Final Annual Report

Colorado River Flow Relationships to Bay Health: Benthic Indicators - 2007

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Preface

The lower Colorado River basin supports a diverse ecological community that relies heavily on the quality and quantity of water moving through the system. The wide range of variables and conditions associated with biological communities in the lower Colorado River and Matagorda Bay presents complexity in understanding its ecological processes. The LCRA-SAWS Water Project (LSWP) has the potential to alter the flow regime for the lower Colorado River and consequently Matagorda Bay; hence, the conduct of the Matagorda Bay Health Evaluation (MBHE) to assess the potential impact of these flow regime modifications. The "Colorado River Flow Relationships to Bay Health: Benthic Indicators" study is an integral part of the MBHE's objective to assess potential impacts/benefits on the aquatic resources of Matagorda Bay with and without the project and also quantify the condition of the aquatic environment under different flow scenarios to satisfy federal and state permitting requirements and ensure the environmental principles set forth for this project.

The benthic community is unique among estuarine organisms for several reasons. First, they are predominantly permanent residents of the bay, unlike much of the more visible nekton that are made up of large populations of migratory organisms. Second, they are relatively long-lived compared to plankton. Third, the benthos are relatively immobile and fixed in space, unlike nekton and plankton that move freely or with currents. This combination of characteristics means that the benthic community integrates changes in ecosystems over long time scales. Benthos are therefore a unique sentinel group, responding to changes in external conditions without the complication of movement to different regions of the estuary or the adjacent coastal zone. Many ecological monitoring programs use benthic abundance, biomass, and diversity as ecological indicators of productivity and health with respect to changes in the environment. Thus, it is necessary to assemble a long-term data set that can be used to define the quantity of freshwater inflow needed to maintain bay productivity and health.

Two of the environmental principles included in the LSWP contract relate directly to this study by stating that before project implementation, studies must prove that the project (1) "protects and benefits the lower Colorado River watershed and the LCRA Service Area, including municipal, industrial, agricultural, recreational, and environmental interests"; and (2) State law (HB 1629) requires that the LSWP "maintain the ecological health and productivity of the Matagorda Bay system."

Study objectives for 2007:

- Characterize the long-term trends of benthos of the Lavaca-Colorado Estuary.

- Characterize the flow-benthos and flow-ecological relationships within Lavaca and Matagorda Bays using field-gathered data in an existing bioenergetics model to provide prediction capabilities necessary to evaluate the full range of flows from low, to moderate, to high on ecological components of the lower Colorado River system throughout the annual hydrologic cycle.

- Characterize the relationship between flow, organic matter delivery, and respiration as an indicator of productivity in Matagorda Bay.

This report contains three chapters, which address each of the study objects.

Acknowledgements

Much of the data used in this report was funded by a variety of agencies over a long period of time. The Texas Water Development Board (TWDB) and Lower Colorado River Authority (LCRA) provided some funding in the past. The TWDB funded the original project where the model used here was developed. The current project is funded by LCRA for the LCRA-SAWS Water Project (LSWP) Contract Number 07-003.

This study was also partially supported by the Harte Research Institute for Gulf of Mexico Studies, Texas A&M University-Corpus Christi. The authors especially thank Mr. Richard D. Kalke for all his help and technical support during all phases of this work, especially sample collection and analysis. Ms. Carol Simanek played a vital role in data management.

Chapter 1

Long Term Benthic Macrofaunal Changes along a Salinity Gradient and Development of the Freshwater Inflow Biotic Index (FIBI)

By:

