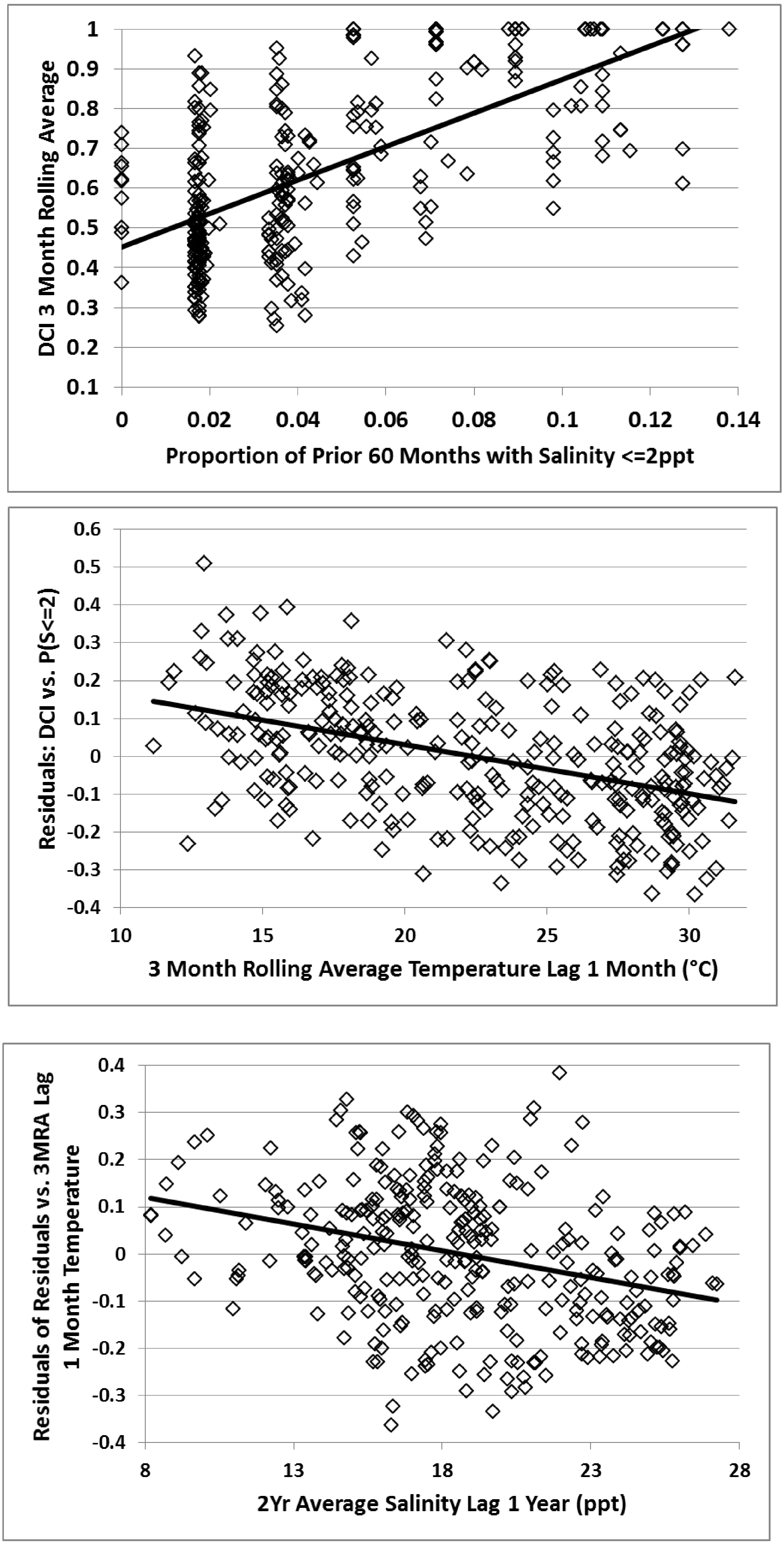

Notes:
Top: Linear relationship between 3MRA DCI and proportion of months with salinity ≤ 2
Middle: Residuals from the top panel regression vs. 3MRA temperature lagged 1 month
Bottom: Residuals from the middle panel regression vs. 2-year rolling salinity average lagged one year

Figure 2-20
Updated Suitability Criterion Multiple Regression Model for 3MRA DCI
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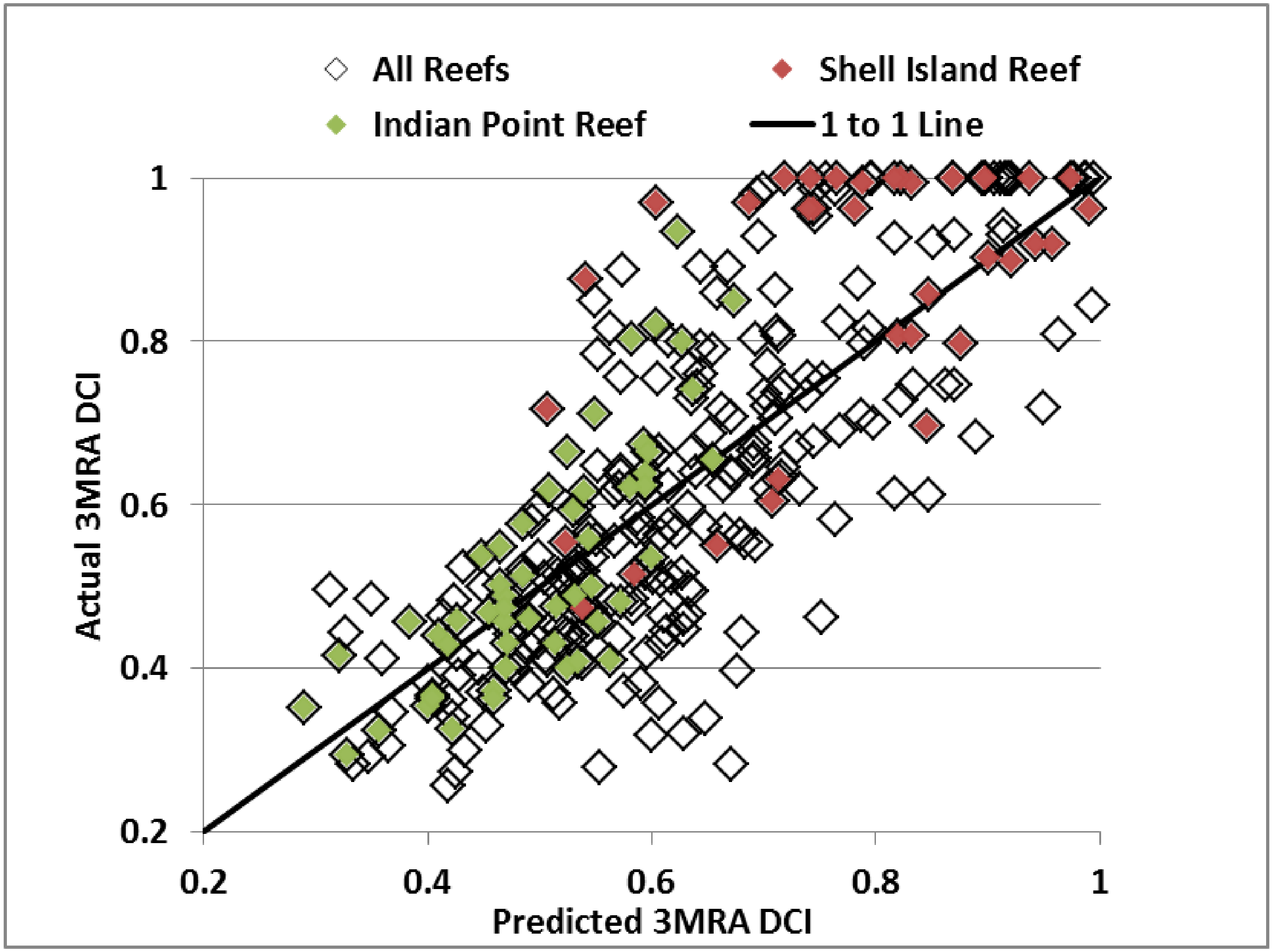

Note:
Results Highlighted for Shell Island Reef and Indian Point Reef

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Figure 2-21
Predicted vs. Actual 3MRA DCI for the Updated Suitability Criterion Model
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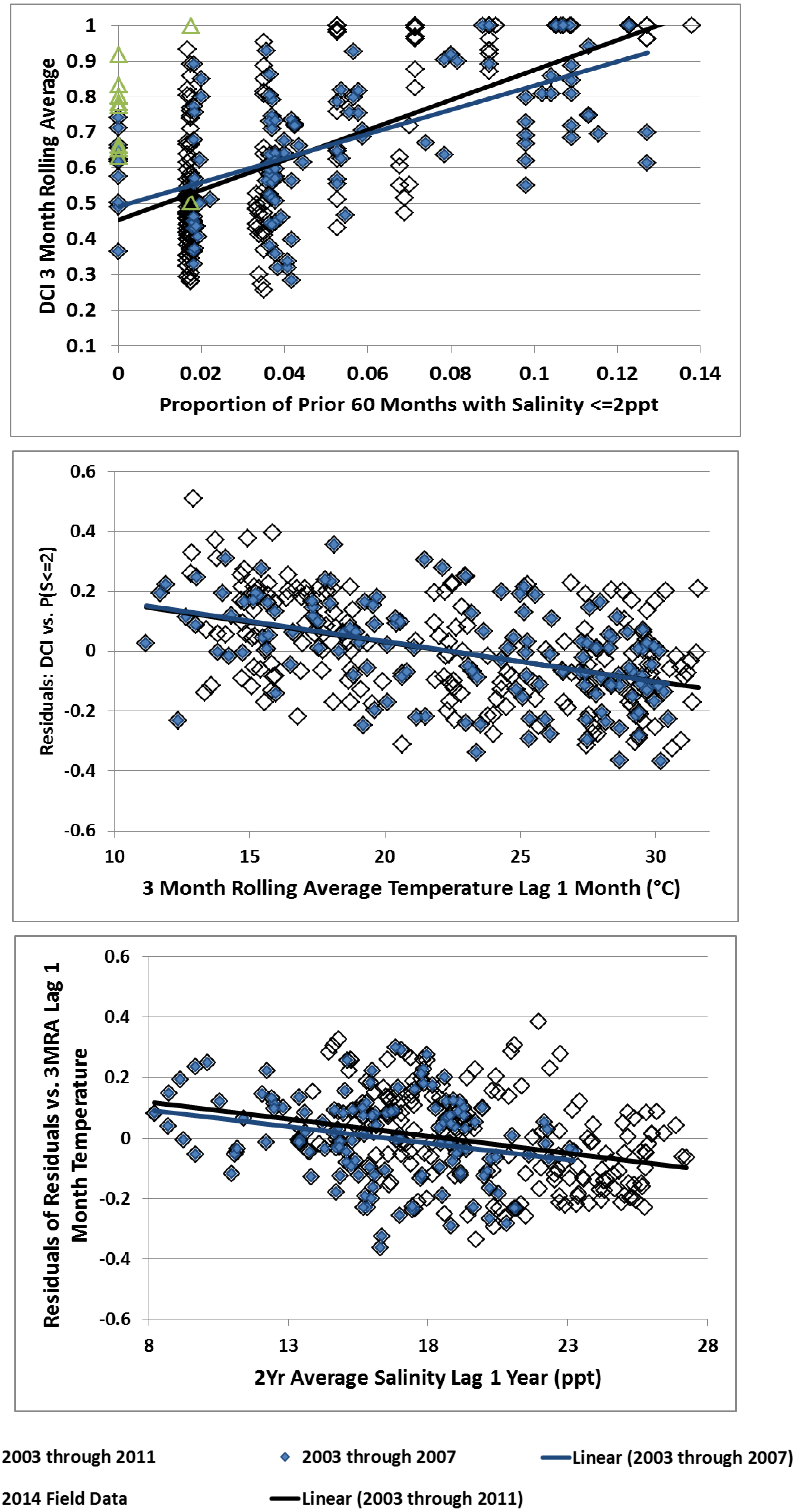

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Figure 2-22
Updated Suitability Criterion Multiple Regression Model for 3MRA DCI Showing Results for New and Old Data and for 2014 Field Data
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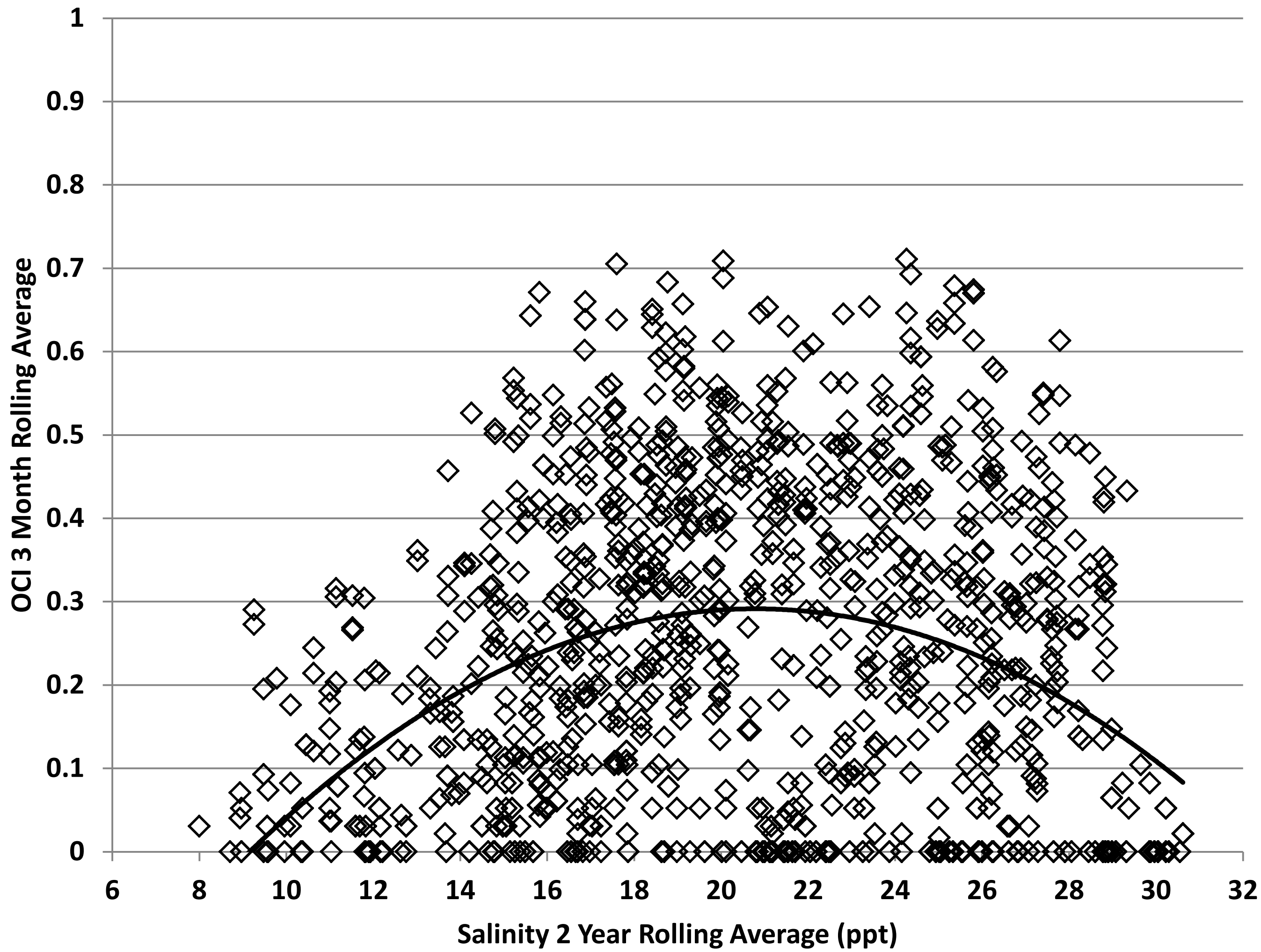

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Figure 2-23
Updated OCI 3MRA vs. 2-Year Rolling Salinity Average
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Figure 3-1
Marsh Productivity Sampling Locations in the Lavaca River Delta
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Figure 3-2
Marsh Productivity Sampling Locations in the Colorado River Delta
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Figure 3-3
Marsh Biomass Field Sampling
Freshwater Inflow Standards and Ecological Response
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Figure 3-4
1m^2 Throw Trap Sampler Used in the 2014 Field Study
Freshwater Inflow Standards and Ecological Response
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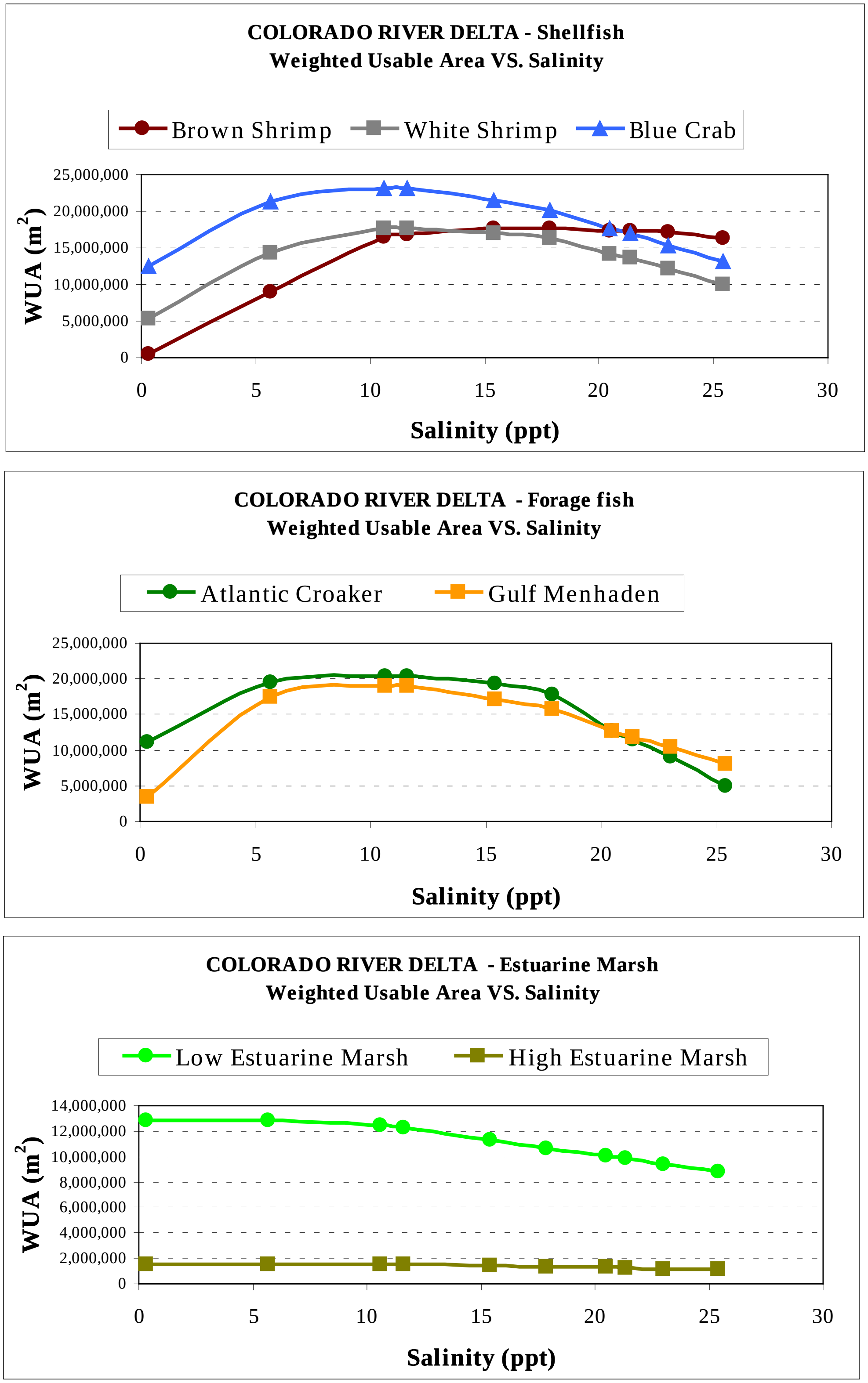

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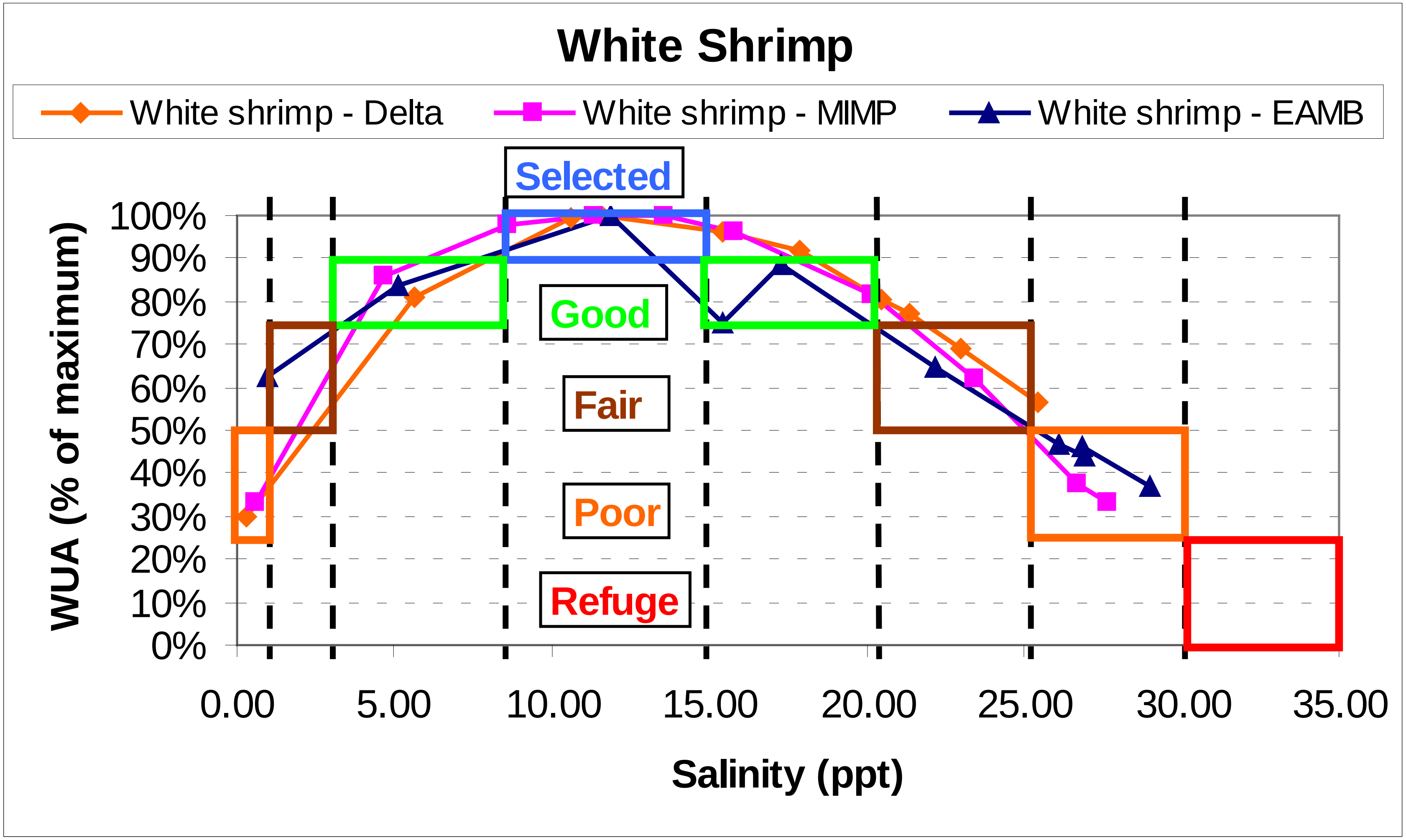
Figure 3-5
Weighted Usable Area to Salinity Relationships for Individual Trophic Levels in the Colorado River Delta (MBHE 2008a)



Note: Percentage of Maximum WUA for white shrimp in the Colorado River Delta (Delta), Mad Island Marsh Preserve (MIMP), and Eastern Arm of Matagorda Bay (EAMB)