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A variety of biological monitoring techniques have been employed over the past several decades, and benthic indices of biological (or biotic) integrity (BIBIs) have been particularly useful for assessing aquatic systems. Until recently, however, most indices focused on assessing effects related to changes in water quality (e.g., the effects of chemical pollutants or eutrophication). Water quantity (i.e., the freshwater inflow regime), is also a key factor controlling ecosystem health in bays and estuaries. An index that would simply and effectively assess the environment based on salinity regimes and other hydrographic variables (as a proxy for measuring the effects of inflow) is needed to address issues of growing coastal populations that require development of new water resources. The goal of the present study was to (1) analyze benthic macrofaunal community structure and sediment and hydrologic parameters collected over space and time in the Lavaca-Colorado estuary, Texas, and to (2) develop a Freshwater Inflow Biotic Index (FIBI) for determining how changes in salinity regime (as a proxy for freshwater inflow) affect benthic populations, in turn reflecting the ecological condition of an estuary. Distinct benthic community differences in abundance, biomass, and diversity were found among the study stations, with higher benthic metrics observed at the intermediate and marine-influenced stations relative to the freshwater-influenced stations. Based on benthic succession theory and analysis of field-collected data, 12 biotic metrics were chosen that characterize the state of benthic community structure in response to inflow (i.e., salinity). The metrics were ranked and reduced to one variable using principal component analysis (PC) to form the FIBI. The FIBI PC variable was significantly correlated with hydrographic variables, indicating that benthic communities respond to changes in salinity and do so in a relatively predictable manner. If inflow is reduced (i.e., salinity is increased), it will cause the upstream communities to take on characteristics of downstream communities. The FIBI was successful in characterizing effects of a salinity gradient in the Lavaca-Colorado estuary, and application of the FIBI approach should be successful in other estuarine ecosystems.

INTRODUCTION

Concern over anthropogenic changes to the environment has grown over recent decades, and ever-increasing environmental legislation has brought worries about ecosystem health to the forefront of both scientific and political spheres. To address these concerns, it is necessary to develop novel scientific methods to assess, distinguish, and mitigate effects from natural and anthropogenic changes (Alber 2002; Bergquist et al. 2006). Because ecological interactions are complex, it is necessary to explain these relationships in a simple and easy-to-understand manner. Methods that both simplify the assessment process and are economically feasible are crucial to the success of ecosystem-based management strategies.

Environmental flows (i.e., hydrologic regimes in rivers, inflow from rivers to estuaries, and associated salinity changes) are one ecosystem service that is particularly vulnerable to water resource development. Historical studies have stressed the importance of environmental flows to estuarine systems, and its status as a major factor in estuary function and health has long-been established in coastal areas worldwide, including the Gulf of Mexico (Chapman 1966; Copeland 1966; Kalke 1981; Kalke and Montagna 1991). Freshwater inflows serve a variety of important functions in these habitats, including creation and preservation of low-salinity nurseries, sediment and nutrient transport, allochthonous organic matter inputs, and driving movement and reproductive timing of critical estuarine species (Longley 1994). Corresponding changes in salinity have also been demonstrated as primary factors controlling the distribution of marine and freshwater organisms within a Texas estuary (Mannino and Montagna 1997; Kalke and Montagna 1991; Montagna and Kalke 1992; Attrill et al. 1996; Montagna et al. 2002).

Compared with many other estuarine organisms, benthic macrofauna (body length > 0.5 mm) are especially sensitive to changes in freshwater inflow, and can be useful in determining its effects on estuarine systems over time. Although the effects of water flow are dynamic and vary over space and time, benthos are relatively long-lived and sessile; they continuously sample the overlying water conditions, integrating the ephemeral hydrologic conditions over time and providing a long-term record of short-term changes. Thus, benthic monitoring can be an important tool used to assess the ecological health and integrity of aquatic ecosystems.

A variety of benthic monitoring techniques have been employed over the past several decades. One important outcome from these programs is the “succession dynamics model” that describes how benthic community change is predictable with time since a disturbance, and in an analogous fashion with distance from a source of pollution (Pearson and Rosenberg 1978, Rhoads et al. 1978). Benthic indices of biological (or biotic) integrity (BIBIs) have been useful for assessment of aquatic systems that are under increasing levels of disturbance and environmental stress (Weisberg et al. 1997, Carr et. al. 2000, Llanso et. al. 2002, Maloney and Feminella, 2006). An index that could simply and effectively assess environmental condition based on salinity regimes and other hydrographic variables (as a proxy for measuring the effects of inflow directly) would be particularly valuable in creating effective management strategies for water resource development.

The present study examined macrofaunal and hydrographic data collected from the Lavaca-Colorado estuary on the central Texas coast in the western Gulf of Mexico. The goals of the present study were to (1) analyze benthic macrofaunal community structure and sediment and hydrologic parameters, and to (2) develop a Freshwater Inflow Biotic Index (FIBI) for determining how changes in salinity regimes (as a proxy for freshwater inflow) affect benthic populations, in turn reflecting the ecological condition of an estuary. Development of the FIBI was based on succession theory (Pearson and Rosenberg 1978, Rhoads et al. 1978), methodology employed by a variety of indices of biotic integrity (Karr 1991, Weisberg et al. 1997, Carr et. al. 2000, Llanso et. al. 2002, Maloney and Feminella, 2006), and well as other studies linking environmental variables with biological measures (Green and Montagna 1996, Peterson et al 1996, Long et al. 2003).

METHODS

Study Design and Area

Six stations representing a broad range of salinity habitats were chosen to assess effects of a salinity gradient in the Lavaca-Colorado estuary (Fig. 1.1; Table 1.1). In the western portion of the estuary, the Lavaca River generates a fresh-to-marine salinity gradient, with station A (28° 40′ 12″ N, 96° 34′ 48″ W) and B (28° 38′ 24″ N, 96° 34′ 48″ W) located in the highly freshwater-influenced portion of Lavaca Bay. Intermediate station C (28° 32′ 24″ N, 96° 28′ 12″ W) is located mid-way between Lavaca and Matagorda Bay, while marine-influenced station D (28° 28′ 48″ N, 96° 17′ 24″ W) is located near the Matagorda ship channel pass. In the eastern portion of the estuary, the Lower Colorado River generates another fresh to marine salinity gradient in Matagorda Bay, where stations D, E (28° 33′ 0″ N, 96° 12′ 36″ W), and F (28° 36′ 0″ N, 96° 02′ 24″ W) are located.

Although data exists for stations A – D from 1988 (Kalke and Montagna 1991; Montagna and Kalke 1995), all analyses for the current study were performed using a balanced design where stations A – F were sampled synoptically. This data was for the period of time when samples were collected at all stations: April 1993 - July 2000 and July 2004 - October 2007.

Field and Laboratory Measurements

Hydrographic measurements and water samples were collected at the surface and bottom of the water column at each station during each sampling period. A multiparameter instrument (Hydrolab Surveyor II or YSI 6 series) was used to measure water temperature, pH, dissolved oxygen, redox potential, salinity, and specific conductivity; salinity was also measured with a refractometer. Water samples were collected using a horizontally-mounted van Dorn Bottle. Benthic macrofauna were collected from each station during each sampling period using three replicate 6.7-cm diameter cores (35.4 cm^2 area). The cores were split vertically into surface (0-3 cm deep) and bottom (3 -10 cm deep) sections to allow examination of vertical distribution patterns; all samples were preserved with 5% buffered formalin. Sediment samples were collected during the fall quarterly sampling periods from 2001 – 2003 for geological analyses.