Reference: MBHE 2008a

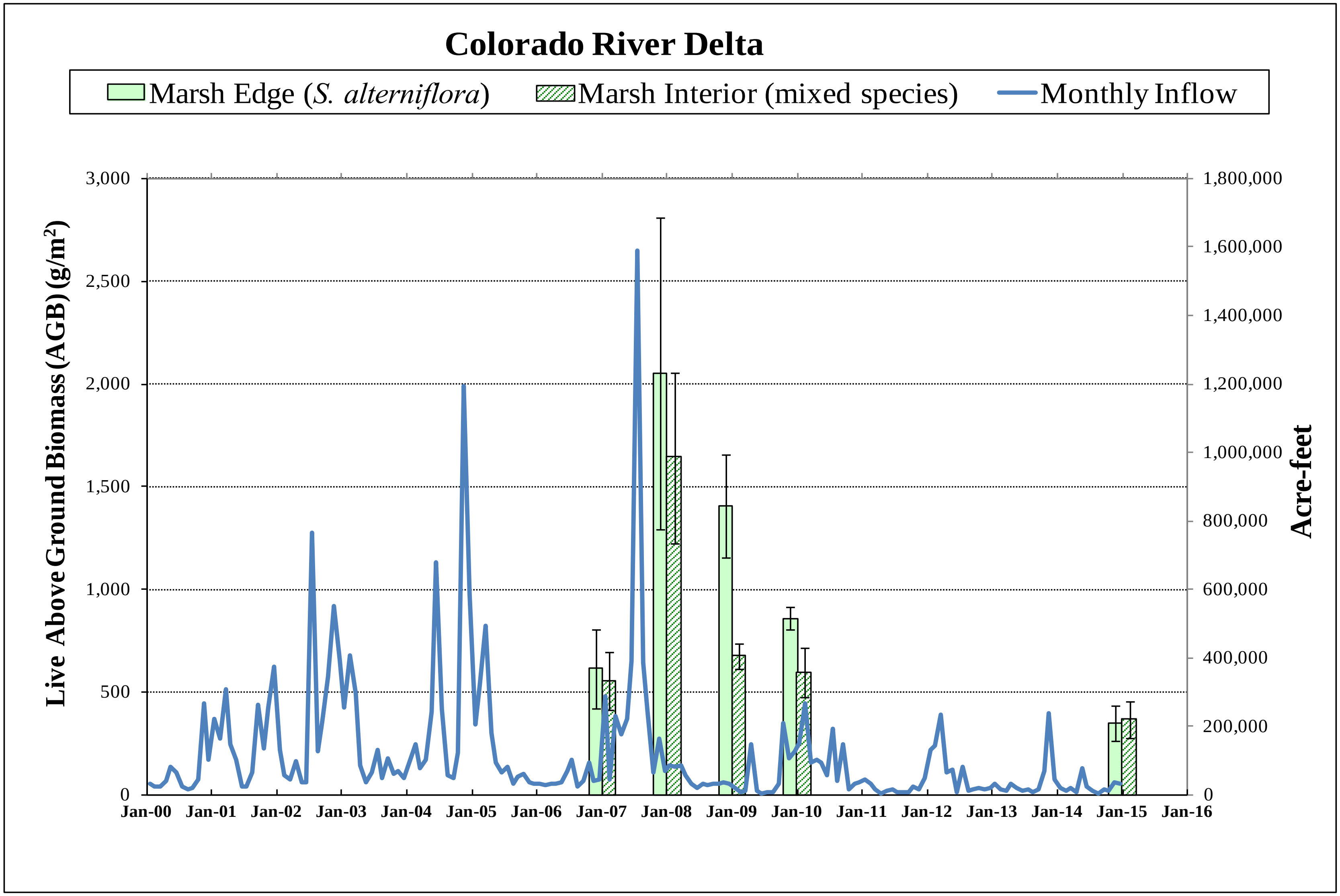


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Figure 3-7
Marsh Vegetation Biomass (Mean ± 1 STD) Over Time in the Colorado River Delta
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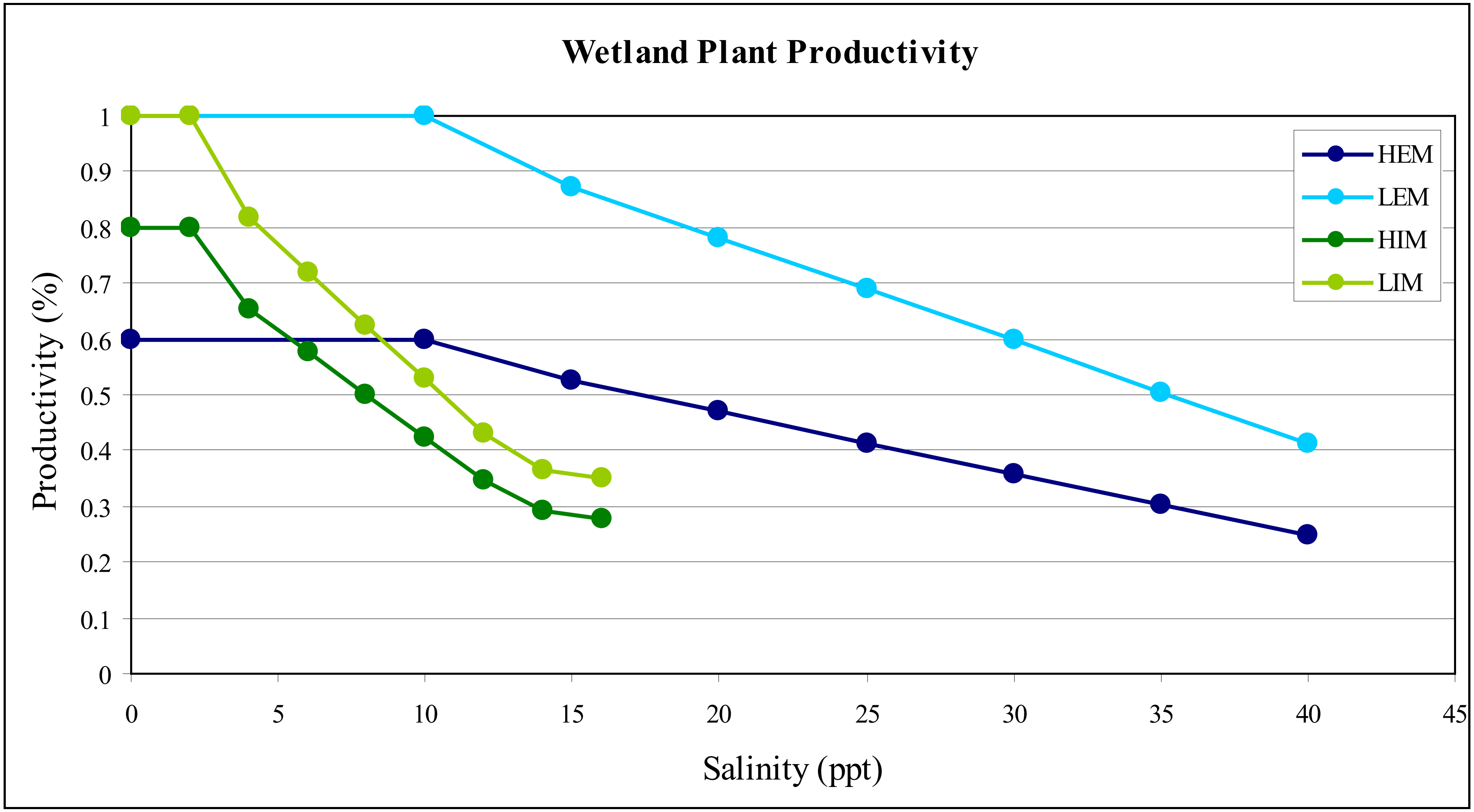

Note: Wetland plant productivity (expressed as the percent of maximum potential productivity) at a range of salinity conditions for low estuarine marsh (LEM). LEM is based on 50% inundation frequency. Reference: MBHE 2006c

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Figure 3-8
Wetland Plant Productivity
Freshwater Inflow Standards and Ecological Response
TWDB

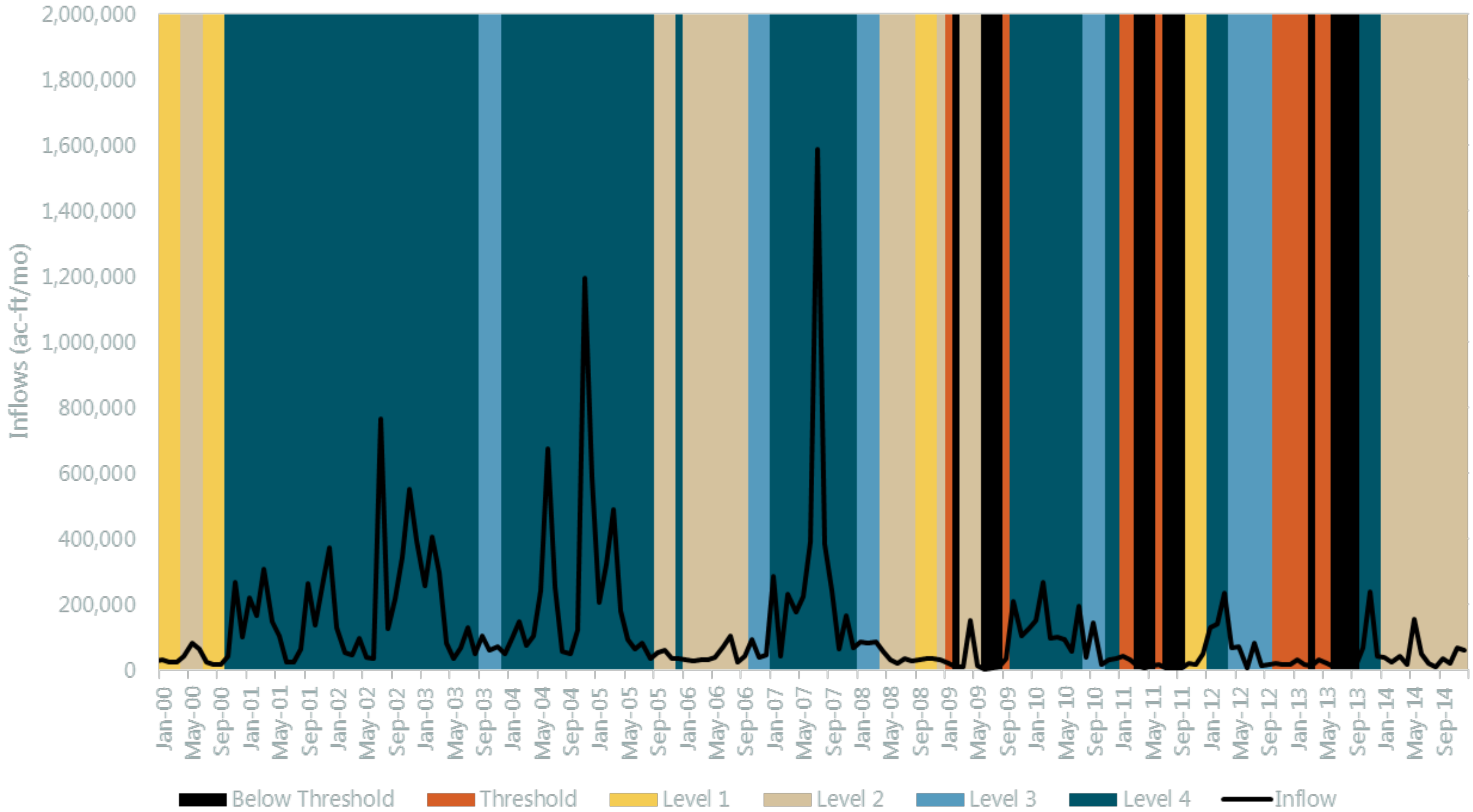
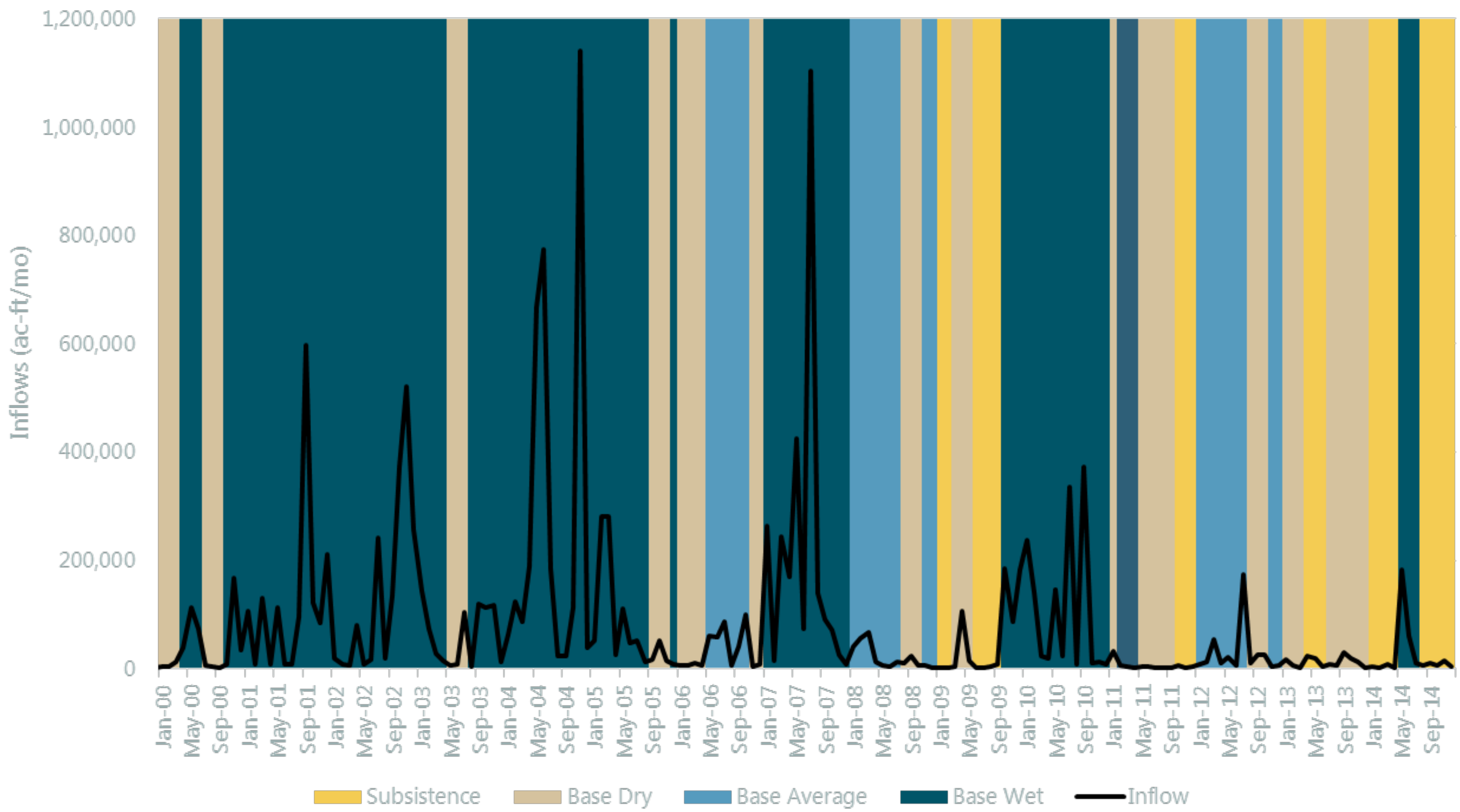


Figure 3-9
Colorado River Inflows and Resulting MBHE Criteria Levels
Freshwater Inflow Standards and Ecological Response
TWDB



ANCHOR QEA

Figure 3-10
Lavaca River and Garcitas Creek Inflows and Resulting Inflow Criteria Levels
Freshwater Inflow Standards and Ecological Response
TWDB

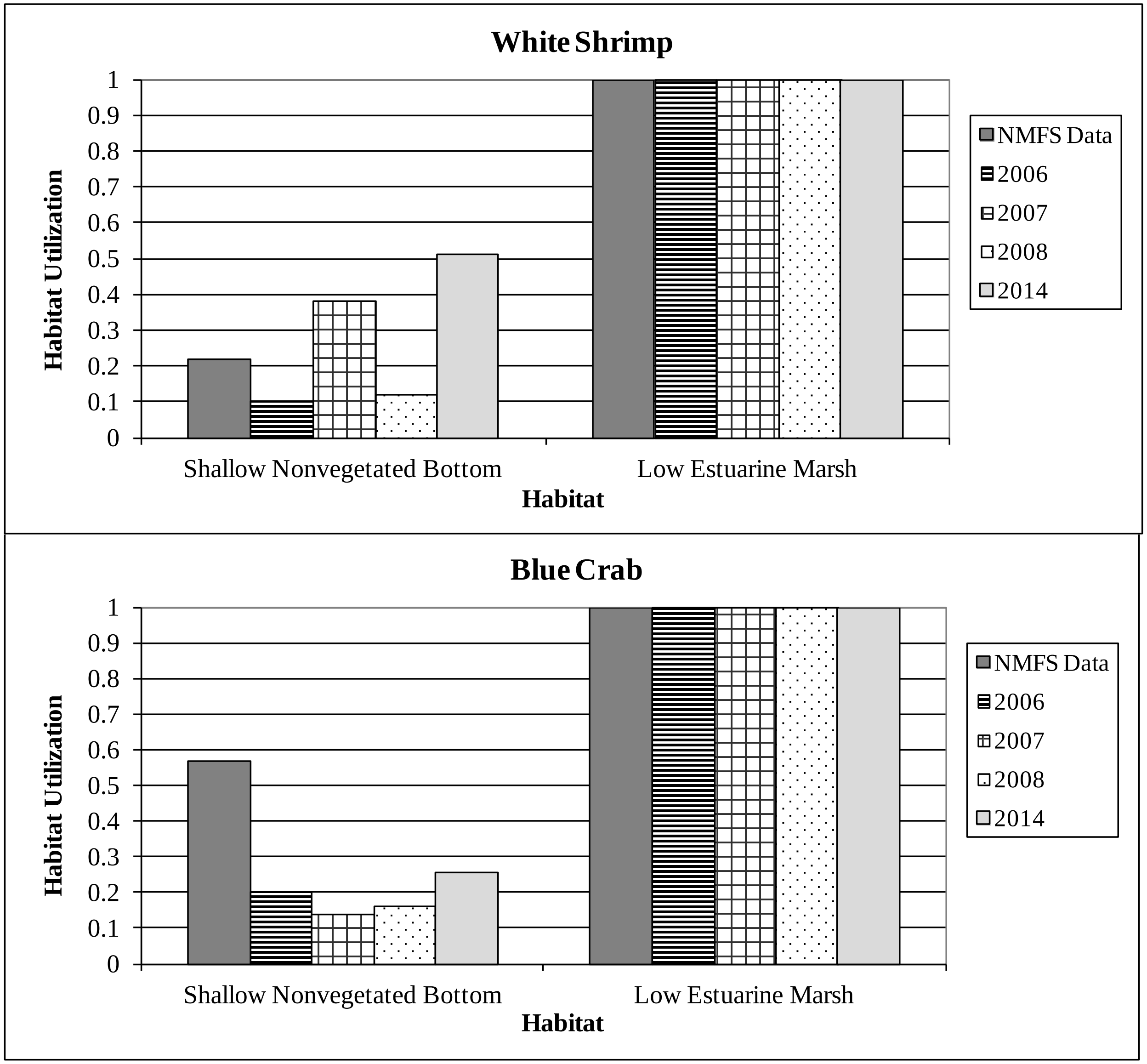


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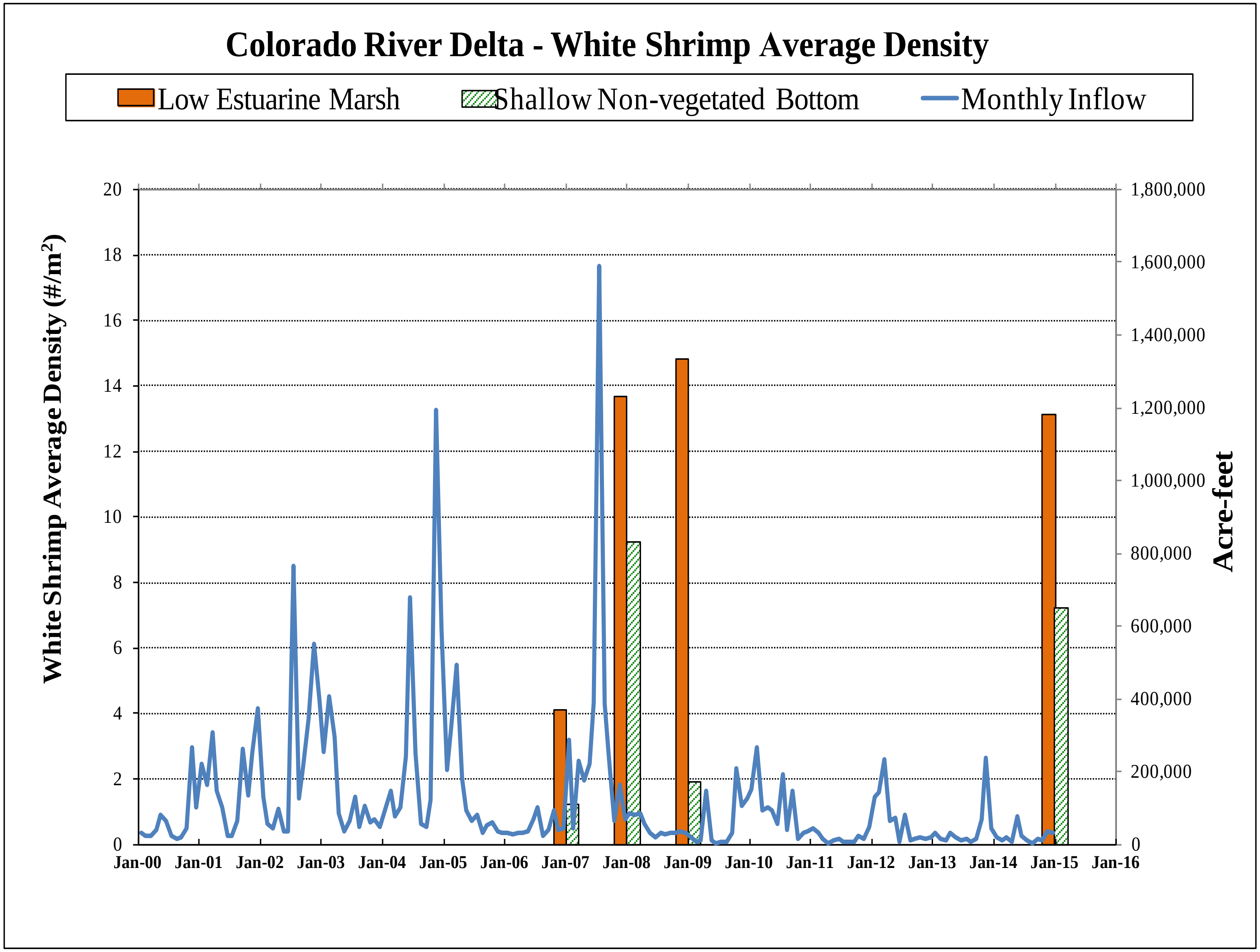
Figure 3-11
Habitat Utilization of Shallow Nonvegetated Bottom and Low Estuarine Marsh
Freshwater Inflow Standards and Ecological Response
TWDB



ANCHOR QEA

Figure 3-12
White Shrimp Average Density in Throw Trap Sampling Over Time in the Colorado River Delta
Freshwater Inflow Standards and Ecological Response
TWDB