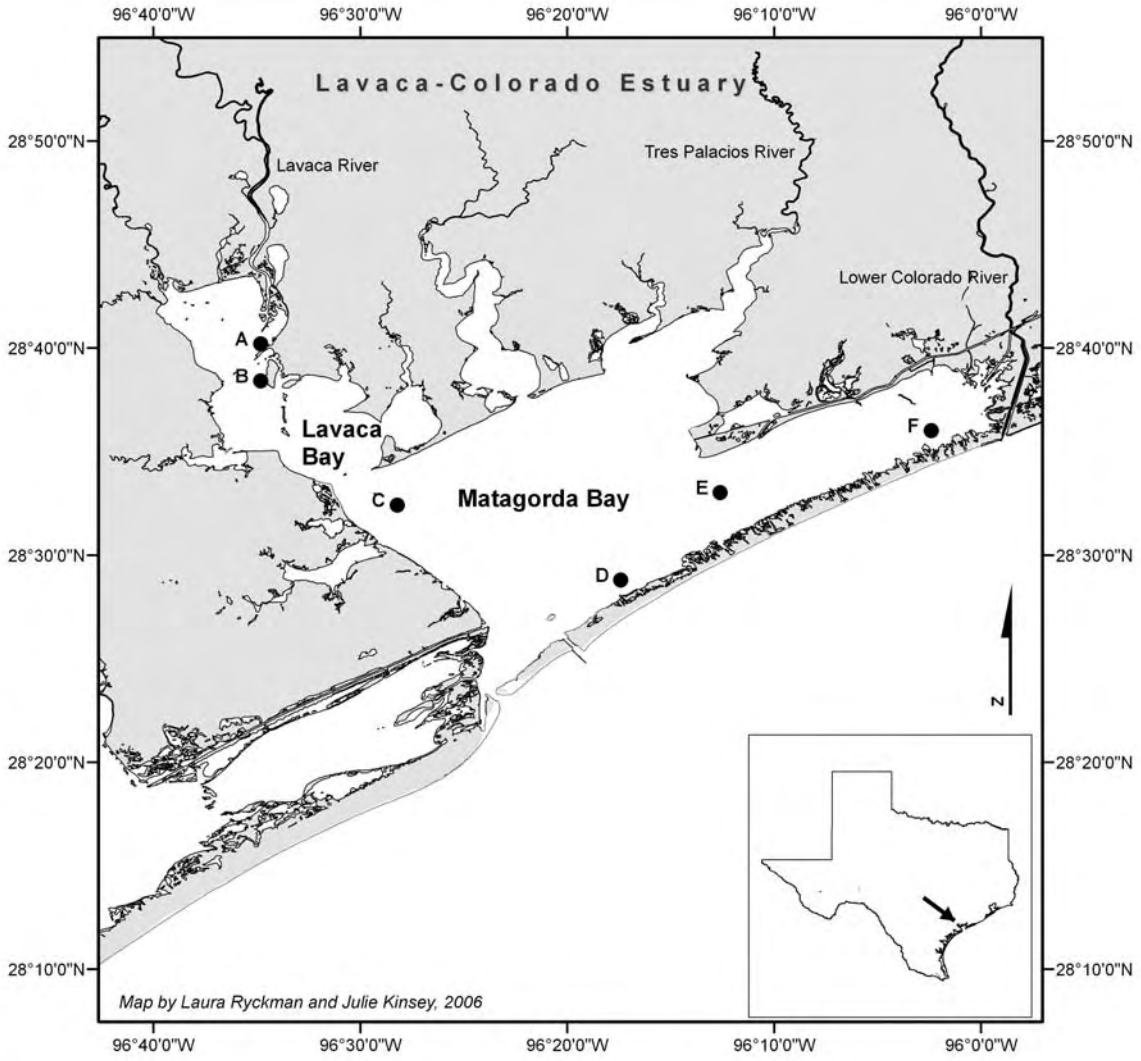

Figure 1.1. Study area and stations. Stations A-D are located along a salinity gradient from fresh to marine from Lavaca Bay to Matagorda Bay. Stations F-D are located along a salinity gradient from fresh to marine from East Matagorda Bay to Matagorda Bay.

Table 1.1. Summary of environmental and macrofauna community characteristicsfor Lavaca-Colorado estuary. Mean ± 1 standard deviation for the period April 1993 – July 2000 and July 2004 – October 2007.

		Station					
Variable type	Variable (unit)	A	B	C	D	E	F
Biological	Abundance (n m^{-2})	6795 (± 4212)	4685 (± 3100)	6780 (± 3873)	13590 (± 11433)	10421 (± 8231)	9650 (± 8657)
	Biomass (g m^{-2})	1.03 (± 0.89)	0.94 (± 1.04)	2.44 (± 2.31)	5.76 (± 5.28)	3.56 (± 3.07)	3.19 (± 4.18)
	Diversity (N1)	2.76 (± 0.90)	2.98 (± 0.99)	6.61 (± 2.28)	7.75 (± 2.99)	6.47 (± 2.52)	4.68 (± 2.87)
Water Column	DO (mg l^{-1})	7.61 (± 1.62)	7.44 (± 1.51)	7.24 (± 1.49)	6.56 (± 1.74)	6.45 (± 2.27)	7.61 (± 2.76)
	Salinity (psu)	12.26 (± 8.95)	15.65 (± 9.15)	21.97 (± 7.88)	26.63 (± 4.62)	24.01 (± 5.85)	18.59 (± 8.4)
	Temperature (˚C)	21.98 (± 6.46)	22.00 (± 6.62)	22.31 (± 6.43)	22.40 (± 6.05)	22.72 (± 6.59)	23.04 (± 6.38)
	pH	8.14 (± 0.89)	8.13 (± 0.64)	8.09 (± 0.35)	8.09 (± 0.38)	8.14 (± 0.44)	8.23 (± 0.48)
	NH_4 (µmol l^{-1})	2.58 (± 2.15)	2.38 (± 1.97)	1.97 (± 1.93)	1.97 (± 2.29)	2.14 (± 2.58)	3.51 (± 4.27)
	NO_3 + NO_2 (µmol l^{-1})	5.15 (± 8.87)	4.19 (± 8.33)	1.85 (± 4.12)	0.88 (± 1.49)	1.49 (± 2.53)	7.78 (± 13.99)
	PO_4 (µmol l^{-1})	1.48 (± 1.21)	1.23 (± 1.05)	0.92 (± 0.62)	0.84 (± 0.59)	1.27 (± 0.95)	2.08 (± 2.12)
	SiO_4 (µmol l^{-1})	88.13 (± 49.08)	76.41 (± 45.31)	54.73 (± 39.24)	36.87 (± 26.32)	52.45 (± 38.77)	70.97 (± 43.85)
	Chl a (mg l^{-1})	6.36 (± 3.64)	6.68 (± 5.28)	6.17 (± 5.63)	6.09 (± 4.78)	6.84 (± 6.98)	9.83 (± 5.26)
Sediment	Rubble (%)	0.01 (± 0.00)	0.01 (± 0.01)	0.02 (± 0.01)	0.01 (± 0.01)	0.01 (± 0.00)	0.01 (± 0.01)
	Sand (%)	0.35 (± 0.12)	0.20 (± 0.04)	0.29 (± 0.13)	0.18 (± 0.04)	0.06 (± 0.03)	0.20 (± 0.12)
	Silt (%)	0.21 (± 0.06)	0.26 (± 0.03)	0.21 (± 0.04)	0.26 (± 0.04)	0.22 (± 0.06)	0.34 (± 0.08)
	Clay (%)	0.43 (± 0.08)	0.52 (± 0.06)	0.48 (± 0.10)	0.54 (± 0.04)	0.71 (± 0.05)	0.45 (± 0.08)
	Porosity (%)	0.52 (± 0.09)	0.64 (± 0.05)	0.58 (± 0.09)	0.60 (± 0.03)	0.68 (± 0.03)	0.58 (± 0.07)
	TOC (%)	0.62 (± 0.21)	0.92 (± 0.21)	0.74 (± 0.23)	0.92 (± 0.08)	0.98 (± 0.11)	0.76 (± 0.18)
	C content (%)	0.88 (± 0.20)	1.50 (± 0.20)	1.35 (± 0.24)	1.69 (± 0.02)	2.98 (± 1.84)	2.09 (± 0.26)
	$\delta^{13}C$ (ppt)	-13.04 (± 0.84)	-14.98 (± 1.38)	-10.97 (± 2.16)	-11.77 (± 0.45)	-10.02 (± 4.21)	-9.60 (± 0.85)
	$\delta^{15}N$ (ppt)	6.92 (± 0.13)	7.38 (± 0.52)	7.48 (± 0.30)	7.73 (± 0.36)	8.29 (± 0.32)	7.71 (± 0.39)
	N content (%)	0.06 (± 0.02)	0.12 (± 0.04)	0.08 (± 0.04)	0.13 (± 0.00)	0.12 (± 0.04)	0.10 (± 0.03)

In the laboratory, water samples for chlorophyll a (Chl a) analysis were filtered onto glass fiber filters and placed on ice (< 0.4 ˚C). Chl a was extracted overnight with methanol and read fluorometrically on a Turner Model 10-AU using a non-acidification technique (Welschmeyer, 1994; EPA method 445.0). Analysis of nutrients was performed using a LaChat QC 8000 ion analyzer with computer-controlled sample selection and peak processing.

Benthic macrofauna from the core samples were extracted on a 0.5 mm sieve, sorted using a dissecting microscope, and identified to the lowest taxonomic level (usually species), and enumerated. Biomass measures were obtained by combining individual macrofauna into higher taxa levels (i.e., Crustacea, Mollusca, Polychaeta, and others), drying at 50 ˚C for 24 h, and then weighing. Mollusk shells were removed with 1 N HCl prior to drying and weighing.

Sediment samples were analyzed for percent contribution by weight of rubble, sand, silt, and clay. A 20-cm^3 sediment sample was mixed with 50 ml of hydrogen peroxide and 75 ml of deionized water to digest organic material in the sample. The sample was wet sieved through a 62 μm mesh stainless steel screen using a vacuum pump and a Millipore Hydrosol SST filter holder to separate rubble and sand from silt and clay. After drying, the rubble and sand were separated on a 125 μm screen. The silt and clay fractions were measured using pipette analysis (Folk 1964). Percent porewater, total carbon (C), total nitrogen (N), total organic carbon (TOC), $\delta^{15}N$, and $\delta^{13}C$ values were also measured. Total N, C, and TOC were measured with a Carlos Erba NC 2500 elemental analyzer. A ThermoElectron Deltaplus mass spectrometer (IRMS) was used to measure isotopic ratios.