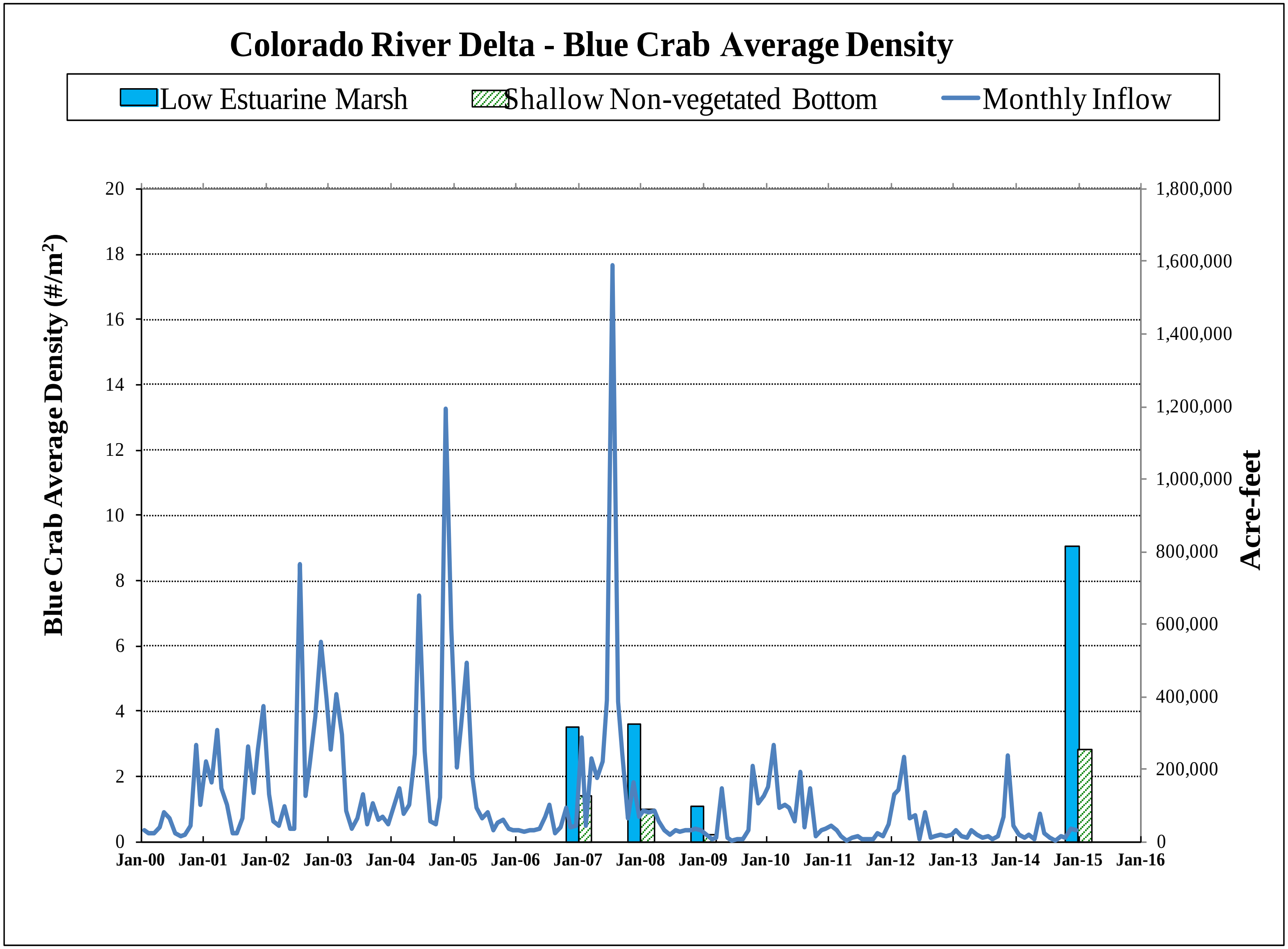


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Figure 3-13

Blue Crab Average Density in Throw Trap Sampling Over Time in the Colorado River Delta

Freshwater Inflow Standards and Ecological Response

TWDB

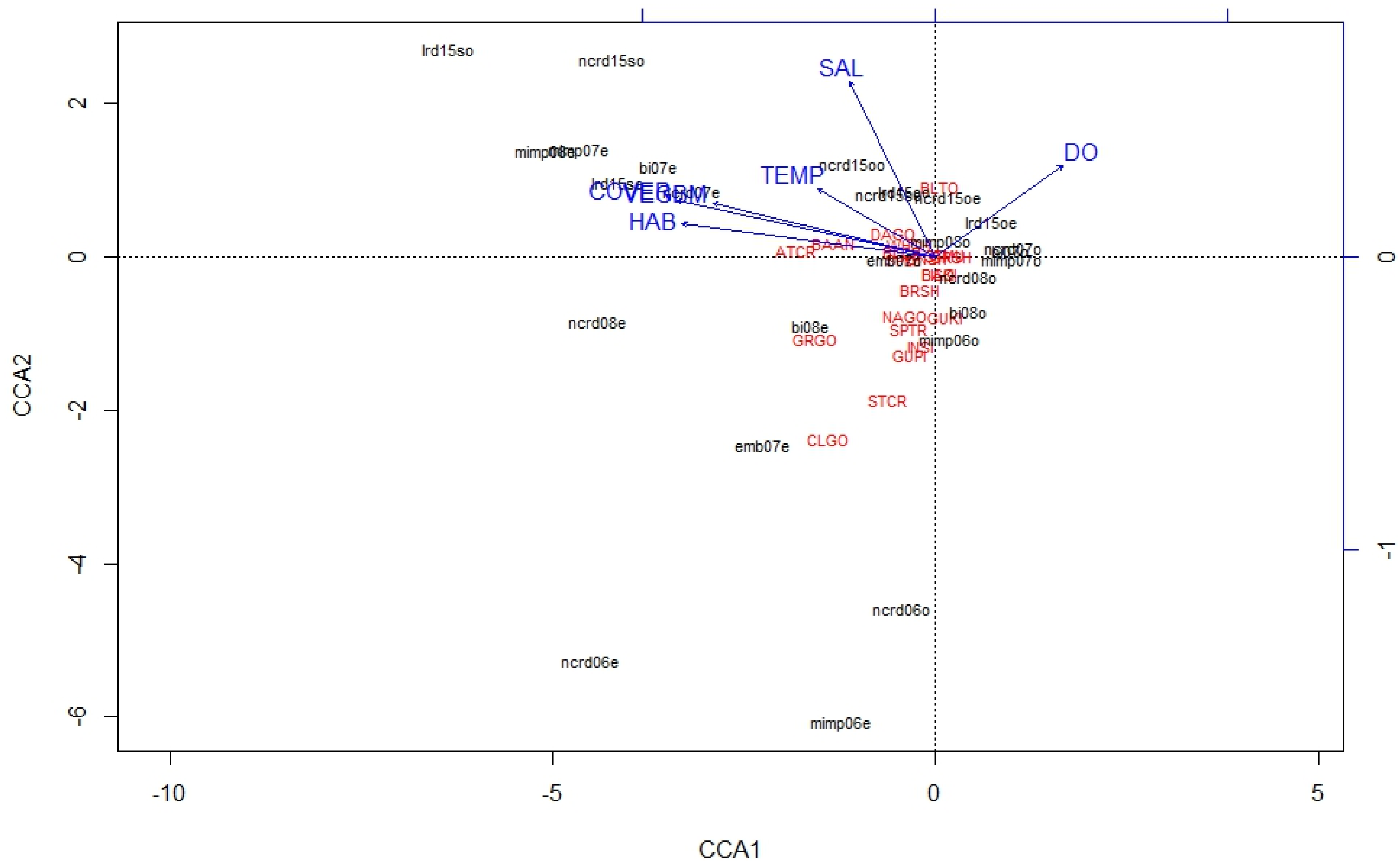

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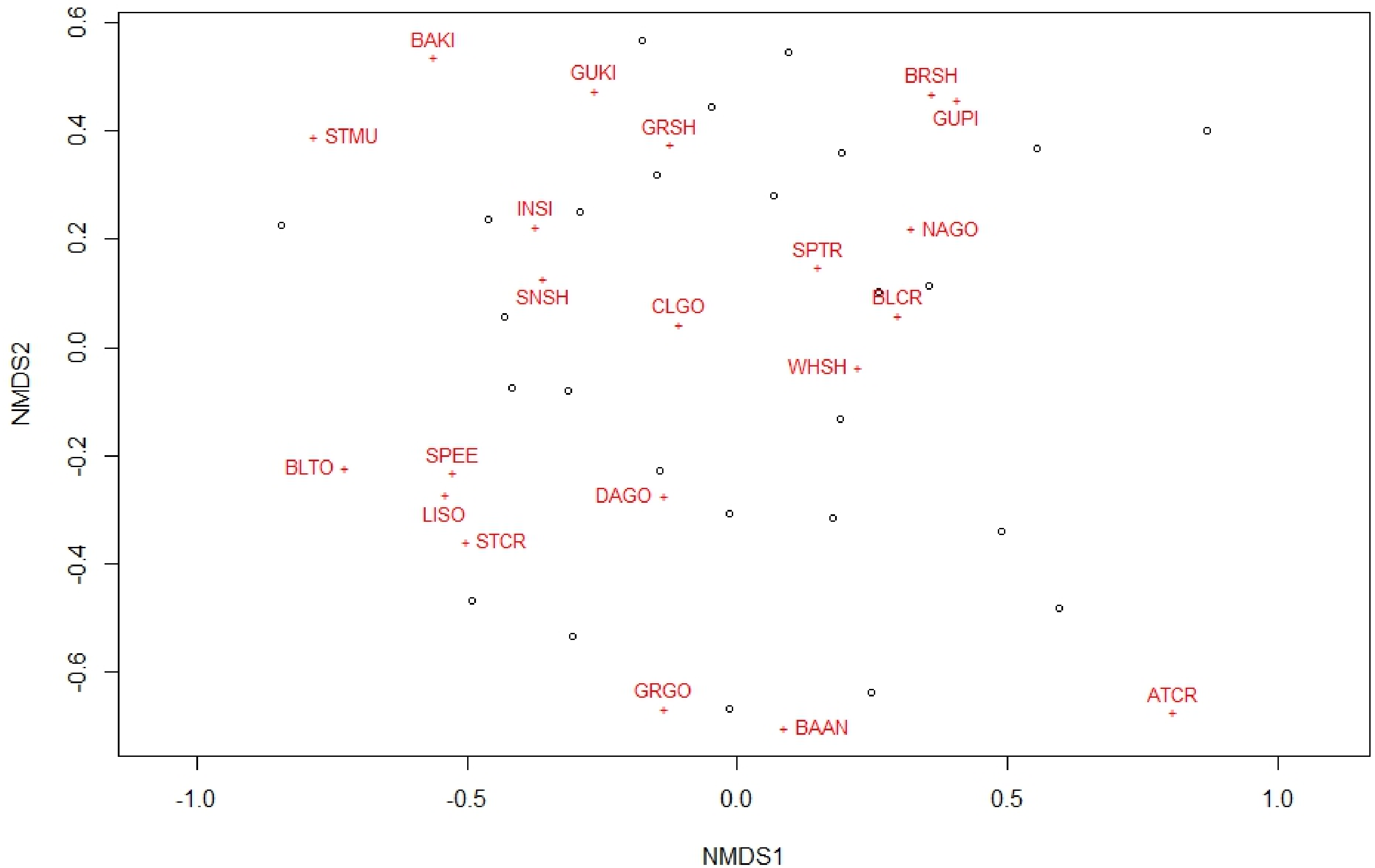
Figure 3-14
Plot of CCA Results from Analysis Including all Variables and Sites
Freshwater Inflow Standards and Ecological Response
TWDB



Note: Species are symbolized by red plus marks, and samples are symbolized by black circles.

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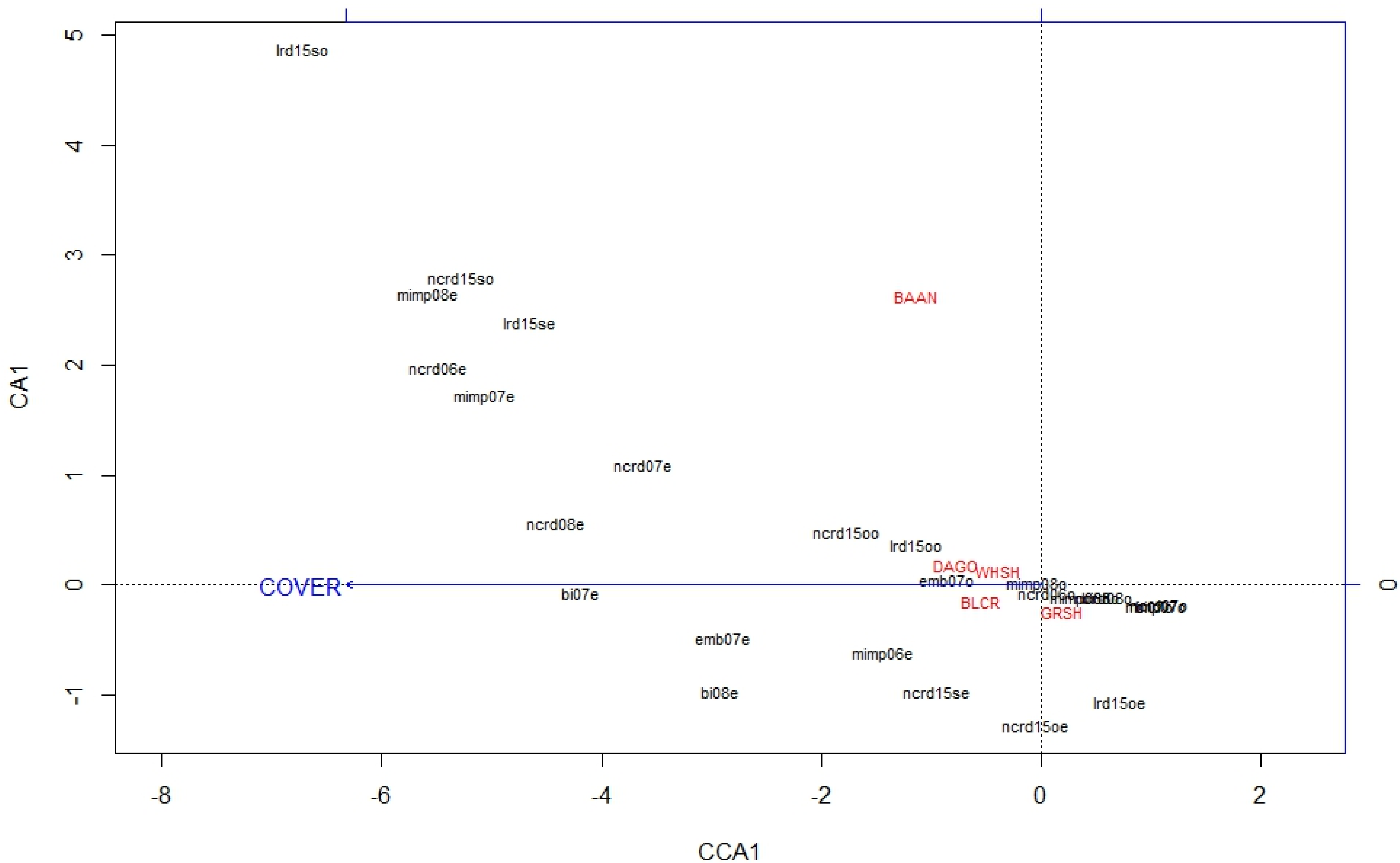
Figure 3-15
Non Metric Multidimensional Scaling Analysis (NMDS) Plot of the Complete Data (All Sites and Species)
Freshwater Inflow Standards and Ecological Response
TWDB



Note: Species are symbolized by red plus marks, and samples are symbolized by black circles.

Figure 3-16
CCA of Reduced Data Including Only Species with Over 100 Observations
Across All Samples and Percent Vegetative Cover
Freshwater Inflow Standards and Ecological Response
TWDB

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Figure 4-1
Rangia Search Areas in the Colorado River Delta
Freshwater Inflow Standards and Ecological Response
TWDB

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Figure 4-2
Rangia Search Areas in the Lavaca River Delta
Freshwater Inflow Standards and Ecological Response
TWDB

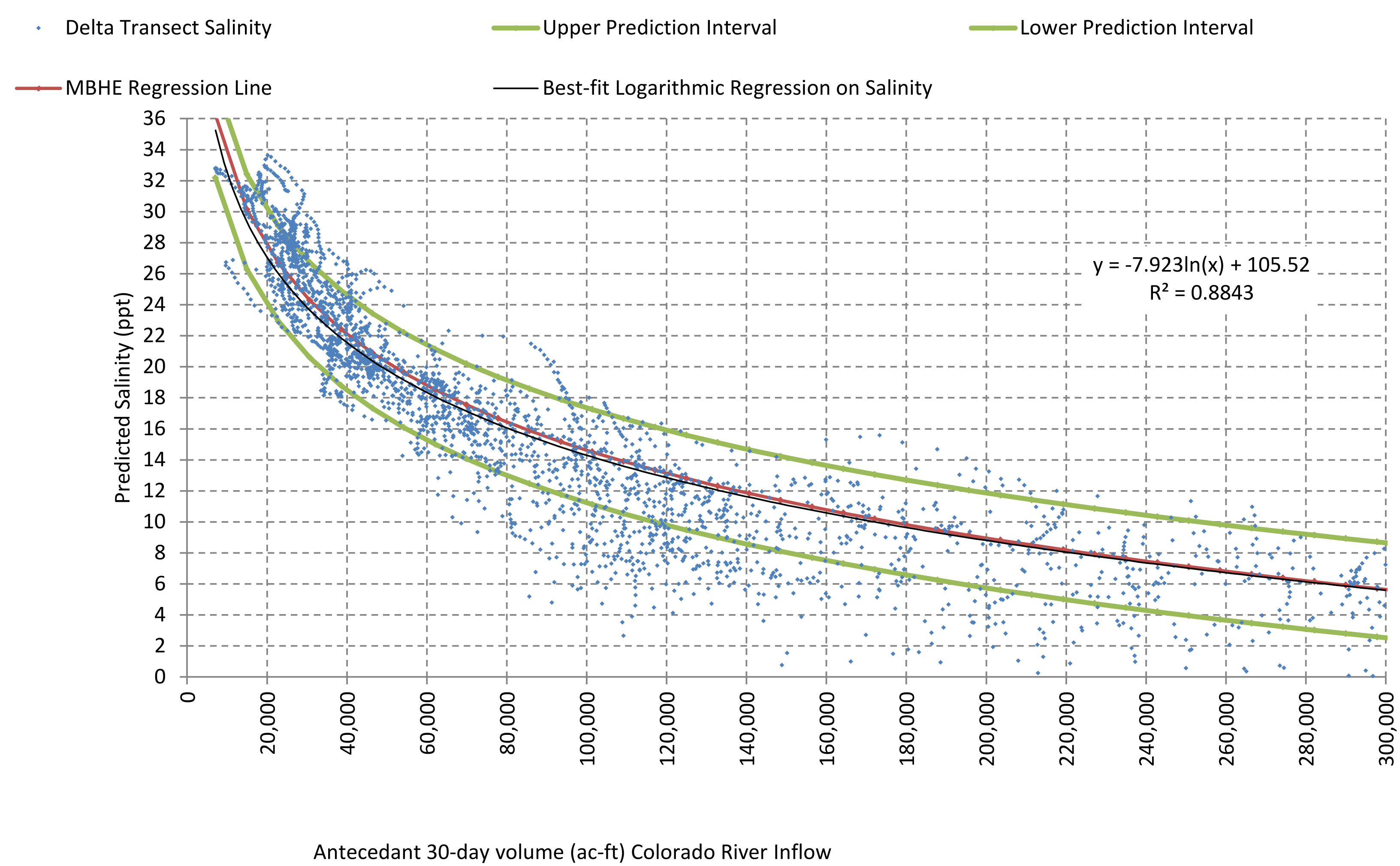

Figure 5-1
Delta Edge Transect, Predicted Salinity vs. Antecedent 30-day Flow (ac-ft) with Regression Curve from MBHE, 2008a
Freshwater Inflow Standards and Ecological Response
TWDB

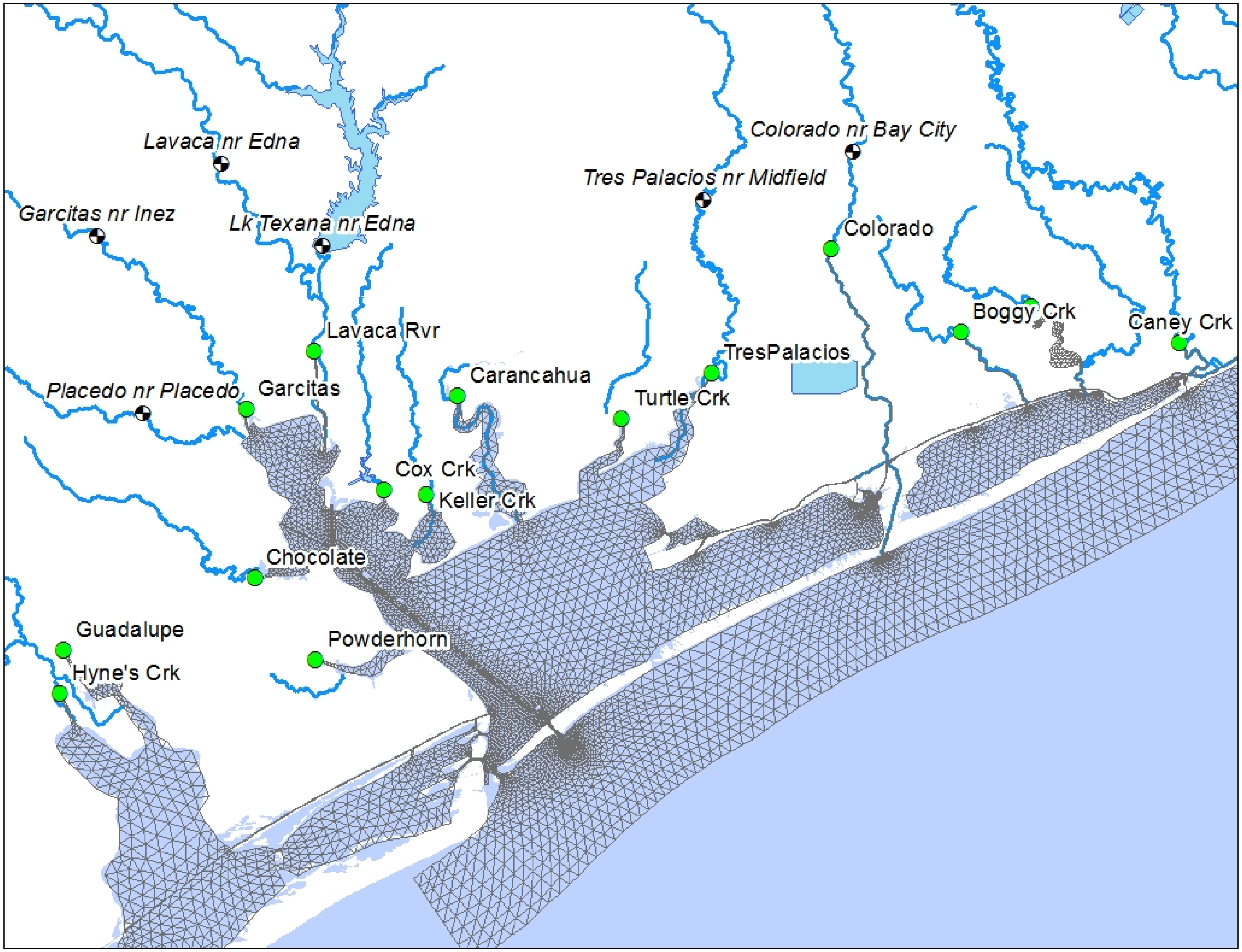

Figure 5-2
Map of Matagorda Bay TxBLEND Model Mesh
Freshwater Inflow Standards and Ecological Response
TWDB

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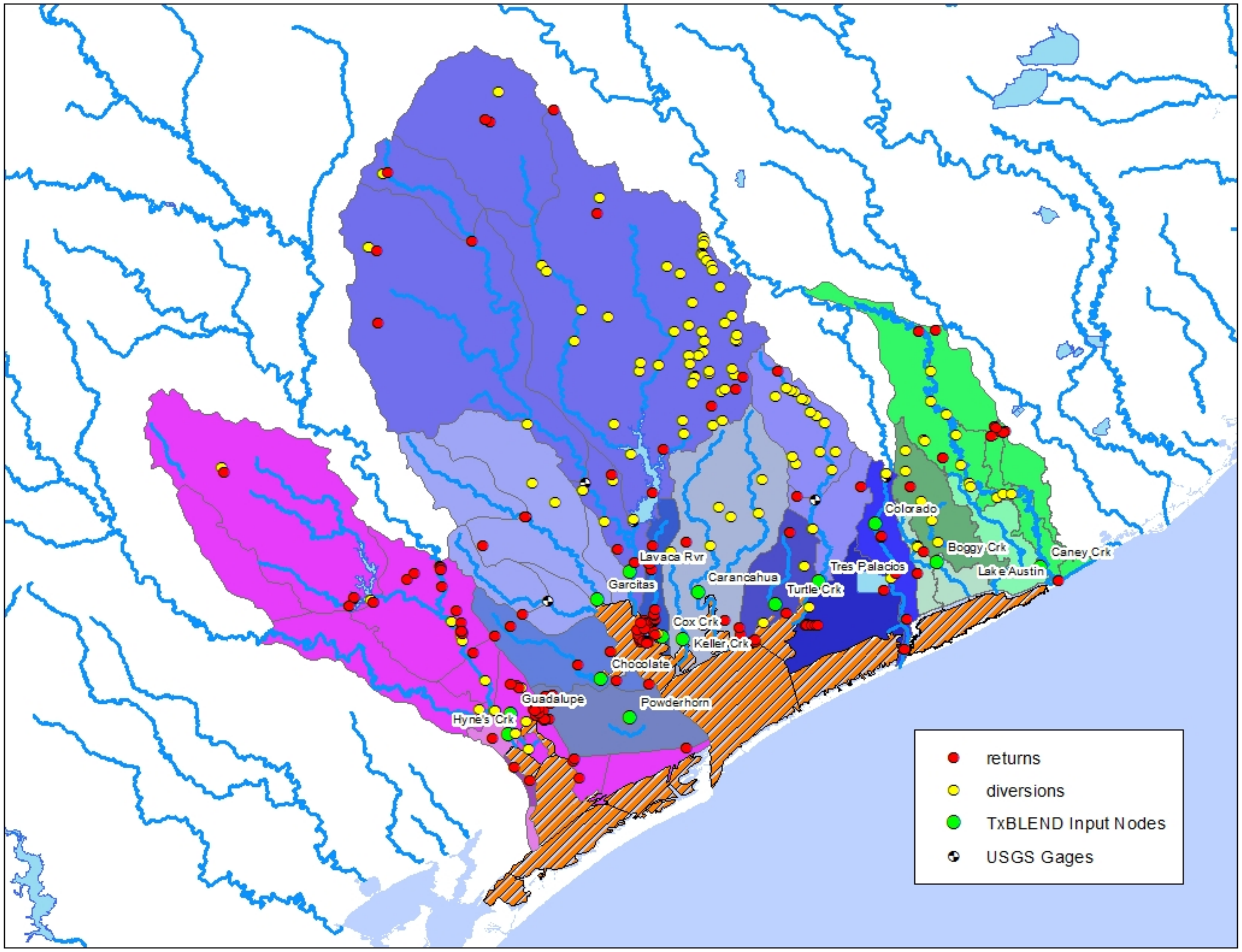

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Figure 5-3
Coastal Watersheds
Freshwater Inflow Standards and Ecological Response
TWDB

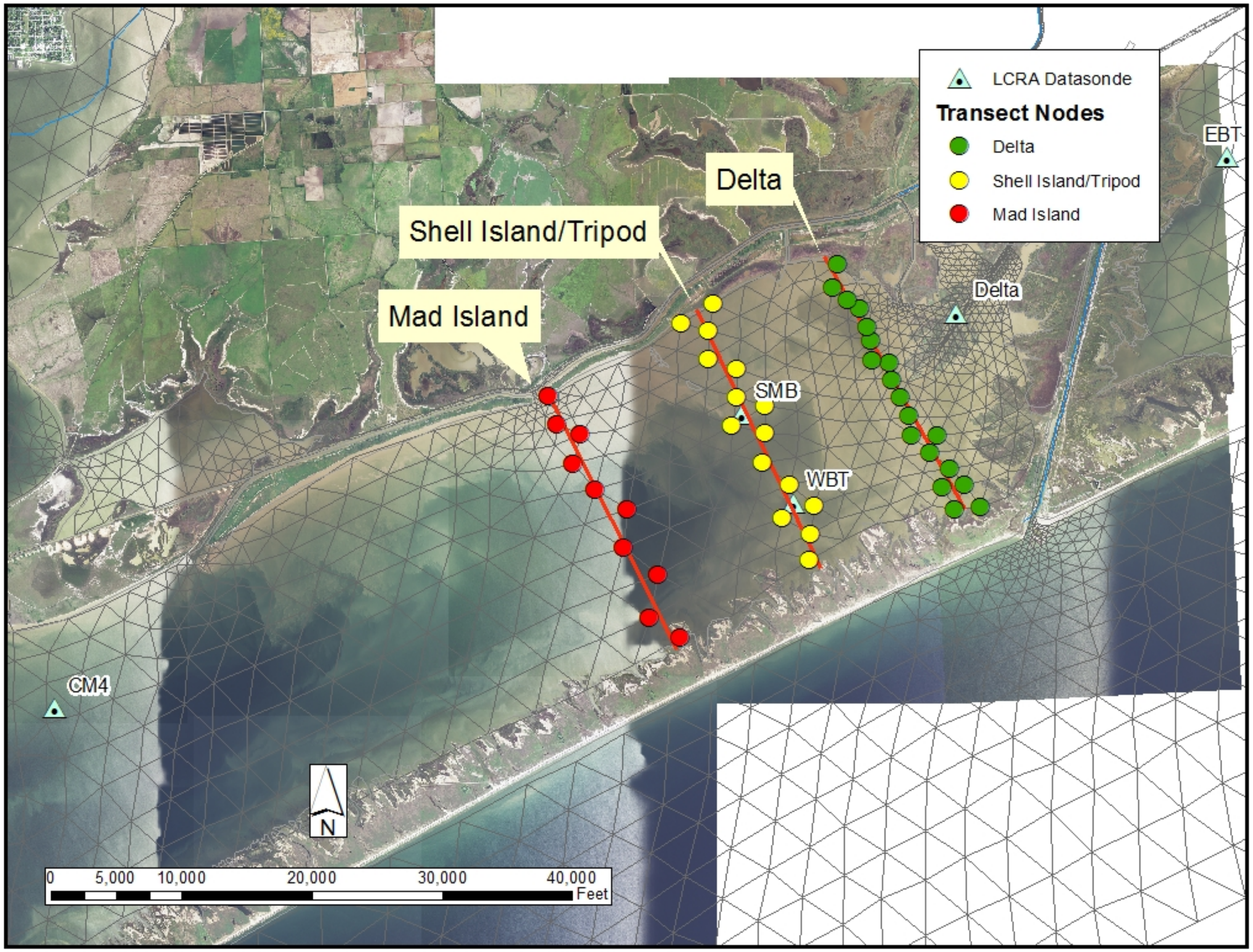

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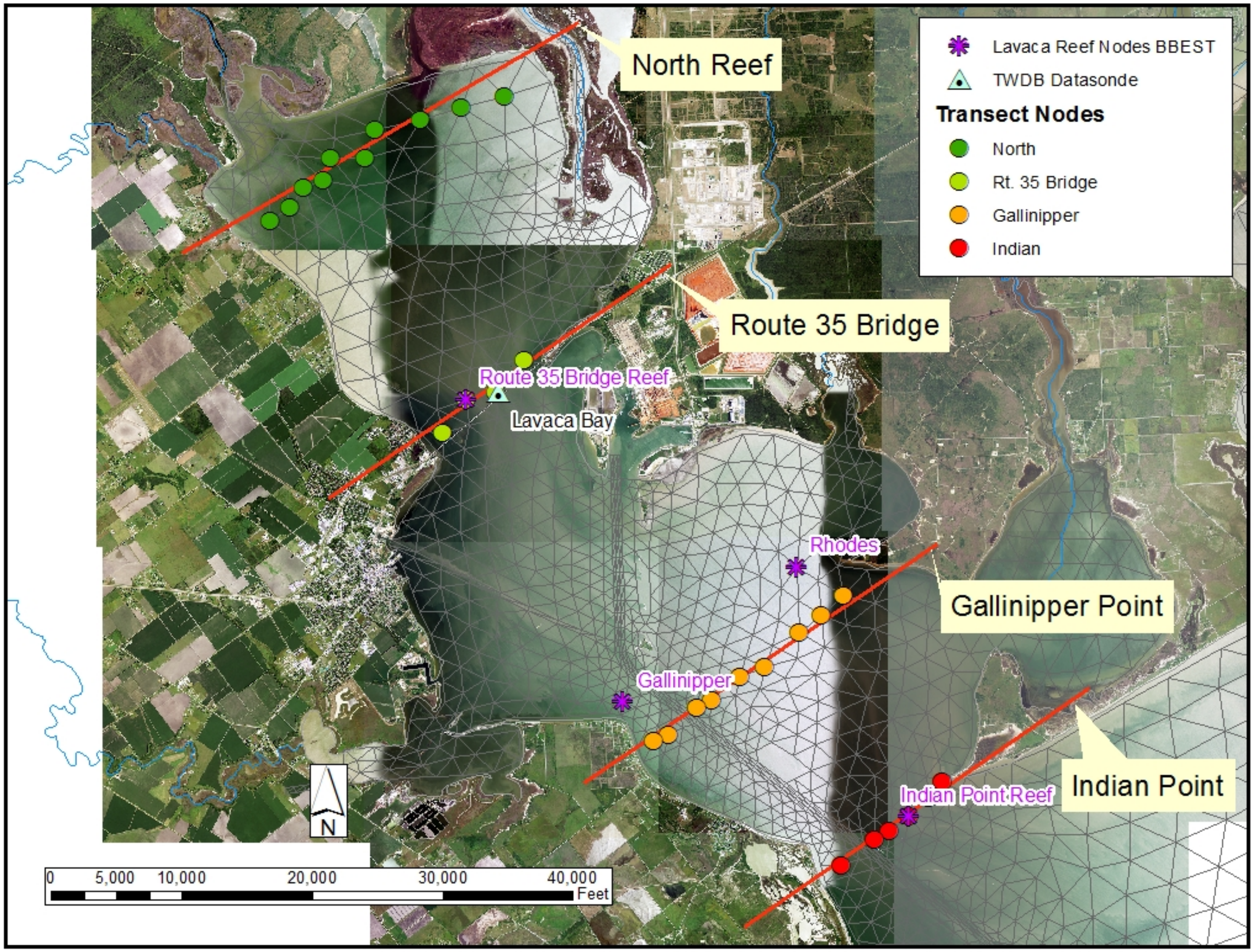
Figure 5-4
Matagorda Design Area Map
Freshwater Inflow Standards and Ecological Response
TWDB



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ANCHOR QEA

Figure 5-5
Lavaca Design Area Map
Freshwater Inflow Standards and Ecological Response
TWDB

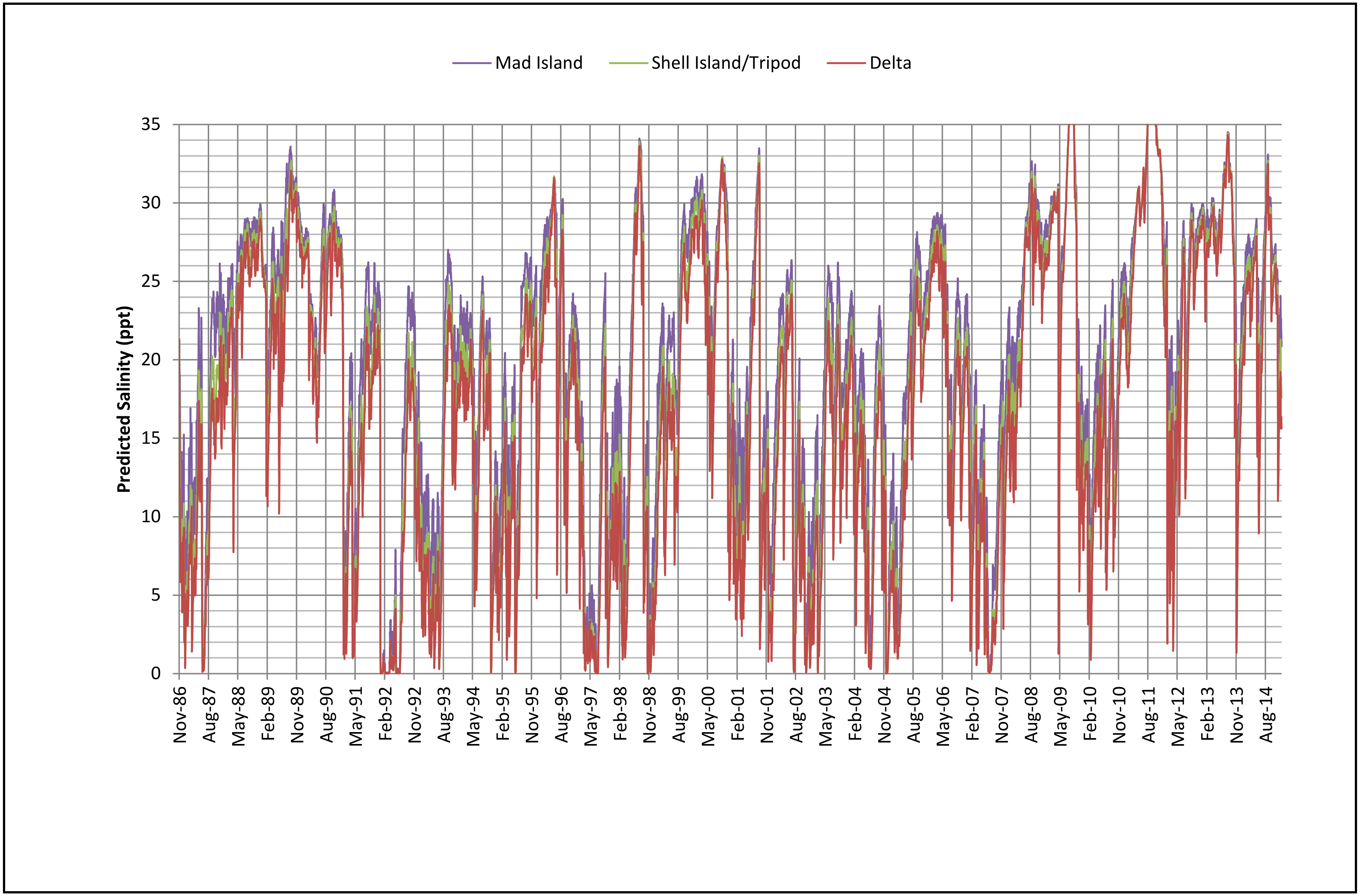

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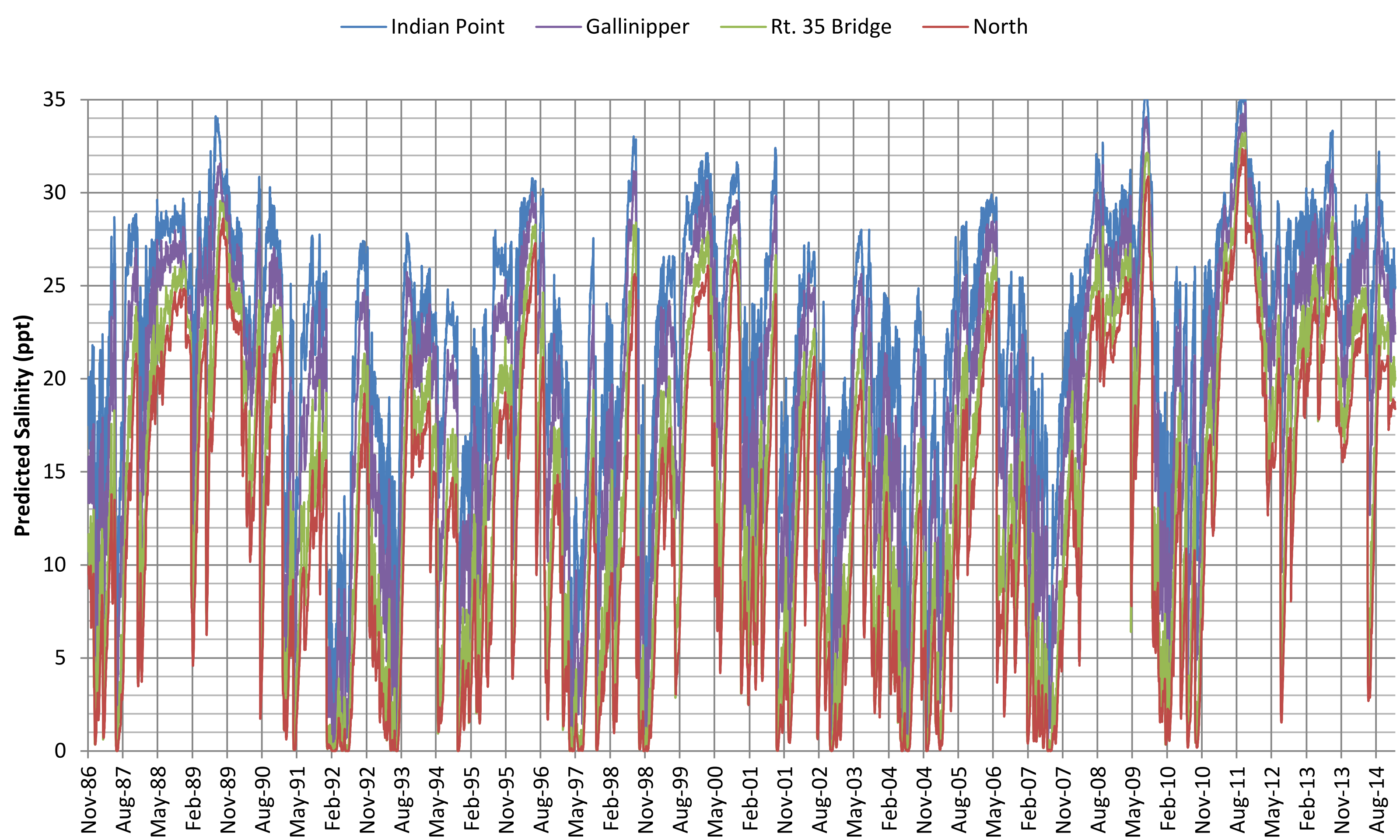
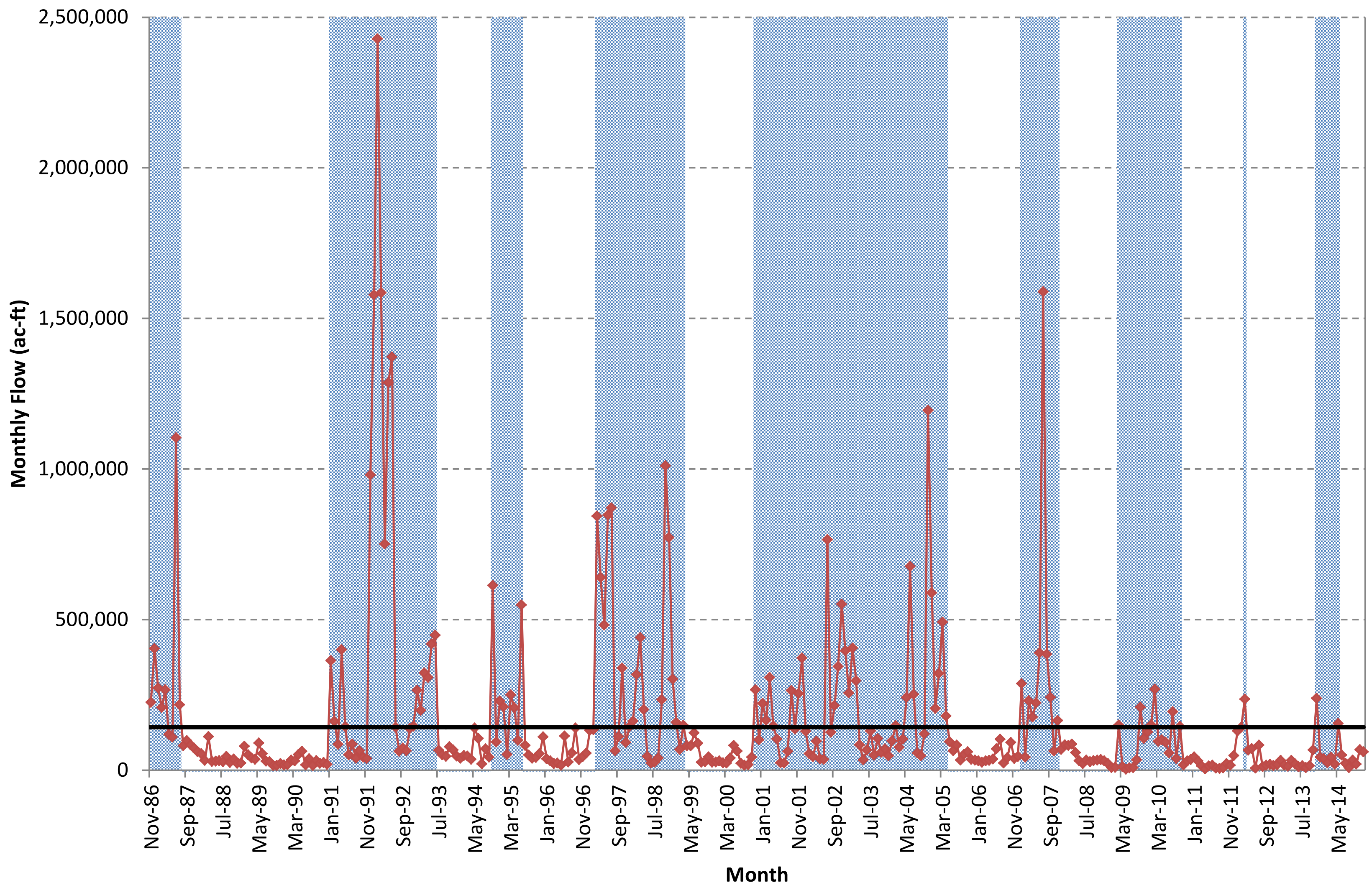
Figure 5-6
Predicted Salinity at Matagorda Design Area Transects for Simulation Period
Freshwater Inflow Standards and Ecological Response
TWDB



ANCHOR QEA

Figure 5-7
Predicted Salinity at Lavaca Design Area Transects for Simulation Period
Freshwater Inflow Standards and Ecological Response
TWDB



ANCHOR QEA

Figure 5-8
Colorado Inflows Over Simulation Period Classified as Normal (Shaded) and Low Flow Periods (White)
Freshwater Inflow Standards and Ecological Response
TWDB

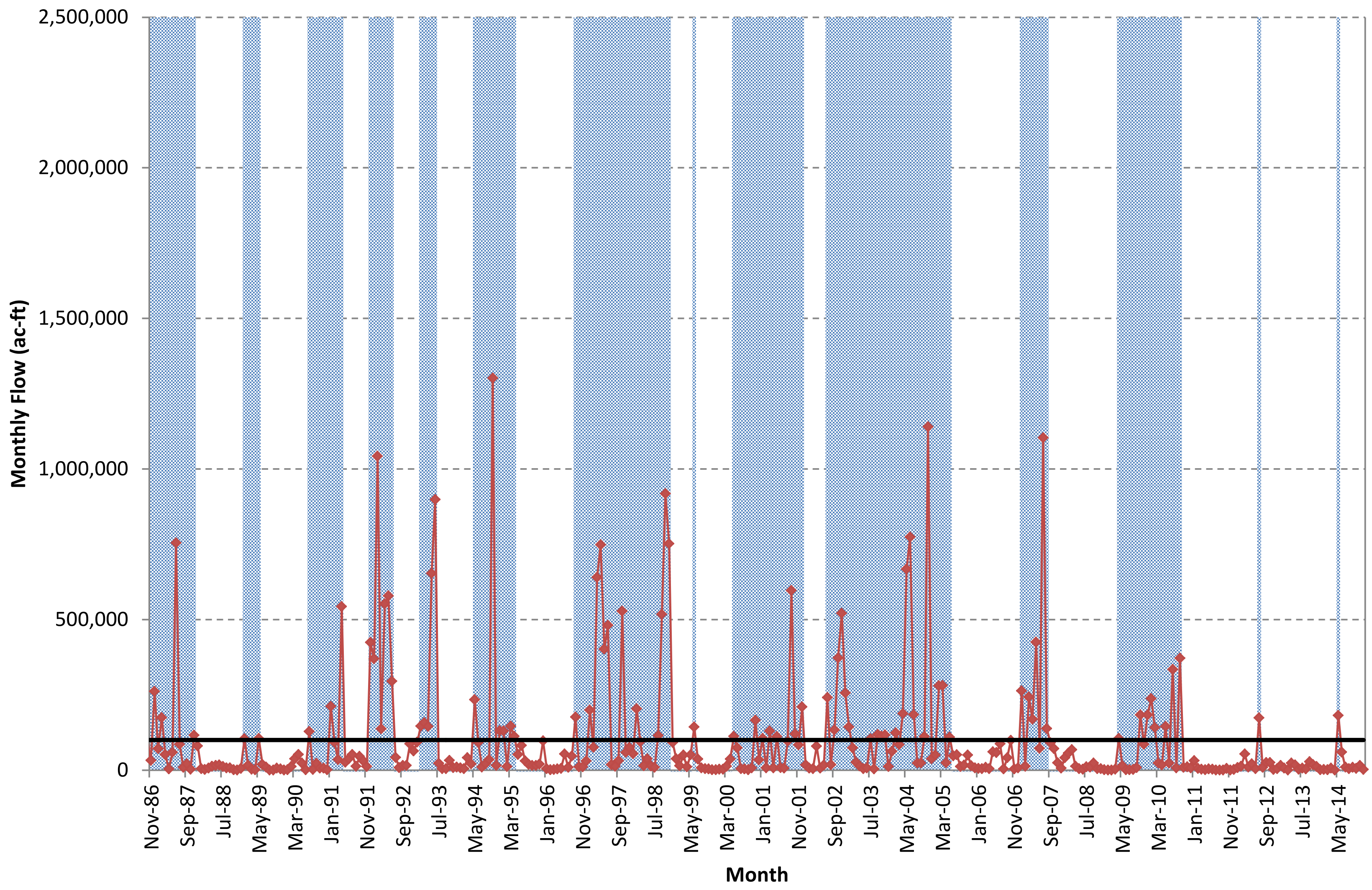


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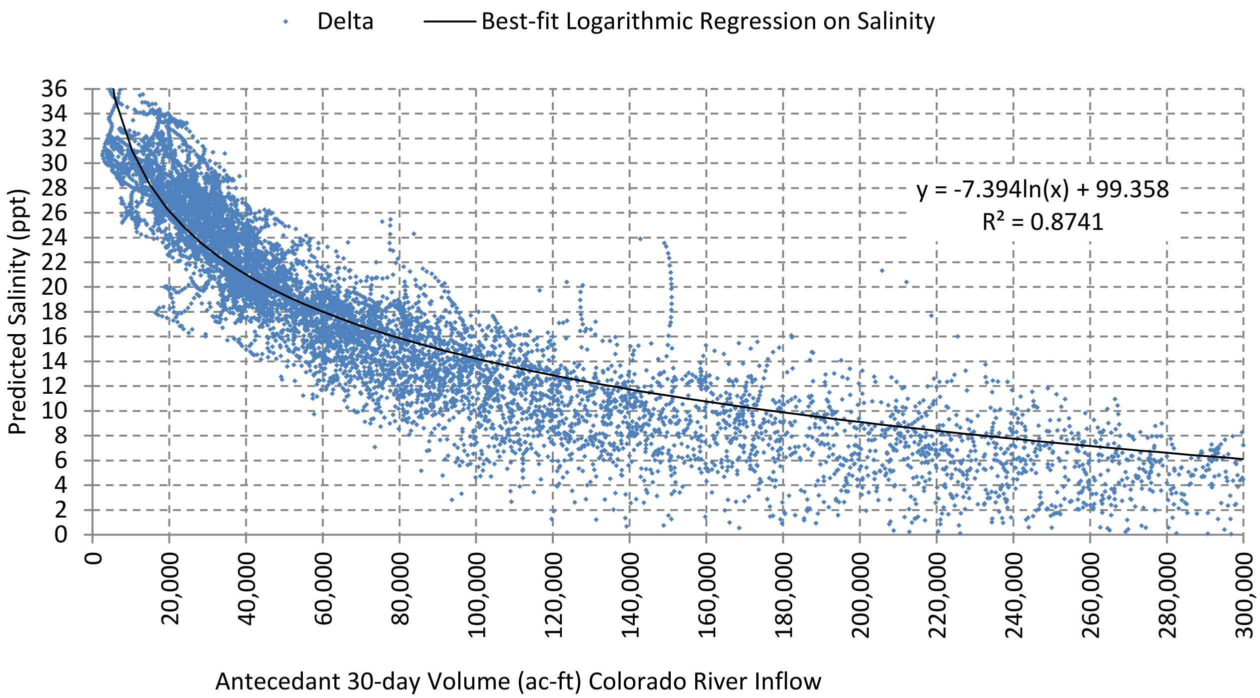
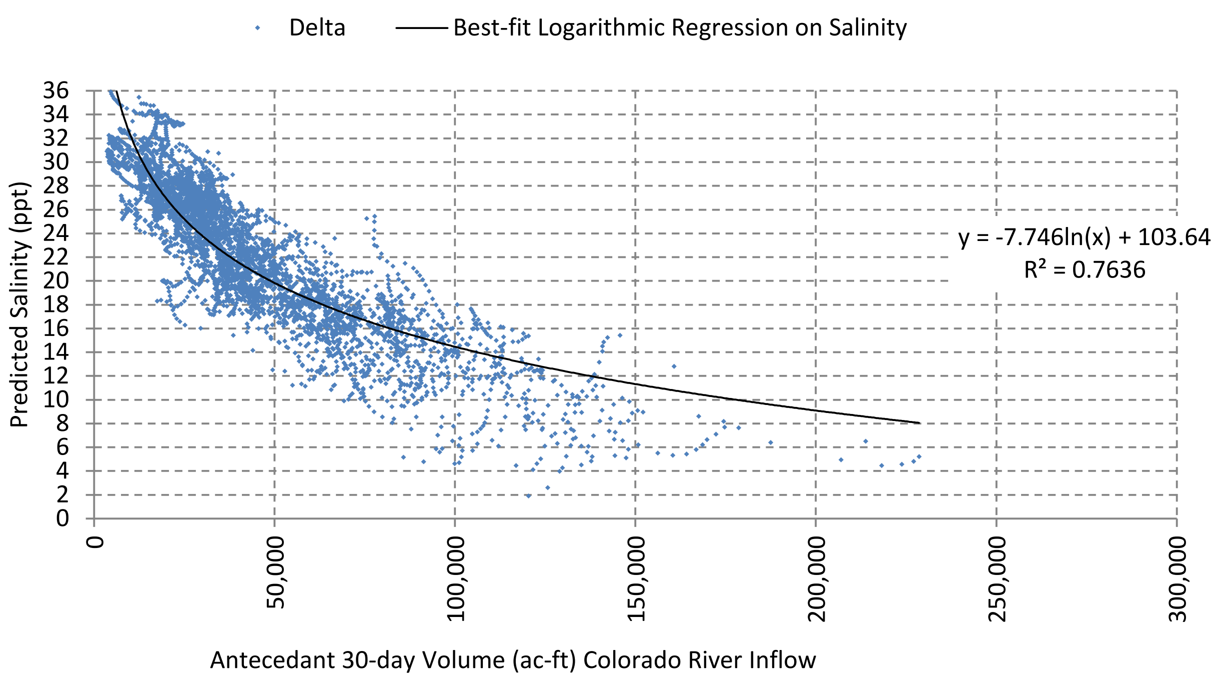
Figure 5-9
Lavaca/Garcitas Inflows Over Simulation Period Classified as Normal (Shaded) and Low Flow Periods (White)
Freshwater Inflow Standards and Ecological Response
TWDB



Entire Period



Low-flow Periods Only

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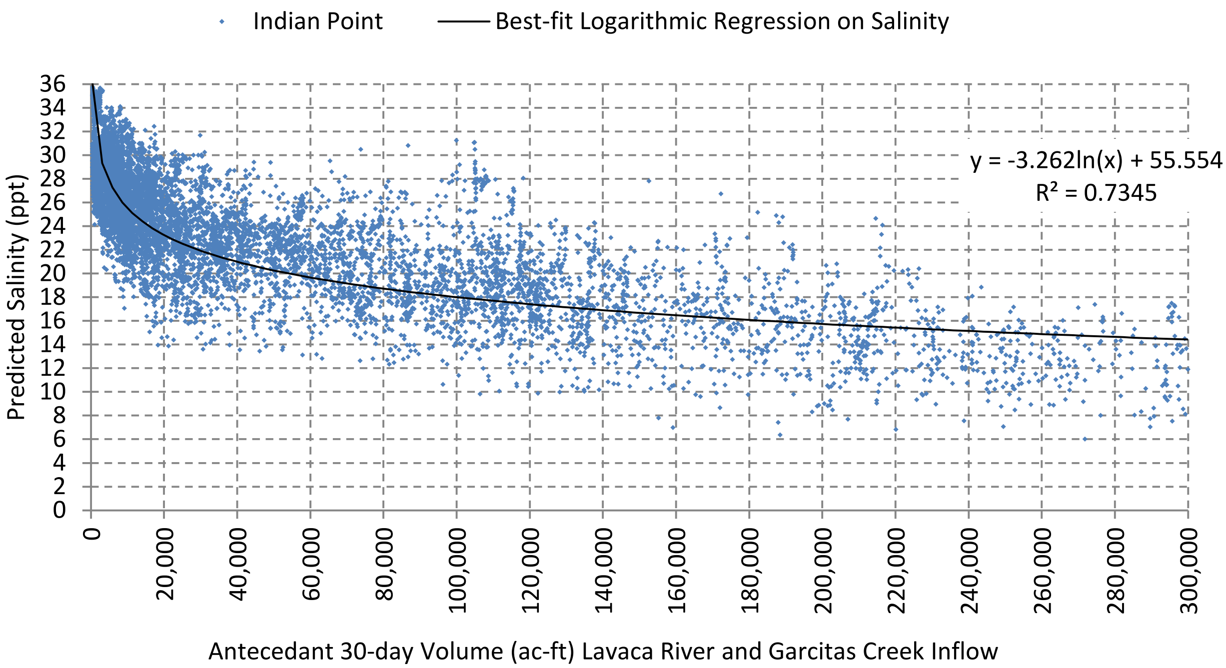
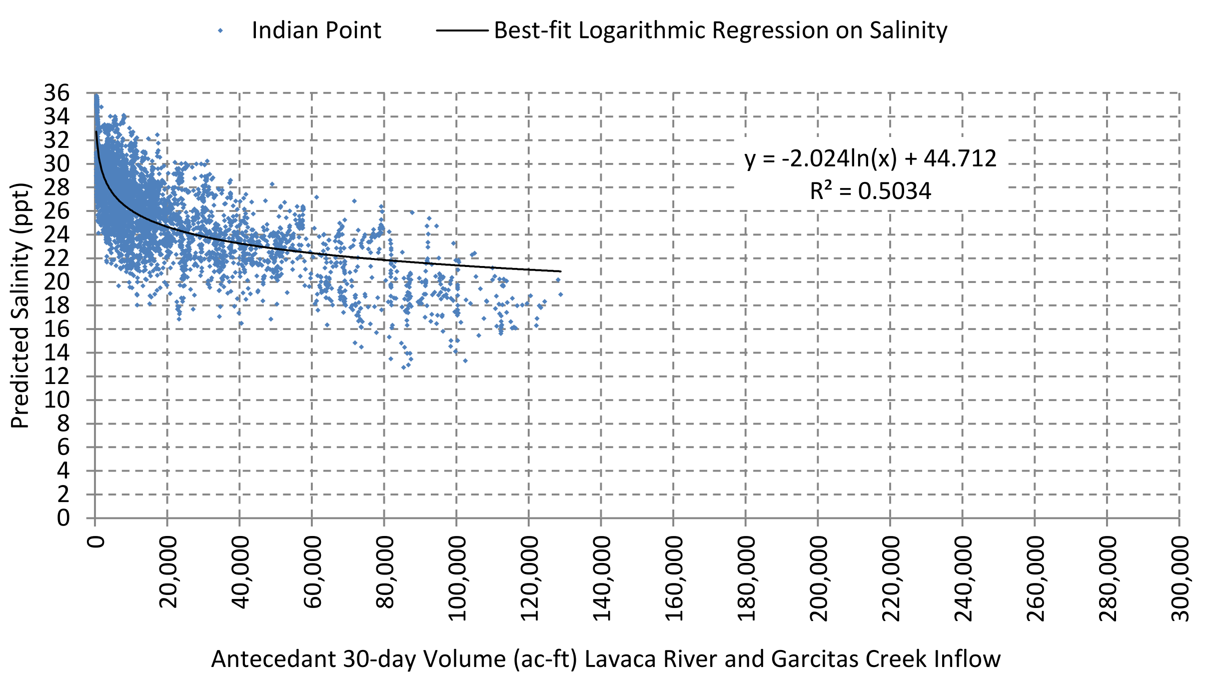
Figure 5-10
Delta Edge Transect, Predicted Salinity vs. Antecedent 30-day Flow (ac-ft)
Freshwater Inflow Standards and Ecological Response
TWDB



Entire Period



Low-flow Periods Only

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Figure 5-11
Indian Point (Middle Ground Reef) Transect,
Predicted Salinity vs. Antecedent 30-day Flow (ac-ft)
Freshwater Inflow Standards and Ecological Response
TWDB

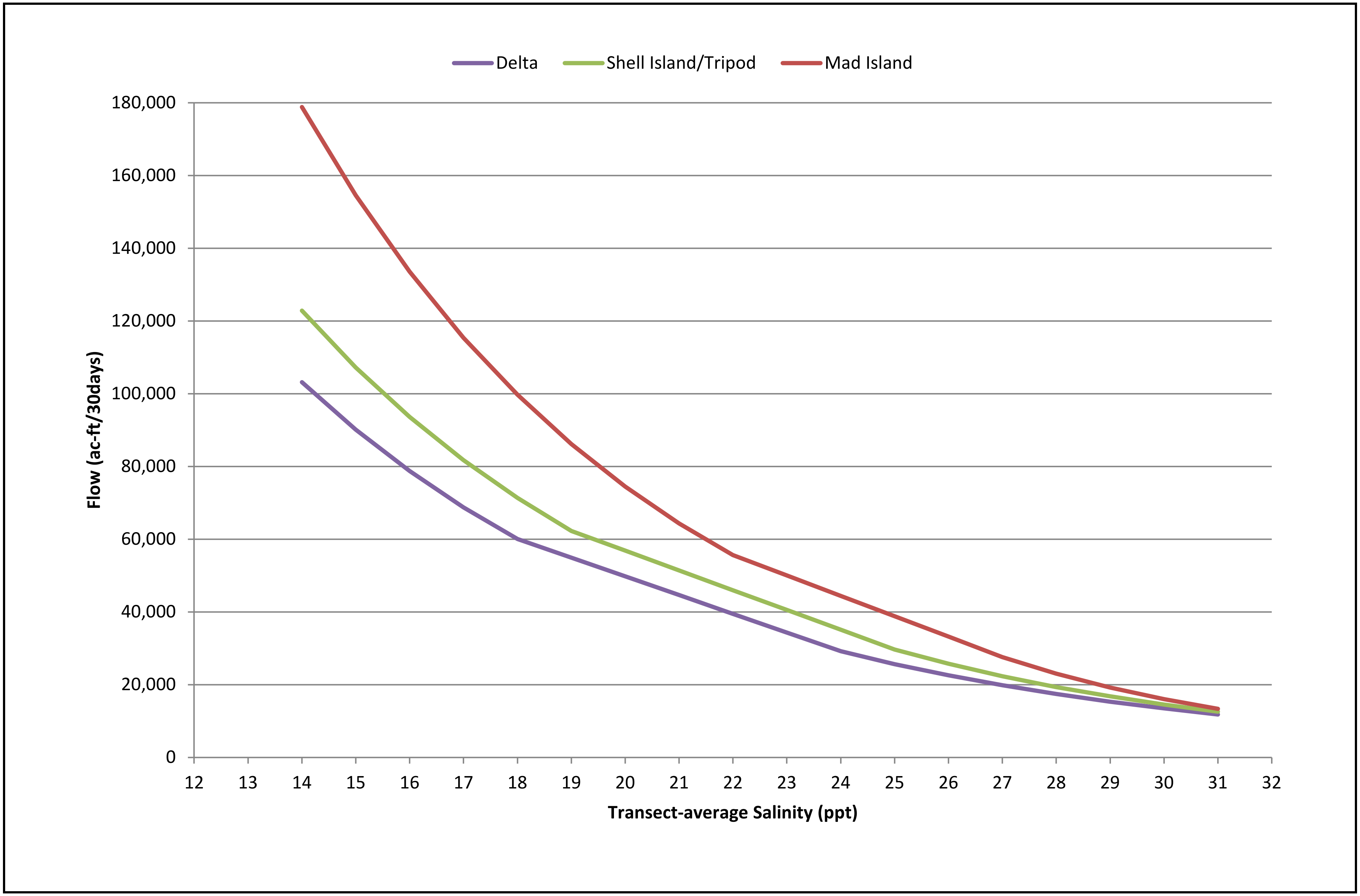

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Figure 5-12
Colorado Blended Regression Curves
Freshwater Inflow Standards and Ecological Response
TWDB

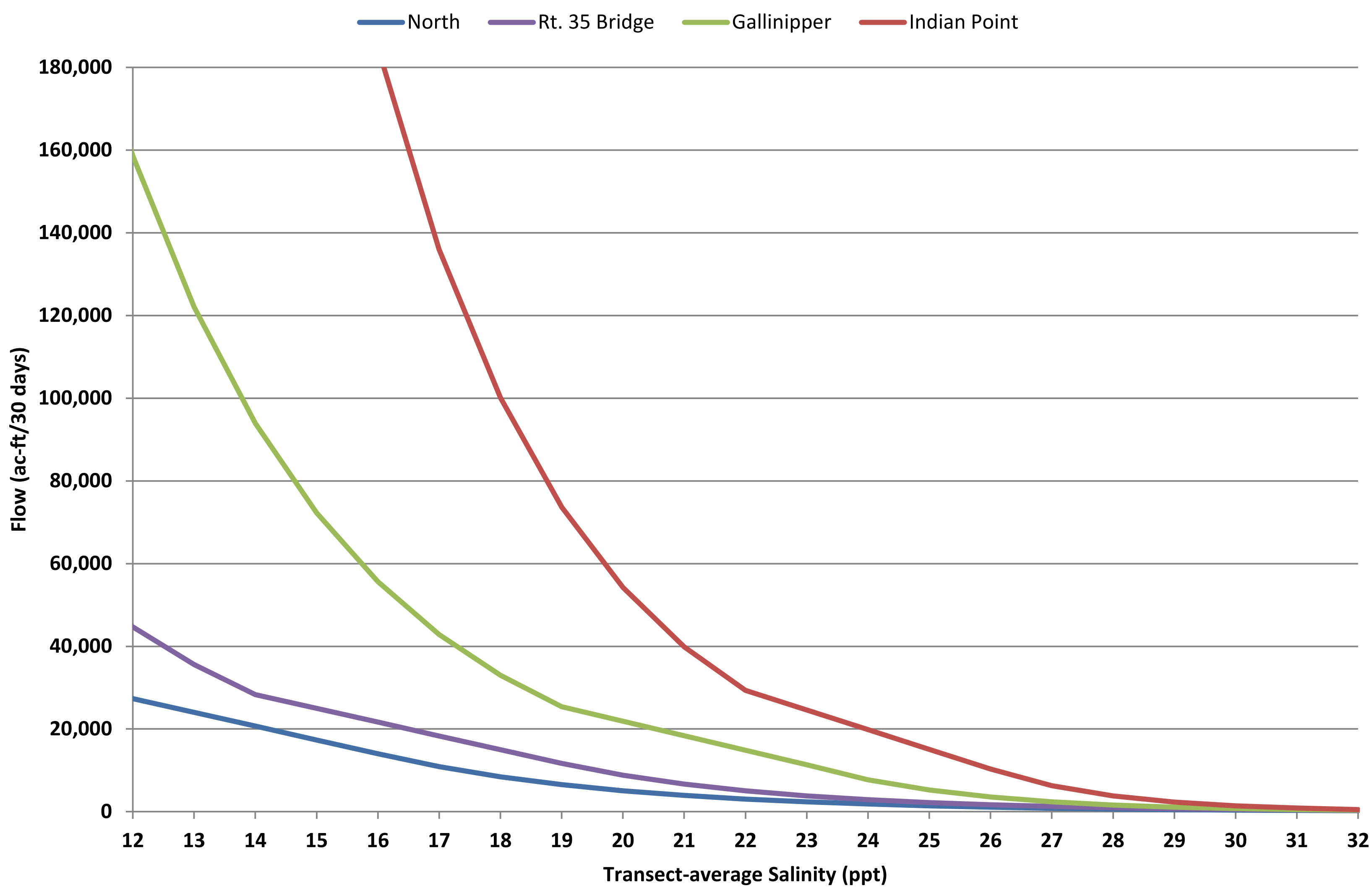

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Figure 5-13
Lavaca Blended Regression Curves
Freshwater Inflow Standards and Ecological Response
TWDB

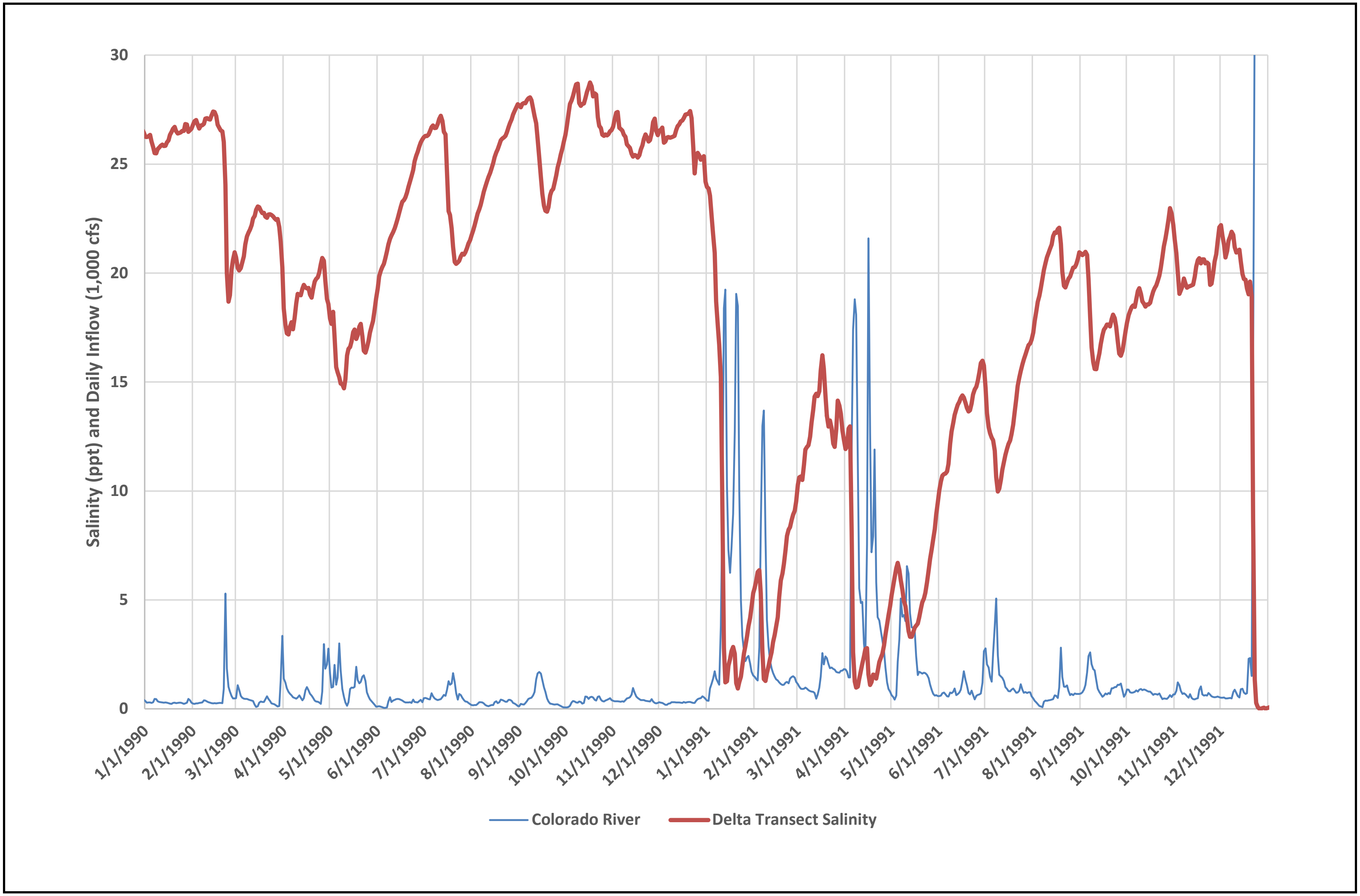

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Figure 5-14
Colorado River Inflow and Modeled Salinity at Delta Transect
Freshwater Inflow Standards and Ecological Response
TWDB

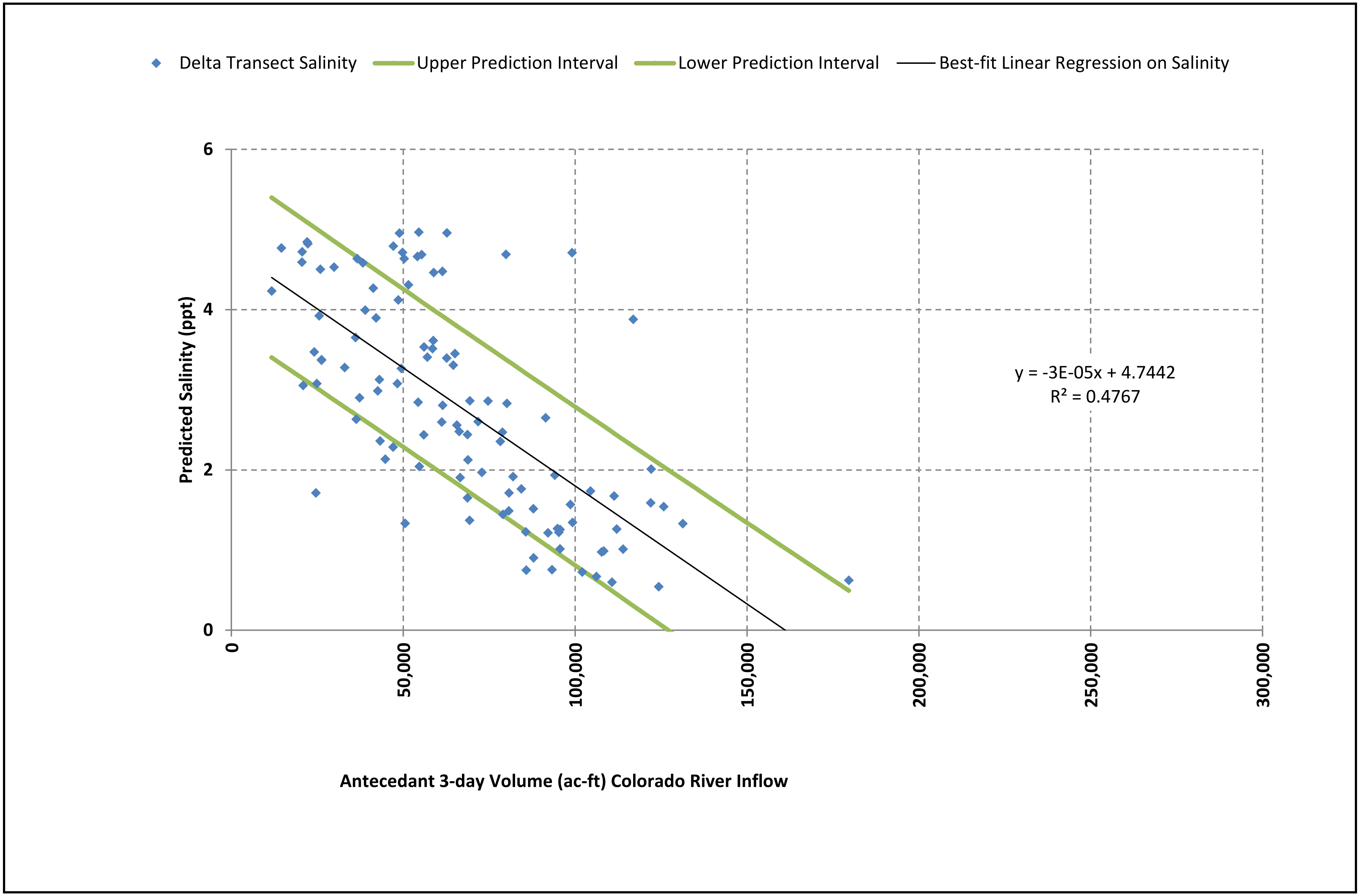

Figure 5-15
Delta Transect, Predicted Salinity vs. Antecedent 3-day Flow (ac-ft)
Filtered to Investigate High Inflow Low Salinity Response
Freshwater Inflow Standards and Ecological Response
TWDB

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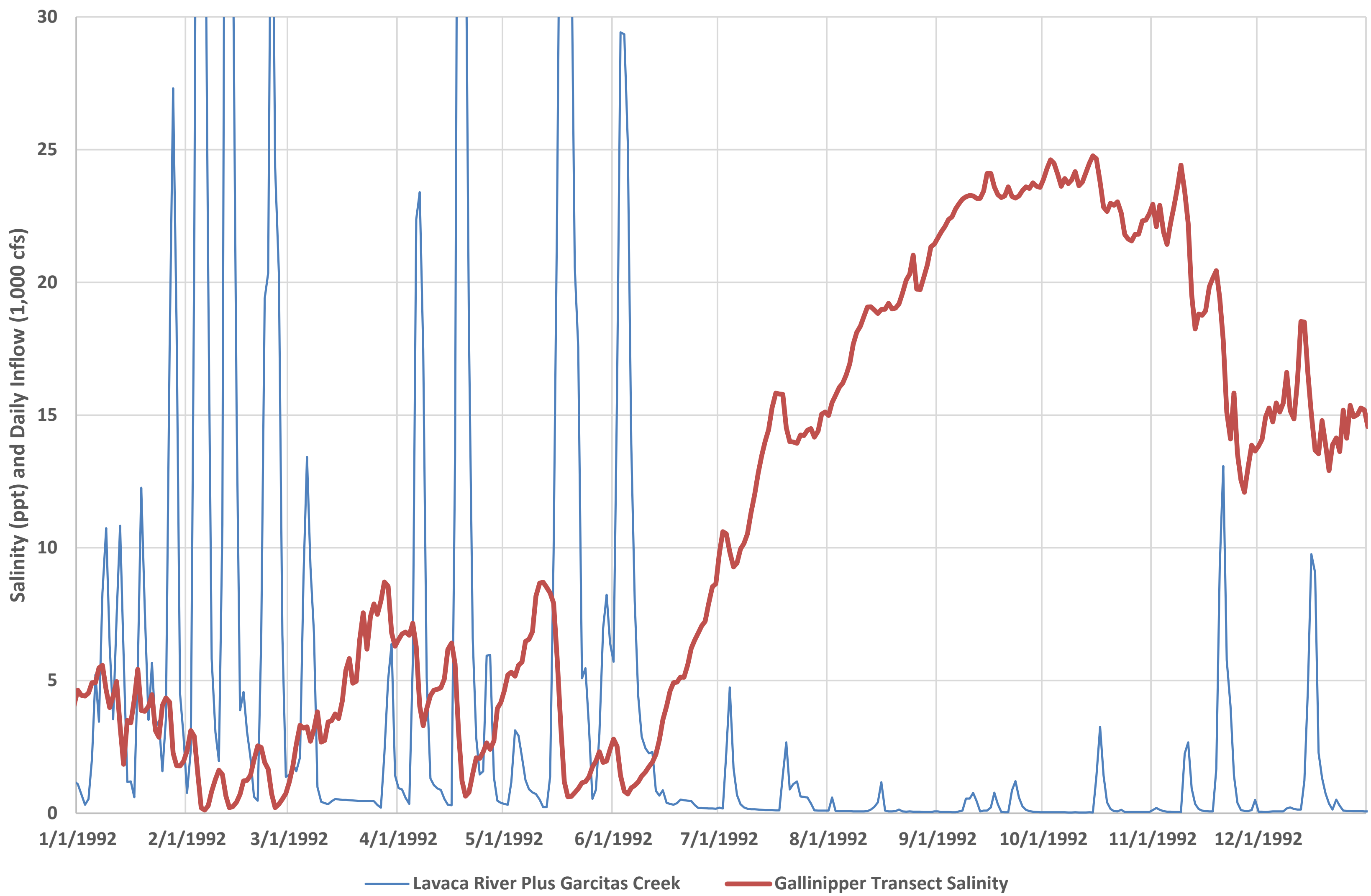

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Figure 5-16
Lavaca River plus Garcitas Creek Inflow and Modeled Salinity at Gallinipper Transect
Freshwater Inflow Standards and Ecological Response
TWDB

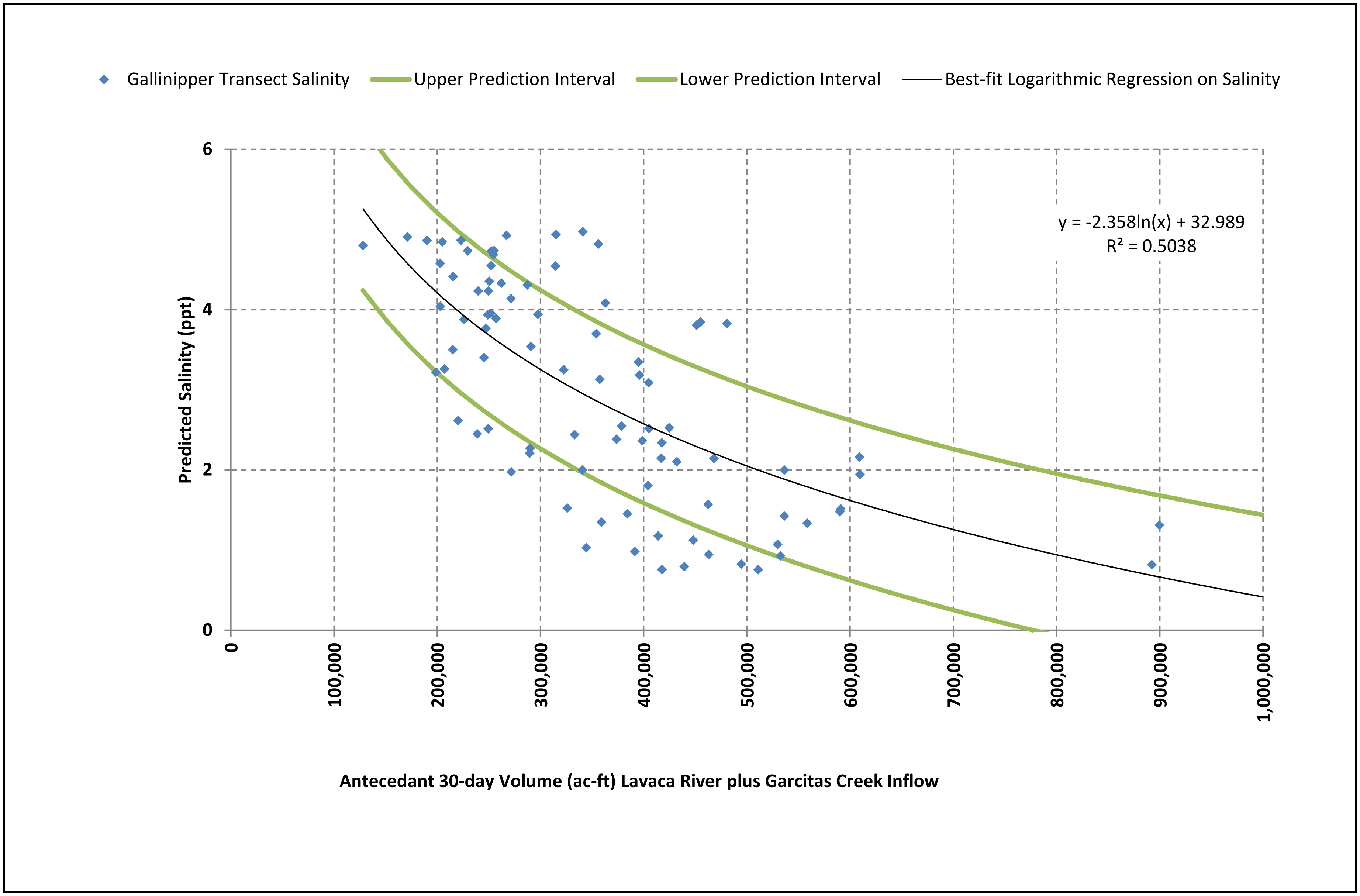

Figure 5-17
Gallinipper Transect, Predicted Salinity vs. Antecedent 30-day Flow (ac-ft)
Filtered to Investigate High Inflow Low Salinity Response
Freshwater Inflow Standards and Ecological Response
TWDB

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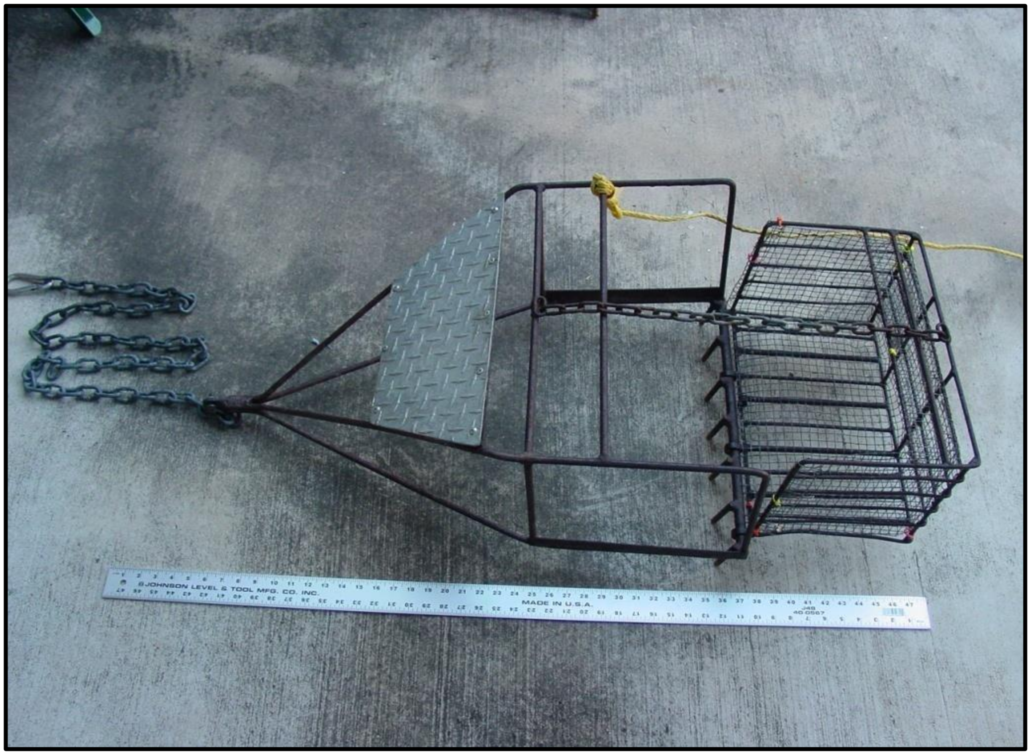
APPENDIX A
PHOTOGRAPHS OF OYSTER SAMPLING

Summer Oyster Samples
Photo Documentation

Sampling Equipment – Oyster Dredge



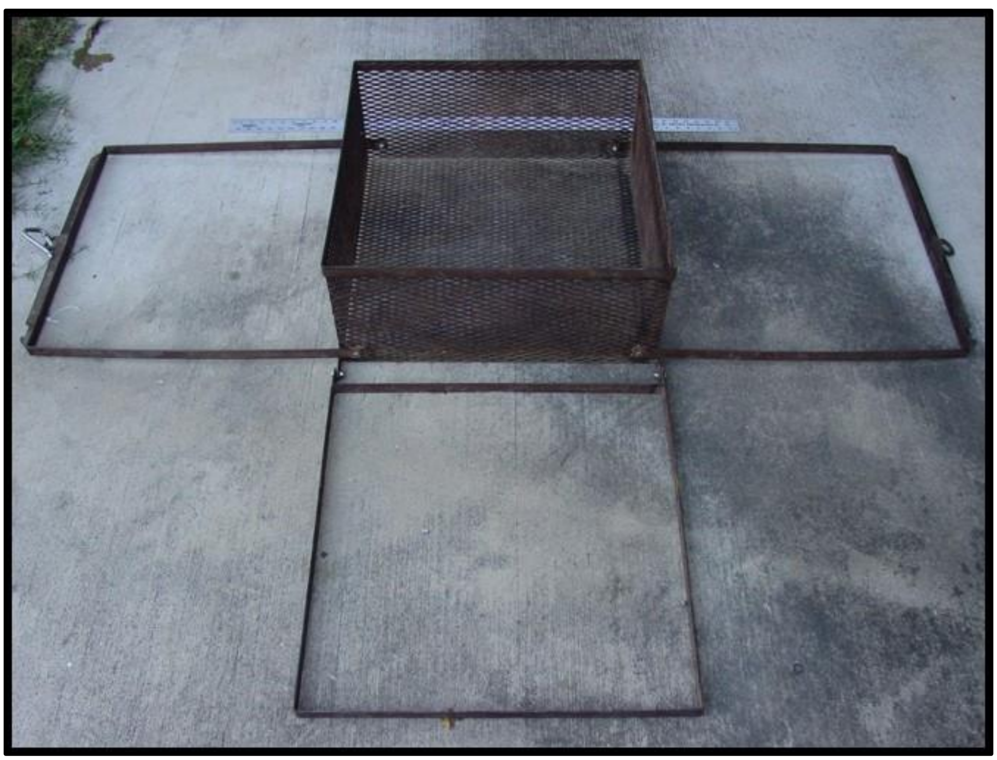
Top view

Side view

Summer Oyster Samples
Photo Documentation

Sampling Equipment – 1/3-meter Quadrat

Closed



Open

Summer Oyster Samples
Photo Documentation

Gallinipper Point

Dredge Tow 1

Dredge Tow 2

Gallinipper Point (cont'd)

Dredge Tow 3

Dredge Tow 4

Summer Oyster Samples
Photo Documentation

Rt. 35

Dredge Tow 1

Dredge Tow 2

Summer Oyster Samples
Photo Documentation

Rt. 35 (cont'd)

Dredge Tow 3

Dredge Tow 4

Summer Oyster Samples
Photo Documentation

Indian Point

Dredge Tow 1

Dredge Tow 2

Indian Point (cont'd)

Dredge Tow 3

Dredge Tow 4

Summer Oyster Samples
Photo Documentation

North Reef

Dredge Tow 1

Dredge Tow 2

Summer Oyster Samples
Photo Documentation

North Reef (cont'd)

Dredge Tow 3

Dredge Tow 4

Summer Oyster Samples
Photo Documentation

Sammy's Reef

Dredge Tow 1

Dredge Tow 2

Summer Oyster Samples
Photo Documentation

Sammy's Reef (cont'd)

Dredge Tow 3

Dredge Tow 4

Summer Oyster Samples
Photo Documentation

Mad Island

Dredge Tow 1

Dredge Tow 2

Summer Oyster Samples
Photo Documentation

Shell Island

Dredge Tow 1

Dredge Tow 2

Summer Oyster Samples
Photo Documentation

CRD 1

Quadrat 1A

Quadrat 1B

Summer Oyster Samples
Photo Documentation

CRD 4

Quadrat 4A

Quadrat 4B

Summer Oyster Samples
Photo Documentation

CRD 5

Quadrat 5A

Quadrat 5B

Summer Oyster Samples
Photo Documentation

CRD 8

Quadrat 8A

Quadrat 8B

Summer Oyster Samples
Photo Documentation

CRD 6

Quadrat 6A

Quadrat 6B

Fall Oyster Samples
Photo Documentation

Mad Island

Dredge Tow 1

Dredge Tow 2

Fall Oyster Samples
Photo Documentation

Mad Island (cont'd)

Dredge Tow 3

Fall Oyster Samples
Photo Documentation

Indian Point

Dredge Tow 1

Fall Oyster Samples
Photo Documentation

CRD 5

Dredge Tow 1

Dredge Tow 2

Fall Oyster Samples
Photo Documentation

Sammy's Reef

Dredge Tow 1

Dredge Tow 2

Sammy’s Reef (cont’d)

Dredge Tow 3

Dredge Tow 4

Fall Oyster Samples
Photo Documentation

CRD 6

Dredge Tow 1

Dredge Tow 2

Fall Oyster Samples
Photo Documentation

CRD 6 (cont'd)

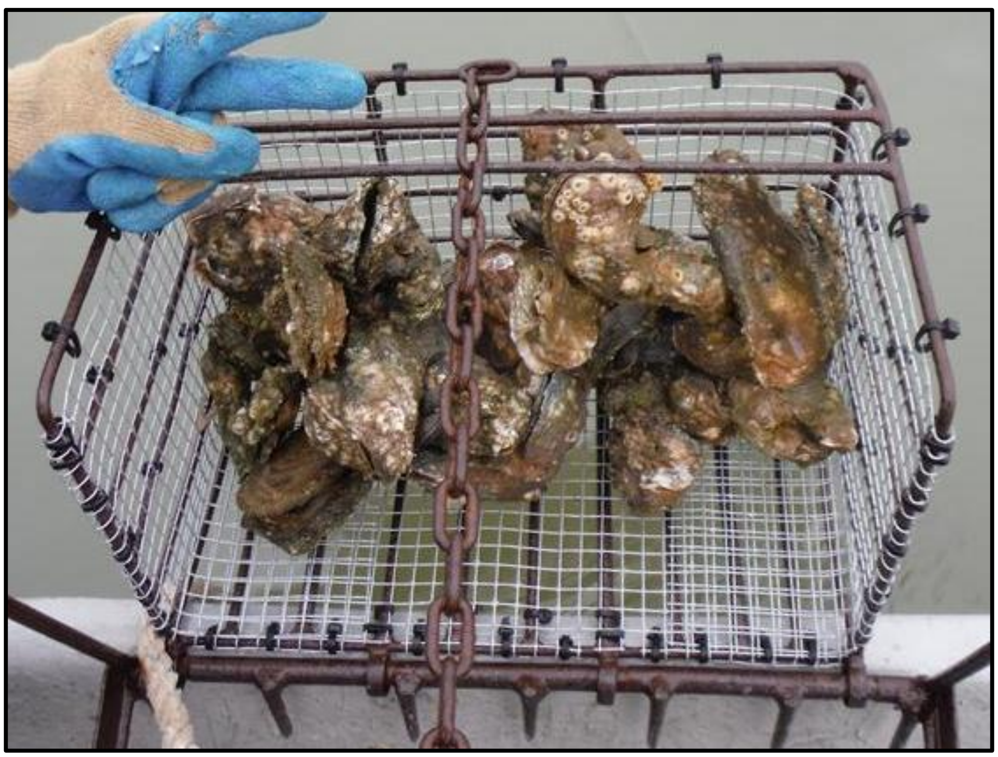
Dredge Tow 3

Dredge Tow 4

Fall Oyster Samples
Photo Documentation

North Reef

Dredge Tow 1



Dredge Tow 2

APPENDIX B
WATER QUALITY RESULTS FROM OYSTER AND THROW TRAP SAMPLING

THROW TRAP WATER QUALITY SAMPLING
FIELD DATA SHEETS
September, 2014

LRD 1	Temperature (C°)	pH	Salinity (ppt)	Conductivity (mS)	D.O. (mg/l)	D.O. Saturation (%)
EA	28.71	7.91	24.29	38.38	5.19	75.7
EB	28.77	7.91	24.39	38.38	5.13	76.0
EC	28.92	7.98	24.51	38.70	5.66	84.6
OA	29.21	8.05	24.8	39.12	5.88	88.3
OB	29.2	8.04	24.78	39.09	5.79	86.6
OC	29.23	8.06	24.87	39.14	5.96	89.2
AVERAGE	**29.01**	**7.99**	**24.61**	**38.80**	**5.60**	**83.40**

LRD 2	Temperature (C°)	pH	Salinity (ppt)	Conductivity (mS)	D.O. (mg/l)	D.O. Saturation (%)
EA	29.08	7.89	22.34	35.73	6.06	89.2
EB	29.25	7.96	22.33	35.56	6.32	93.7
EC	29.35	7.98	22.34	35.61	6.54	96.8
OA	29.4	8.02	22.29	35.52	6.93	103.0
OB	29.34	8.04	22.26	35.56	7.14	105.7
OC	29.52	8.04	22.26	35.49	7.37	109.4
AVERAGE	**29.32**	**7.99**	**22.30**	**35.58**	**6.73**	**99.63**

LRD 3	Temperature (C°)	pH	Salinity (ppt)	Conductivity (mS)	D.O. (mg/l)	D.O. Saturation (%)
EA	29.69	8.00	22.25	35.50	7.44	109.8
EB	29.70	7.99	22.27	35.51	6.86	102.2
EC	29.74	8.00	22.26	35.51	6.70	99.7
OA	29.81	8.01	22.26	35.48	6.79	101.2
OB	29.81	8.02	22.4	35.67	7.09	106.0
OC	29.87	8.02	22.41	35.71	7.13	106.5
AVERAGE	**29.77**	**8.01**	**22.31**	**35.56**	**7.00**	**104.23**

CRD 1	Temperature (C°)	pH	Salinity (ppt)	Conductivity (mS)	D.O. (mg/l)	D.O. Saturation (%)
EA	30.40	7.74	28.40	44.24	5.97	93.3
EB	-	7.6	28.35	44.18	5.93	93.1
EC	31.19	7.65	28.32	44.15	6.39	100.6
OA	31.38	7.66	28.3	44.18	6.36	100.8
OB	31.58	7.66	28.3	44.17	6.54	104.0
OC	31.69	7.68	28.3	44.15	6.90	110.1
AVERAGE	**31.25**	**7.67**	**28.34**	**44.18**	**6.35**	**100.32**

CRD 2	Temperature (C°)	pH	Salinity (ppt)	Conductivity (mS)	D.O. (mg/l)	D.O. Saturation (%)
EA	28.53	7.87	29.26	45.35	5.14	77.9
EB	28.83	7.92	29.83	46.15	6.07	92.5
EC	29.20	7.95	30.04	46.76	6.62	101.4
OA	28.37	7.82	29.15	45.20	5.26	79.8
OB	29.35	7.99	30.32	46.82	7.11	109.5
OC	29.57	8.01	30.43	46.97	7.34	113.9
AVERAGE	**28.98**	**7.93**	**29.84**	**46.21**	**6.26**	**95.83**

CRD 3	Temperature (C°)	pH	Salinity (ppt)	Conductivity (mS)	D.O. (mg/l)	D.O. Saturation (%)
EA	33.16	7.42	27.13	42.58	4.06	66.4
EB	33.01	7.42	27.07	42.58	4.29	69.8
EC	32.92	7.68	27.06	41.41	6.36	99.3
OA	33.03	7.61	27.27	42.76	5.77	94.0
OB	33.05	7.51	27.45	43.02	5.30	86.3
OC	33.02	7.52	27.41	43.01	5.17	83.2
AVERAGE	**33.03**	**7.53**	**27.23**	**42.56**	**5.16**	**83.17**

THROW TRAP WATER QUALITY SAMPLING
FIELD DATA SHEETS
October, 2014

LRD 1	Temperature (C°)	pH	Salinity (ppt)	Conductivity (mS)	D.O. (mg/l)	D.O. Saturation (%)
EA	23.09	-	25.7	39.92	5.96	75.8
EB	23.06	-	25.47	39.88	5.96	79.0
EC	23.06	7.71	25.48	39.89	5.99	79.9
OA	23.29	7.76	25.39	39.76	6.07	81.2
OB	23.33	7.84	25.44	39.84	6.81	91.6
OC	23.35	7.79	25.41	39.8	6.11	82.6
AVERAGE	**23.20**	**7.78**	**25.48**	**39.85**	**6.15**	**81.68**

LRD 2	Temperature (C°)	pH	Salinity (ppt)	Conductivity (mS)	D.O. (mg/l)	D.O. Saturation (%)
EA	24.13	7.92	23.69	37.38	7.68	104.2
EB	24.37	7.89	23.63	37.33	7.6	104.1
EC	24.77	7.97	23.6	37.27	7.81	107.8
OA	24.82	7.98	23.68	37.39	7.72	106.2
OB	24.89	8	23.68	37.38	7.98	110.1
OC	25.01	8	23.71	37.44	7.99	110.1
AVERAGE	**24.67**	**7.96**	**23.67**	**37.37**	**7.80**	**107.08**

LRD 3	Temperature (C°)	pH	Salinity (ppt)	Conductivity (mS)	D.O. (mg/l)	D.O. Saturation (%)
EA	25.02	8.07	21.03	33.59	8.06	110.0
EB	25.61	8.07	21.14	33.75	8.41	115.8
EC	25.95	8.08	21.16	33.82	8.57	118.7
OA	26.14	8.04	21.06	33.66	7.94	110.5
OB	26.2	8.08	21.03	33.62	8.36	116.3
OC	26.46	8.09	21.05	33.64	8.62	120.3
AVERAGE	**25.90**	**8.07**	**21.08**	**33.68**	**8.33**	**115.27**

CRD 1	Temperature (C°)	pH	Salinity (ppt)	Conductivity (mS)	D.O. (mg/l)	D.O. Saturation (%)
EA	26.70	7.85	27.07	42.31	5.78	80.6
EB	26.62	7.83	26.99	42.07	5.66	82.2
EC	27.09	7.82	27.01	42.21	5.63	79.9
OA	27.9	7.8	27.12	42.36	5.94	85.8
OB	28.64	7.82	27.0	42.16	5.03	74.4
OC	28.83	7.76	26.93	42.13	5.38	80.2
AVERAGE	**27.63**	**7.81**	**27.02**	**42.21**	**5.57**	**80.52**

CRD 2	Temperature (C°)	pH	Salinity (ppt)	Conductivity (mS)	D.O. (mg/l)	D.O. Saturation (%)
EA	23.33	7.98	25.06	39.29	5.83	77.5
EB	23.71	8.02	25.18	39.47	5.69	77.6
EC	24.00	7.95	25.37	39.75	5.52	73.0
OA	24.08	7.99	25.92	40.52	5.56	76.7
OB	24.31	8.01	26.10	40.80	5.83	79.6
OC	24.4	8.03	26.69	41.06	6.61	88.3
AVERAGE	**23.97**	**8.00**	**25.72**	**40.15**	**5.84**	**78.78**

CRD 3	Temperature (C°)	pH	Salinity (ppt)	Conductivity (mS)	D.O. (mg/l)	D.O. Saturation (%)
EA	29.61	8.02	22.24	39.74	7.34	109.3
EB	29.13	8.12	24.2	37.31	7.65	113.7
EC	29.96	8.16	25.62	40.28	9.00	138.9
OA	29.89	8.13	25.43	40.04	8.89	135.7
OB	29.94	8.13	25.30	39.85	8.66	131.9
OC	29.51	8.12	24.23	38.31	8.17	122.8
AVERAGE	**29.67**	**8.11**	**24.50**	**39.26**	**8.29**	**125.38**

OYSTER WATER QUALITY SAMPLING
FIELD DATA SHEETS
Late Summer, 2014

Lavaca River Delta

Site	Date	Sampling Method	Sampling Depth (ft)	Temperature (C°)	pH	Salinity (ppt)	Conductivity (mS)	D.O. (mg/l)	D.O. Saturation (%)	Turbidity (NTU)
Gallipiner Point	8/25/2014	Dredge	6.50	30.47	7.90	30.24	46.78	4.26	65.60	20.7
Route 35 Bridge	8/25/2014	Dredge	7.00	30.62	7.93	26.53	41.61	3.79	57.90	22.5
Indian Point	8/25/2014	Dredge	7.00	30.40	7.94	30.95	47.75	4.41	64.38	10.9
Upper Lavaca/ North Reef	8/25/2014	Dredge	5.50	30.18	8.11	20.98	33.67	4.88	72.00	10.7
AVERAGE			**6.50**	**30.42**	**7.97**	**27.18**	**42.45**	**4.34**	**64.97**	**16.20**

Colorado River Delta

Site	Date	Sampling Method	Sampling Depth (ft)	Temperature (C°)	pH	Salinity (ppt)	Conductivity (mS)	D.O. (mg/l)	D.O. Saturation (%)	Turbidity (NTU)
Sammy's Reef	8/25/2014	Dredge	6.00	30.96	8.12	31.51	48.52	4.56	71.10	32.10
Mad Island Reef	8/25/2014	Dredge	5.00	32.01	8.13	31.97	49.52	3.43	56.70	22.50
Shell Island	8/25/2014	Dredge	2.50	33.32	8.15	31.80	49.07	3.89	65.00	39.30
CRD 1	9/1/2014	Quadrat	1.50	27.84	7.71	27.89	43.42	3.59	51.30	44.40
CRD 4	9/1/2014	Quadrat	2.70	28.29	7.65	27.18	42.46	3.67	53.90	200.00
CRD 5	9/1/2014	Quadrat	2.00	28.77	7.38	27.34	41.18	2.92	43.70	14.70
CRD 6	9/1/2014	Quadrat	1.60	30.07	7.94	26.03	43.39	7.69	75.40	58.10
CRD 8	9/1/2014	Quadrat	1.30	29.10	7.78	27.85	40.89	5.03	117.40	89.40
AVERAGE			**2.83**	**30.05**	**7.86**	**28.95**	**44.81**	**4.35**	**66.81**	**62.56**

OYSTER WATER QUALITY SAMPLING
FIELD DATA SHEETS
Late Fall, 2014

Lavaca River Delta

Site	Date	Sampling Method	Sampling Depth (ft)	Temperature (C°)	pH	Salinity (ppt)	Conductivity (mS)	D.O. (mg/l)	D.O. Saturation (%)	Turbidity (NTU)
Indian Point	11/15/2014	Dredge	5.00	11.48	8.10	30.95	46.69	9.09	--	--
Upper Lavaca/ North Reef	11/15/2014	Dredge	4.00	9.47	8.14	25.52	40.17	10.07	--	--
AVERAGE			**4.50**	**10.48**	**8.12**	**28.24**	**43.43**	**9.58**	**--**	**--**

Colorado River Delta

Site	Date	Sampling Method	Sampling Depth (ft)	Temperature (C°)	pH	Salinity (ppt)	Conductivity (mS)	D.O. (mg/l)	D.O. Saturation (%)	Turbidity (NTU)
Sammy's Reef	11/16/2014	Dredge	7.00	11.28	8.15	30.85	47.57	8.79	--	--
Mad Island Reef	11/16/2014	Dredge	4.50	11.00	8.18	30.41	46.95	9.08	--	--
CRD 5	11/16/2015	Dredge	1.50	13.46	8.12	25.32	39.68	8.16	--	--
CRD 6	11/16/2015	Dredge	2.00	13.18	8.20	28.16	43.73	10.77	--	--
AVERAGE			**3.75**	**12.23**	**8.16**	**28.69**	**44.48**	**9.20**	**--**	**--**

APPENDIX C
MARSH VEGETATION RESULTS

MARSH VEGETATION SAMPLING
FIELD DATA SHEETS
October, 2014

Site: Colorado River Delta 1
Date: 10/29/2014
Transect Length: 7.62 meters

Biomass Plot (0.25 m^2)	**EDGE PLOT 1**		**ABG**	**INTERIOR PLOT 1**		**ABG**
Species	**Stem count**	**% Cover**		**Stem count**	**% Cover**	
Batis maritima						
Borrichia frutescens						
Bolboschoenus robustus						
Distichlis spicata						
Iva frutescens						
Juncus roemerianus						
Lycium carolinianum						
Monanthochloe littoralis						
Salicornia virginica						
Schoenoplectus robustus						
Spartina alterniflora	30	70	**296 g/m2**	33	70	**376 g/m2**
Spartina patens						
Spartina spartinae						
Symphyotrichum tenuifolium						
Sueda linearis						

Biomass Plot (0.25 m^2)	**EDGE PLOT Replicate**		**ABG**	**INTERIOR PLOT Replicate**		**ABG**
Species	**Stem count**	**% Cover**		**Stem count**	**% Cover**	
Batis maritima						
Borrichia frutescens						
Bolboschoenus robustus						
Distichlis spicata						
Iva frutescens						
Juncus roemerianus						
Lycium carolinianum						
Monanthochloe littoralis						
Salicornia virginica						
Schoenoplectus robustus						
Spartina alterniflora	37	70	**344 g/m2**	40	70	**440 g/m2**
Spartina patens						
Spartina spartinae						
Symphyotrichum tenuifolium						
Sueda linearis						

MARSH VEGETATION SAMPLING
FIELD DATA SHEETS
October, 2014

Site: Colorado River Delta 2
Date: 10/29/2014
Transect Length: 9.75 meters

Biomass Plot (0.25 m^2)	EDGE PLOT 1		ABG	INTERIOR PLOT 1		ABG
Species	**Stem count**	**% Cover**		**Stem count**	**% Cover**	
Batis maritima						
Borrichia frutescens						
Bolboschoenus robustus						
Distichlis spicata						
Iva frutescens						
Juncus roemerianus						
Lycium carolinianum						
Monanthochloe littoralis						
Salicornia virginica						
Schoenoplectus robustus						
Spartina alterniflora	28	80	**376 g/m2**	24	65	**256 g/m2**
Spartina patens						
Spartina spartinae						
Symphyotrichum tenuifolium						
Sueda linearis						

Biomass Plot (0.25 m^2)	EDGE PLOT Replicate		ABG	INTERIOR PLOT Replicate		ABG
Species	**Stem count**	**% Cover**		**Stem count**	**% Cover**	
Batis maritima						
Borrichia frutescens						
Bolboschoenus robustus						
Distichlis spicata						
Iva frutescens						
Juncus roemerianus						
Lycium carolinianum						
Monanthochloe littoralis						
Salicornia virginica						
Schoenoplectus robustus						
Spartina alterniflora	46	85	**472 g/m2**	25	65	**264 g/m2**
Spartina patens						
Spartina spartinae						
Symphyotrichum tenuifolium						
Sueda linearis						

MARSH VEGETATION SAMPLING
FIELD DATA SHEETS
October, 2014

Site: Colorado River Delta 3
Date: 10/29/2014
Transect Length: 7.62 meters

Biomass Plot (0.25 m^2)	EDGE PLOT 1		ABG	INTERIOR PLOT 1		ABG
Species	**Stem count**	**% Cover**		**Stem count**	**% Cover**	
Batis maritima						
Borrichia frutescens						
Bolboschoenus robustus						
Distichlis spicata						
Iva frutescens						
Juncus roemerianus						
Lycium carolinianum						
Monanthochloe littoralis						
Salicornia virginica						
Schoenoplectus robustus						
Spartina alterniflora	44	80	**384 g/m2**	41	80	**384 g/m2**
Spartina patens						
Spartina spartinae						
Symphyotrichum tenuifolium						
Sueda linearis						

Biomass Plot (0.25 m^2)	EDGE PLOT Replicate		ABG	INTERIOR PLOT Replicate		ABG
Species	**Stem count**	**% Cover**		**Stem count**	**% Cover**	
Batis maritima						
Borrichia frutescens						
Bolboschoenus robustus						
Distichlis spicata						
Iva frutescens						
Juncus roemerianus						
Lycium carolinianum						
Monanthochloe littoralis						
Salicornia virginica						
Schoenoplectus robustus						
Spartina alterniflora	19	70	**225 g/m2**	41	85	**480 g/m2**
Spartina patens						
Spartina spartinae						
Symphyotrichum tenuifolium						
Sueda linearis						

MARSH VEGETATION SAMPLING
FIELD DATA SHEETS
October, 2014

Site: Lavaca River Delta 1
Date: 10/28/2014
Transect Length: 14.32 meters

Biomass Plot (0.25 m^2)	**EDGE PLOT 1**		**ABG**	**INTERIOR PLOT 1**		**ABG**
Species	**Stem count**	**% Cover**		**Stem count**	**% Cover**	
Batis maritima				10	80	**200 g/m2**
Borrichia frutescens						
Bolboschoenus robustus						
Distichlis spicata						
Iva frutescens						
Juncus roemerianus						
Lycium carolinianum						
Monanthochloe littoralis						
Salicornia virginica						
Schoenoplectus robustus						
Spartina alterniflora	34	60	**120 g/m2**			
Spartina patens						
Spartina spartinae						
Symphyotrichum tenuifolium						
Sueda linearis						

Biomass Plot (0.25 m^2)	**EDGE PLOT Replicate**		**ABG**	**INTERIOR PLOT Replicate**		**ABG**
Species	**Stem count**	**% Cover**		**Stem count**	**% Cover**	
Batis maritima				15	80	**200 g/m2**
Borrichia frutescens						
Bolboschoenus robustus						
Distichlis spicata						
Iva frutescens						
Juncus roemerianus						
Lycium carolinianum						
Monanthochloe littoralis						
Salicornia virginica						
Schoenoplectus robustus						
Spartina alterniflora	50	60	**168 g/m2**			
Spartina patens						
Spartina spartinae						
Symphyotrichum tenuifolium						
Sueda linearis						

MARSH VEGETATION SAMPLING
FIELD DATA SHEETS
October, 2014

Site: Lavaca River Delta 2
Date: 10/28/2014
Transect Length: 7.28 meters

Biomass Plot (0.25 m^2)	**EDGE PLOT 1**		**ABG**	**INTERIOR PLOT 1**		**ABG**
Species	**Stem count**	**% Cover**		**Stem count**	**% Cover**	
Batis maritima				24	30	**240 g/m2**
Borrichia frutescens						
Bolboschoenus robustus						
Distichlis spicata						
Iva frutescens						
Juncus roemerianus						
Lycium carolinianum						
Monanthochloe littoralis						
Salicornia virginica						
Schoenoplectus robustus						
Spartina alterniflora	25	55	**288 g/m2**	4	5	**40 g/m2**
Spartina patens						
Spartina spartinae						
Symphyotrichum tenuifolium				22	50	**88 g/m2**
Sueda linearis						

Biomass Plot (0.25 m^2)	**EDGE PLOT Replicate**		**ABG**	**INTERIOR PLOT Replicate**		**ABG**
Species	**Stem count**	**% Cover**		**Stem count**	**% Cover**	
Batis maritima				15	40	**136 g/m2**
Borrichia frutescens						
Bolboschoenus robustus						
Distichlis spicata						
Iva frutescens						
Juncus roemerianus				43	30	**80 g/m2**
Lycium carolinianum						
Monanthochloe littoralis						
Salicornia virginica						
Schoenoplectus robustus						
Spartina alterniflora	24	60	**192 g/m2**	14	15	**64 g/m2**
Spartina patens						
Spartina spartinae						
Symphyotrichum tenuifolium						
Sueda linearis						

MARSH VEGETATION SAMPLING
FIELD DATA SHEETS
October, 2014

Site: Lavaca River Delta 3
Date: 10/28/2014
Transect Length: 8.8 meters

Biomass Plot (0.25 m^2)	**EDGE PLOT 1**		**ABG**	**INTERIOR PLOT 1**		**ABG**
Species	**Stem count**	**% Cover**		**Stem count**	**% Cover**	
Batis maritima						
Borrichia frutescens						
Bolboschoenus robustus						
Distichlis spicata						
Iva frutescens						
Juncus roemerianus				46	60	**216 g/m2**
Lycium carolinianum						
Monanthochloe littoralis						
Salicornia virginica						
Schoenoplectus robustus						
Spartina alterniflora	14	80	**112 g/m2**	5	3	**32 g/m2**
Spartina patens						
Spartina spartinae						
Symphyotrichum tenuifolium						
Sueda linearis						

Biomass Plot (0.25 m^2)	**EDGE PLOT Replicate**		**ABG**	**INTERIOR PLOT Replicate**		**ABG**
Species	**Stem count**	**% Cover**		**Stem count**	**% Cover**	
Batis maritima						
Borrichia frutescens						
Bolboschoenus robustus						
Distichlis spicata						
Iva frutescens						
Juncus roemerianus				17	45	**72 g/m2**
Lycium carolinianum						
Monanthochloe littoralis						
Salicornia virginica						
Schoenoplectus robustus						
Spartina alterniflora	23	35	**192 g/m2**	7	5	**64 g/m2**
Spartina patens						
Spartina spartinae						
Symphyotrichum tenuifolium						
Sueda linearis						

APPENDIX D
THROW TRAP SITE-SPECIFIC SPECIES RESULTS

Table 1 – Species list of organisms sampled with the throw drop trap in the CRD and LRD during each sampling period in 2014. Data presented as total number of individuals (average size of first 30 individuals in mm).

Organism	**Colorado River Delta**				**Lavaca River Delta**			
	September		October		September		October	
	Edge	SNB	Edge	SNB	Edge	SNB	Edge	SNB
Atlantic croaker (*Micropogonias undulatus*)	5 (61.0)	-	-	-	-	-	-	-
Bay anchovy (*Anchoa mitchilli*)	26 (27.12)	87 (25.98)	-	35 (22.86)	394 (28.17)	329 (30.45)	5 (23.0)	3 (30.67)
Bayou killifish (*Fundulus pulvereus*)	-	-	11 (29.55)	-	-	-	-	3 (28.0)
Blackcheek tonguefish (*Symphurus plagiusa*)	5 (27.2)	7 (25.71)	6 (23.0)	9 (24.44)	2 (32.0)	1 (18.0)	14 (26.07)	16 (29.38)
Blue crab (*Callinectes sapidus*)	69 (10.55)	8 (9.25)	94 (11.69)	40 (11.95)	16 (12.06)	4 (16.75)	36 (15.72)	3 (10.67)
Brown shrimp (*Farfantepenaeus aztecus*)	1 (58.0)	-	10 (25.7)	35 (18.6)	18 (27.9)	-	3 (44.67)	2 (30.0)
Clown goby (*Microgobius gulosus*)	-	-	-	1 (30.0)	-	-	-	-
Darter goby (*Gobionellus boleosoma*)	76 (22.03)	50 (19.26)	144 (28.15)	132 (21.20)	35 (16.26)	9 (11.89)	22 (26.86)	6 (41.0)
Flagfin mojarra (*Eucinostomus melanopterus*)	-	2 (21.0)	-	-	-	-	-	-
Grass shrimp (*Palaemontes vulgaris*)	398 (26.90)	4 (21.25)	909 (24.71)	237 (20.16)	142 (30.13)	1 (21.0)	878 (28.83)	30 (25.63)
Green goby (*Microgobius thalassinus*)	-	7 (18.86)	5 (28.6)	2 (36.5)	-	-	-	-
Gulf killifish (*Fundulus grandis*)	-	-	-	9 (26.44)	-	-	4 (42.25)	-
Gulf menhaden (*Brevoortia patronus*)	-	-	-	-	45 (77.0)	-	-	-
Gulf pipefish (*Syngnathus scovelli*)	1 (70.0)	-	-	-	-	-	-	-
Gulf toadfish (*Opsanus beta*)	-	-	-	-	-	-	1 (97.0)	-
Hardhead catfish (*Arius felis*)	-	-	-	-	1 (71.0)	5 (82.8)	-	1 (94.0)
Hogchoaker (*Trinectes maculatus*)	-	-	-	-	-	-	-	1 (33.0)
Inland silverside (*Menidia beryllina*)	-	-	-	-	4 (50.25)	-	-	12 (52.67)
Inshore lizardfish (*Synodus foetens*)	-	-	-	-	-	1 (41.0)	-	-
Lined sole (*Achirus lineatus*)	4 (26.75)	5 (23.8)	-	2 (15.0)	-	-	-	-
Naked goby (*Gobiosoma bosc*)	-	-	12 (25.75)	1 (27.0)	-	-	12 (33.58)	6 (33.83)
Oyster toadfish (*Opsanus tau*)	-	-	-	-	-	-	-	1 (95.0)
Pink shrimp (*Farfantepenaeus duorarum*)	-	-	1 (32.0)	-	-	-	3 (21.0)	-
Sharptail goby (*Oligolepis acutipennis*)	-	4 (28.5)	-	1 (37.0)	-	-	-	-
Sheepshead minnow (*Cyprinodon variegatus*)	-	-	1 (35.0)	-	3 (89.0)	-	-	-
Skilletfish (*Gobiesox strumosus*)	-	-	-	-	-	-	1 (40.0)	-
Snapping shrimp (*Alpheus heterochaelis*)	1 (37.0)	3 (38.67)	30 (22.43)	21 (19.43)	-	-	3 (42.33)	-
Speckled worm-eel (*Myrophis punctatus*)	1 (70.0)	1 (70.0)	-	-	1 (114.0)	-	1 (108.0)	-
Speckled seatrout (*Cynoscion nebulosus*)	14 (35.71)	-	-	1 (86.0)	21 (18.24)	1 (16.0)	2 (88.5)	-
Stone crab (*Menippe mercenaria*)	-	-	2 (9.0)	1 (7.0)	-	-	11 (9.36)	11 (6.91)
Striped mullet (*Mugil cephalus*)	1 (105.0)	-	-	-	-	-	-	-
Violet goby (*Gobioides broussonetii*)	-	1 (164.0)	1 (83.0)	-	-	-	-	-
White shrimp (*Litopenaeus setiferus*)	168 (58.62)	33 (49.82)	68 (21.57)	97 (26.84)	52 (45.77)	21 (26.43)	44 (56.11)	19 (54.74)

APPENDIX E
THROW TRAP VEGETATION COVER RESULTS

THROW TRAP VEGETATION COVER
FIELD DATA SHEETS
September, 2014

LRD 1	Throw Trap Depth (cm)	Species	Percent Cover
EA	48	*Spartina alterniflora*	50
EB	54	*Spartina alterniflora*	25
EC	48	*Spartina alterniflora*	20
AVERAGE	**50.00**		**31.67**

LRD 2	Throw Trap Depth (cm)	Species	Percent Cover
EA	49	*Spartina alterniflora*	45
EB	61	*Spartina alterniflora*	10
EC	60	*Spartina alterniflora*	15
AVERAGE	**56.67**		**23.33**

LRD 3	Throw Trap Depth (cm)	Species	Percent Cover
EA	70	*Spartina alterniflora*	10
EB	62	*Spartina alterniflora*	30
EC	58	*Spartina alterniflora*	60
AVERAGE	**63.33**		**33.33**

CRD 1	Throw Trap Depth (cm)	Species	Percent Cover
EA	39	*Spartina alterniflora*	15
EB	38	*Spartina alterniflora*	5
EC	39	*Spartina alterniflora*	3
AVERAGE	**38.67**		**7.67**

CRD 2	Throw Trap Depth (cm)	Species	Percent Cover
EA	18	*Spartina alterniflora*	10
EB	39	*Spartina alterniflora*	40
EC	43	*Spartina alterniflora*	5
AVERAGE	**33.33**		**18.33**

CRD 3	Throw Trap Depth (cm)	Species	Percent Cover
EA	33	*Spartina alterniflora*	20
EB	32	*Spartina alterniflora*	5
EC	32	*Spartina alterniflora*	30
AVERAGE	**32.33**		**18.33**

THROW TRAP VEGETATION COVER
FIELD DATA SHEETS
October, 2014

LRD 1	Throw Trap Depth (cm)	Species	Percent Cover
EA	14	*Spartina alterniflora*	5
EB	12	*Spartina alterniflora*	10
EC	26	*Spartina alterniflora*	30
AVERAGE	**17.33**		**15.00**

LRD 2	Throw Trap Depth (cm)	Species	Percent Cover
EA	22	*Spartina alterniflora*	25
EB	10	*Spartina alterniflora*	30
EC	15	*Spartina alterniflora*	3
AVERAGE	**15.67**		**19.33**

LRD 3	Throw Trap Depth (cm)	Species	Percent Cover
EA	20	*Spartina alterniflora*	30
EB	24	*Spartina alterniflora*	25
EC	23	*Spartina alterniflora*	8
AVERAGE	**22.33**		**21.00**

CRD 1	Throw Trap Depth (cm)	Species	Percent Cover
EA	18	*Spartina alterniflora*	5
EB	13	*Spartina alterniflora*	10
EC	11	*Spartina alterniflora*	2
AVERAGE	**14.00**		**5.67**

CRD 2	Throw Trap Depth (cm)	Species	Percent Cover
EA	14	*Spartina alterniflora*	5
EB	24	*Spartina alterniflora*	3
EC	17	*Spartina alterniflora*	2
AVERAGE	**18.33**		**3.33**

CRD 3	Throw Trap Depth (cm)	Species	Percent Cover
EA	14	*Spartina alterniflora*	5
EB	13	*Spartina alterniflora*	3
EC	12	*Spartina alterniflora*	8
AVERAGE	**13.00**		**5.33**

APPENDIX F
DATA AND TXBLEND ELECTRONIC FILES

Provided on enclosed CD

APPENDIX G
EXECUTIVE ADMINISTRATOR'S DRAFT REPORT COMMENTS

ATTACHMENT

Studies to Evaluate Achievement of Freshwater Inflow Standards and Ecological Response

Anchor QEA

Contract # 1400011715

TWDB comments to Final Draft Report

The report is well-written, comprehensive and generally reflects the scope of work. As noted in the report, the results reflect only a snapshot in time for bay conditions. It will be important for the reader to keep in mind that conclusions of the report are based on a limited extension of the data record and that additional data, information, and modeling specific to Matagorda and Lavaca bays over a full range of inflow regimes is needed to corroborate or suggest adjustments to the full suite of environmental flow standards for this estuarine system.

REQUIRED CHANGES

General Draft Final Report Comments

The report appears to corroborate the environmental flow standards recommended by the BBEST, BBASC, and adopted by the TCEQ with the exception of indicating a need for additional high-flow standards to ensure freshets that are necessary to ensure the health of Lavaca Bay. The shift from using the RMA hydrodynamic model to the TxBLEND hydrodynamic model appears to be justifiable and desirable for pursuing future efforts along these lines..

TCEQ's rules provide that an application cannot cause or contribute to impairment of the inflow regimes in the rule and that impairment is determined during a water availability analysis for a new appropriation of water (§298.330(a)). The identification of the most downstream points in basins in §298.305 is related to the determination of impairment, as calculated in the WAMs, and is not intended to guide efforts to develop the actual inflow regimes. For example, the most downstream point in the Colorado and Lavaca Basins and Garcitas Creek would be the most downstream point in the respective WAMs, and the flows to be analyzed in TCEQ's determination on an application for a new appropriation of water would be the WAM flows at those points, without any adjustments.

Please spell-check and proofread for grammar and logic all sections, including tables, figures and appendices. Also please ensure all page breaks do not split section titles from their associated paragraphs.

Specific Draft Final Report comments

Cover Page – Please add the following language to the cover page of the final report:

> *PURSUANT TO SENATE BILL 1 AS APPROVED BY THE 83RD TEXAS LEGISLATURE, THIS STUDY REPORT WAS FUNDED FOR THE PURPOSE OF STUDYING ENVIRONMENTAL FLOW NEEDS FOR TEXAS RIVERS AND ESTUARIES AS PART OF THE ADAPTIVE MANAGEMENT PHASE OF THE SENATE BILL 3 PROCESS FOR ENVIRONMENTAL FLOWS ESTABLISHED BY THE 80TH TEXAS LEGISLATURE. THE VIEWS AND CONCLUSIONS EXPRESSED HEREIN ARE THOSE OF THE AUTHOR(S) AND DO NOT NECESSARILY REFLECT THE VIEWS OF THE TEXAS WATER DEVELOPMENT BOARD.*

Executive Summary, page E-4 – Please clarify that under the current circumstances in which *Rangia* were not present in the system, their use as an indicator species was not recommended (or feasible) for this study, rather than concluding that they may not be a good indicator species for future work. Their biology suggests that they could serve as an indicator species when present in a bay system. Based on other information, it appears that low *Rangia* abundance has been reported in other Texas bay systems where they previously were present in high numbers, suggesting a possibly more complex response than the conclusion acknowledges.

Section 2.2.2 Oyster and Dermo Condition Index Development, page 14 (1st ¶) – Please include the units for the density of oysters metric.

Section 2.2.2 Oyster and Dermo Condition Index Development, page 14 (last ¶) – Please correct the "Soniat 2005" citation to read "Soniat *et al.* 2005"

Section 2.2.2 Oyster and Dermo Condition Index Development, page 15 (last sentence) – Please rephrase the sentence to read, "*...data are only available (depending on the reef) from 1998 to 2011 in Galveston Bay and from 2003/2004 to 2011 in Matagorda, Lavaca, and San Antonio bays.*"

Section 2.2.3 Previous Suitability Criterion Models, page 17 – Please clarify whether this section is a summary of previous work solely or an update to the previous models using newly collected data. One reviewer commented that both DCI and OCI are insufficient to explain oyster count variability and that the use of DCI is suspect as it is not a sufficiently robust model. The reviewer requests inclusion of a full methodology in the document which includes number of replicate samples used to determine oyster viability is needed. If this request is applicable, please include the information in the appropriate section.

Section 2.3.1 Methods and Materials, page 18-19 – Please include the number of replicates per site per sampling event that were available for analysis.

Section 2.4 Data Trends, page 25 – Please provide additional information on the RFTM dermo assay methodology for detecting Dermo.

Section 2.4 Data Trends, page 25 – The second bullet states: "*Many reefs had relatively low dermo in winter 2008 and winter 2009 which may be a result of significant inflows and reduced salinity in 2007.*" Please consider that these results also may be due to extremely low temperatures in 2008 and 2009. The referenced dermo evaluation also includes a discussion of the issues with colder temperatures making it difficult to see dermo cells using the RFTM method in samples collected during the winter months. Dermo cells may be reduced in size and not reproductive at temperatures lower than 10° C. False negatives are common when cold water temperatures have been persistent for a while. Use of whole body count of dermo cells or PCR is very important for determining whether a reef population is actually free of dermo.

Section 2.4 Data Trends, page 25 – The third bullet states: "*Some stations with generally low dermo exhibit an increase in dermo during the recent drought. In particular the November 2014 sample in North Reef was the highest ever recorded at that location. This may be a consequence of consistently high salinity over the antecedent four years. However variability is high as evidenced but the zero dermo WP measured at this reef in August 2014.*" Please consider that this observation may be due also to these reefs rarely being infected by dermo and thus being more susceptible to the disease. Oysters adapted to higher salinity conditions and ever present, low levels of dermo tend to be more resistant to the disease. (Bushek *et al.* (2012) suggests such a situation in Delaware Bay where upstream reefs that are never infected during normal flow conditions or floods have higher mortality from dermo if conditions become saltier than normal.)

Section 2.8 Updated Monthly Model Construction, page 31 – Please consider rephrasing the first sentence to read something along the lines of: "*The failure of the old DCI suitability criterion model to predict dermo.... indicated a need for additional data in order to potentially improve the model.*"

Section 2.9.1 Updated Dermo Model, page 34 (1st ¶, 2nd to last sentence) – Please verify whether the word "*salinity*" should be replaced with "*temperature*": "*...likely because the temperature measured in the same month as the dermo will not yet have impacted...*"

Section 3.2 Matagorda Bay Health Evaluation Habitat Model Overview, page 41(2nd full ¶) – Please clarify the sentence to read: "*Also noteworthy is that condition changes within the CRD are driven primarily by Colorado River flows, while changes in East Matagorda Bay are more controlled by localized freshwater inflow.*"

Section 3.3.2.1 Matagorda Bay Health Evaluation Key Species, page 47- 48 (last ¶) – In response to the first question posed about the use of the density of mobile organisms as indicators of estuarine health, one reviewer offered that mobile species are suitable if the seasonal life stage movements of the selected species selected are incorporated into the sampling regime and analysis.

In response to the third question posed about the clumping of marsh vegetation, one reviewer offered that we need to consider the possibility that a modeled threshold inflow may allow hypersaline conditions to develop in the upper reaches and fringing wetlands of the estuaries. Therefore, it may be important to ensure threshold inflows are high enough to prevent hypersalinity in the areas we consider most important for refugia.

One reviewer disagreed with the conclusion that the MBHE level 2 conditions were protective of white shrimp and blue crab, due to the limited amount of data analyzed.

Section 3.3.2.2 Throw Trap Community Analysis, page 51– Please consider reevaluating or rephrasing the conclusions in the last two sentences in this section based on statements made in Tolan and Nelson (2013) which support a statistically significant relationships between salinity and estuarine communities. Example statements from Tolan and Nelson (2013) include: "*...the same environmental driver that could be used to differentiate tidal streams locations (namely, salinity) was the common driver shaping community structure.*" (page 11); "*...salinity again appears to play at least some role in structuring community compositions.*" (page 13); and, "*At the scale of the community, salinity appears to structure nekton diversity and abundance levels,...*" (page 13). The reported conclusions also contradict *Section 6.2, page 72, last paragraph* that suggests the community analysis indicated direct relationships with salinity.

Section 4.1 Limited Rangia *Surveys, page 52 (last ¶, 1st sentence)* – Please consider revising the sentence, "*The Matagorda delta system is unique...*" to read "*The Colorado River estuary is unique...*". The Colorado River estuary extends upstream past the GIWW; however, the report describes the delta as the area downstream of the GIWW in the eastern arm of Matagorda Bay.

Section 5.1 Hydrodynamic Model, page 56 (end of 1st ¶) – The report includes the following statement: "*The final inflow results that would have been calculated had the TxBLEND model been employed in the earlier study are discussed in Section 5.*" Please specify the location of that discussion in *Section 5*, or if missing please include. Also if Table 5.9 includes relevant information, please reference it here as well.

Section 5.1 Hydrodynamic Model, pages 56 - 57 – Please include a statement to address whether the extended period of record of the TxBLEND modeling is affected by the reconfiguration of the Colorado River. One reviewer is concerned that the reconfiguration may have a significant impact on the hydrodynamic modeling results and the study analysis. Note: TWDB accounts for the reconfiguration of inflow volume. More information can be provided by contacting TWDB staff.

Section 5.1 Hydrodynamic Model, page 57 (last full ¶) – The text refers the reader to *Section 5.7* for a discussion of inconsistencies in the TxBLEND input files. Please correct this to say *Section 5.6*. One reviewer expressed concerns about the potential significance of the inconsistencies in the TxBLEND input files and recommended further discussion on this topic and its relevance to the study findings.

Section 5.2 Design Area, page 58 (last full ¶) – Please clarify whether salinity zonation occurs along each transect (from shore-to-shore) – for both observed measurements and model output – and if so whether the decision to use an average salinity for each transect does not compromise the analysis.

Section 5.5 Comparison with Previous Studies, page 64 (in reference to Table 5 – 10) – Please add a discussion in the text addressing the somewhat anomalous results which describe a decrease in subsistence flow values, compared to previous BBEST calculations while all other flow values increase relative to the same baseline.

Section 6.1 Oysters and Dermo, page 68 – Earlier sections of the report conclude that predicted values of the original MBHE oyster health model versus actual new data were not a good fit (*Figures 2-16, 2-17 and 2-18*) and thus justified reconstructing a new regression model. The fact that the new data fits better with the new model, but the overall resulting inflow recommendations do not differ from the original study results is confusion, as noted in this statement: "*The updated dermo and oyster monthly regression model results are generally consistent with the MBHE results and the existing flow standards, and there does not appear to be an urgent need to modify the existing flow standards* ..." One reviewer, nonetheless encouraged the BBASC to continue collecting data for model refinement in order to improve the inflow standards relative to true drought conditions as have occurred since 2008.

Section 6.1.2 Potential New Long-Term Average Inflow Criteria for Lavaca Bay, page 70-72 – Please consider redefining the term "*dermo WP*" here or where appropriate in the conclusions and recommendations section.

Section 6.2 Marsh Productivity, page 72 (2nd ¶) – Please clarify the statement "*However, considering the freshwater inflow conditions experienced during this study*" to remind the reader of the specific inflow conditions that did occur during the study (i.e., high versus low, *etc.*).

Also, please clarify the meaning of the last sentence, "*...being able to quantify actual predictions in the sense of a rigorous inflow criteria validation.*"

Section 6.2 Marsh Productivity, page 73 – Refer to earlier comment about the use of *Rangia* as an indicator species given for the *Executive Summary* (above). Please consider rephrasing the conclusion that *Rangia* are not good indicator species for these systems to account for the fact that they may not presently be good indicators for these systems or simply that they were not a good indicator for this particular study due to their absence. Indeed, the two sentences preceding this conclusion seem to acknowledge that more information is needed to draw informed conclusions about factors affecting *Rangia*.

Section 6.3 Salinity Modeling, pages 73-74 (carry-over ¶) – Please address the noted uncertainty created by relying on flow values from the Bay City gage during times of low inflows, which is acknowledged on Pages 65 – 66, and the potential effect of not being able to adequately model bay inflow and salinity at gaged flows below 2,000 cfs. Please include a discussion as to whether and to what degree this impacts the ability to determine Threshold and Level 1 inflows from the Colorado River.

Section 6.3 Salinity Modeling, page 74 (2nd ¶, 2nd sentence) – Please rephrase the sentence, "*Prior to this analysis, the hydrodynamic model (RMA) was froze with a limited period of record, and the analyses applied to the two bays differed without good justification for these differences.*" To read, "Prior to this analysis, the RMA hydrodynamic model simulated a limited period of record, and the inflow analyses applied to the two bays differed due to limitations which prevented the re-running of the RMA model during the BBEST analysis for Lavaca Bay." Consider following this with "*The approach to utilize the TxBLEND hydrodynamic and salinity transport model and apply the methodology to Lavaca Bay presented in this report will allow for the continued application of a consistent approach that can continue…*"

One reviewer also suggested adding the language "*... in this report should be adopted and used in future studies since it will allow for the continued application of a consistent approach that can continue ...*", if this is a recommendation to the BBASC from this study report.

Figures and Tables Comments

Table 2-1, Reef 19 – Please verify whether the number of stations is four (as indicated in the column *Number of Stations*) or five (as indicated in the column *Stations in Reef*).

Table 2-2 – Please correct the column heading "*Size Range (mm)*", removing the "2".

Table 2-2, CRD 5, 9/1/2014 – Please verify the salinity value reported here for CRD 5 (27.3 ppt) with the value reported in Appendix B (7.3 ppt). Please clarify that CRD-labeled sites are equal to OR-labeled sites or standardize the term.

Table 2-3 – Please correct the column heading "*Size Range (mm)*", removing the "2".

Table 2-3 – Please verify the salinity values reported here with the values reported in Appendix B.

Table 3-6 – Please consider modifying Table 3-6 (or adding a new table) to show data supporting the statement that numbers of individuals and species were higher in October. Please also correct the name of the *speckled worm-eel* in the table.

Figure 3-8 – Please consider defining "*maximum potential productivity*" in the figure note.

Bibliography Comments

Please add the following to the Bibliography –

Hofmann *et al.* 1995 (referenced in *Section 2.1, page 8*)

Hofmann *et al.* 1992 (referenced in *Section 2.1, page 9*)

NMFS database citation (referenced in *Section 3.3.2.1, page 47, 2nd paragraph*)

Ragone, Calvo and Burreson. 1994 (referenced in *Section 2.1, page 9*; or change the citation to match the text in this section)

Appendix Comments

Appendix B – Please clarify whether the "*OR*" designations are the same as the "*CRD*" designations used in the text and tables. If so, please standardize all to the CRD term.

Appendix C – If available, please include data for Lavaca Bay.

Appendix D, Table 1 – Please correct the table caption to change "NCRD" to read "CRD".
Please verify the common or scientific names for the following:
pink shrimp – *Farfantepenaeus duorarum*
speckled worm-eel – *Myrophis punctatus*
Please verify the spelling of *Micropogonias undulatus* and *Farfantepenaeus aztecus.*

<u>SUGGESTED CHANGES</u>

Specific Draft Final Report Comments

Section 1.4 Overview of Historical Hydrology and Salinity, page 4-5 – Please consider clarifying the text to reflect that the points identified in the rules are intended to apply to the determination of impairment and not to the development of inflow regimes.

Section 2.3.2.1 Dermo Analysis Results, page 21 (2nd ¶) – Please provide supporting data for the statement: "Of the reefs common between this study's field collection and the Oyster Sentinel program, *Sammy's Reef, Gallinipper Point, Indian Point, and Mad Island Reef had <u>lower</u> disease levels, whereas North Reef and Shell Island had <u>similar</u> levels when compared to recent past years.*"

Section 2.4 Data Trends, page 22(last ¶, last sentence) – One reviewer provided information that the region of Lavaca Bay where North Reef occurs has experienced hypersaline conditions during dry/low inflow periods. Please comment on whether such hypersaline conditions are considered in the analysis and/or how hypersalinity impacts the relationship between dermo, oysters and salinity modeling.

Section 2.6.1 Average Dermo and Salinity, page 28 – One reviewer suggests that the use of Confederate Reef in the dermo/oyster analysis is inappropriate as the oyster count, water temperature, and salinity data is collected from submerged locations of the reef; whereas, dermo data were collected from the shallow intertidal areas of the reef. Due to differences in the water depth and susceptibility and exposure of oysters to dermo in these two locations of the same reef complex, it is not appropriate to compare the data. The reviewer recommends that the analysis would have been better if this reef had been excluded, even though it was used in the MBHE.

The reviewer offers this information: Dr. Sammy Ray never collected oyster samples to run dermo analysis from Confederate Reef, because most of the oysters were dead. TPWD did not collect oysters for him at that location; however, Dr. Ray did collect oysters on his own from the shallow intertidal areas of the reef. There is a big difference between dermo in the shallow water versus dermo in the deeper submerged reef. It is not appropriate to compare intertidal dermo results with submerged reef dermo results as intertidal oysters are more resilient and less infected from dermo, because they are out of the water a good bit of their life cycle. Submerged reefs have no chance of avoiding infection or predation in high salinity conditions. It is a mistake to compare all bay data as one big database even if it is to see how the old model predicts versus actual new data. In addition, the discussion about Confederate Reef's average dermo WP value of 1.7, or that values of 2.0 are present only for the short term basis and indicative of how much infection is present on reefs in Matagorda and Lavaca Bay systems seems in error. Figure 2-7o shows the potential relationship of the WP, from dermo at intertidal portion of Confederate Reef, with other data (counts, salinity, temperature) that were collected from the submerged part of Confederate Reef.

Section 2.7 Matagorda Bay Health Evaluation Suitability..., page 30 – Please consider that using Galveston Bay information to increase robustness in the data set may have been inappropriate due to the variance in rainfall and other factors which make Galveston more representative of the north central Gulf Coast than the mid-Texas Gulf Coast. It is recommended to reconsider using Galveston data for future efforts on this topic.

Section 2.9.1 Updated Dermo Model, page 35(top ¶) – Please consider mentioning other possible factors which may have contributed to the lack-of-fit in the updated dermo model such as the need for a more complex or multivariate model.

Section 3 Marsh Productivity Evaluation, page 37 – The contract-driven fall sampling effort would have benefited from consideration of other species that are at the right life stage to be collected by the gear. Please consider addressing why such species were not included in the analysis.

Section 3.2 Matagorda Bay Health Evaluation Habitat Model Overview, page 40 – Please consider the request that future categorizations such as those describing *Good, Fair, Poor, etc.* should be supported by the scientific literature or should be defined by a neutral panel of scientists. For assurances to the readers, please consider stating that the use of these categories is retained from the definitions determined for the MBHE for consistency between the two studies.

Section 5.4 Design Criteria Inflows, page 62 – Given that this study was mean to evaluate and improve the existing recommendations from the MBHE and BBEST/BBASC process, it is understandable that the same three-month seasonal categories were utilized. Please consider offering some recommendation, if appropriate and perhaps in the conclusions or future directions sections of the report, on the need to revise these seasonal categories to reflect the natural, actual seasonal patterns and the need to include some transitional months (*i.e.*, March) or to create inflow criterion that reflect a partial month.

Section 6.1.1 Potential New Freshet Component, pages 68-69 – The report recommends consideration of alternating between relatively large, infrequent freshet events and smaller, frequent freshet events to achieve the same dermo/oyster conditions. An example of such a freshet is provided; however, please consider providing an example of how "*higher subsequent inflows*" could be calculated and included in the freshet component.

Section 6.1.1 Potential New Freshet Component, page 70 (6th ¶) – This section discusses the potential impacts of long durations of low salinity on oyster survivorship. If possible, please consider addressing whether resource managers are concerned that the 2015 floods will have such an impact.

Section 6.4.1 Marsh and Reef Monitoring, page 75 (1st ¶) – Please consider adding a footnote or elsewhere in the report provide more detail as to the specific questions and concerns expressed by stakeholders which leads to the statement "*...to address questions and concerns of stakeholders voiced during the SB3 process.*"

Figures and Tables Comments

One reviewer commented that the use of the black and blue colors together are difficult to read (perhaps when printed in black and white) and requested consideration of using a different color scheme.

Table 2-4 – Please consider adding a footnote to explain how "OCI" is calculated.

Figures 2-13 and 2-14 – The top margin of the upper graphs have been cropped and are not fully visible (note the truncated axis label).

Figures 5-14 and 5-16 – Please consider renaming the Y-axis label to say "*Salinity (ppt) and Daily Inflow (1000 cfs)*".