Statistical analyses - Field and laboratory measurements

Diversity measures evaluated in this study included species diversity (N1 and H′), and evenness (J′). Species diversity was calculated using Hill's index N1, which is a measure of the effective number of species and indicates the number of abundant or dominant species in a sample (Hill 1973, Ludwig and Reynolds 1988). N1 is the exponential form of the Shannon diversity index (H′): $N1 = e^{H'}$ and will tend toward one as diversity decreases. The evenness index, Pielou's J′ (Pielou, 1966), expresses H′ relative to the maximum value of H′. Evenness is an index that expresses the distribution of species within a sample, and reaches a maximum when all species are equally abundant; in contrast, the less even the community, the more rare species are present. To reduce the influence of rare species, all replicates were pooled in order to effectively examine the number of species in one large sample rather than three small samples.

Principal Component Analysis (PCA) was used to assess relationships between sediment variables and hydrologic variables. PCA reduces a multivariate data set and creates new variables by extracting variance in order of importance. Results of the analysis are a new set of PC variable loading scores, and sample scores. The loadings represent the underlying structure of the dataset, and the scores represent the contribution of each sample. The higher the absolute values of the PC loading scores, the more influence the variable has in the new

PC variable. Results are presented in plots of the vectors of the PC loads to aid interpretation of the underlying structure, and sample scores to visualize spatial and temporal comparisons.

Linear correlations were performed between sediment and water-column PCAs and a set of biological variables including biomass (g m^{-2}), abundance (n m^{-2}), diversity (N1 and H′), and evenness (J′). Non-metric multidimensional scaling (MDS) was used to illustrate similarities between communities on different sampling periods or between stations. For MDS analysis, the Primer software creates a Bray-Curtis similarity matrix among all samples, followed by an MDS plot of the spatial relationships between the samples.

Freshwater Inflow Biotic Index (FIBI)

The macrofaunal and hydrographic data collected from the Lavaca-Colorado estuary were used to create a freshwater inflow biotic index (FIBI) to determine the primary factors driving macrofaunal community structure in different salinity regimes. The approach is similar to that which has been used in other biological indices, i.e., a set of metrics are chosen based on their ability to reflect biological, chemical, or physical attributes of ecological condition (US EPA 2000). It is desirable to identify metrics that display a consistent response to a given stressor or disturbance (Karr 1991, Weisberg et al. 1997, Llanso et. al. 2002). For development of the FIBI, the "stressor" was salinity as a surrogate for freshwater inflow effects. In accordance with other index methods, a scoring system for the metrics was used to classify the ecological condition at a specific location and sampling period. Because the variables must be expressed in the same scale for the scoring system to be useful, a ranking system was used.

Twelve biological metrics were included in the index: biomass (g m^{-2}), abundance (n m^{-2}), diversity (N1), % freshwater indicator species, % brackish water indicator species, % marine indicator species, % biomass 3-10 cm below the sediment-water column interface, % abundance 3-10 cm below the sediment-water column interface, % diversity 3-10 cm below the sediment-water column interface, % predators, % deposit feeders, and % sediment interface (water column) feeders. Diversity was chosen as a metric because high values have been associated with healthy ecosystem function (Thébault and Loreau, 2006). High biomass, abundance, and diversity in the deeper sediment layer (3 - 10 cm below the sediment-water column interface) have also been shown to be associated with later successional stages and therefore relatively undisturbed, high quality communities (Pearson and Rosenberg 1991; Carr et. al. 2000). In the long-term study (Kinsey 2006), biomass, abundance, and diversity were all positively correlated with salinity, as were all trophic groups (interface feeders, predators, and deposit feeders, defined by Tenore et al. 2006). Freshwater, brackish and marine indicator species were also included. All other variables were either found to be significantly correlated with salinity or were known to be associated with changes in salinity or inflow (Kalke and Montagna 1991; Montagna and Kalke 1992, 1995, Ritter et al. 2005). Sediment characteristics were not included as metrics because they were not found to significantly differ among stations, nor were they correlated with any of the biological variables in the long-term study (Kinsey 2006).

Freshwater inflow indicator species were categorized as either marine indicative, brackish indicative, or freshwater indicative. Abundance values within each category were summed, and the percent abundance in each indicator category with respect to total abundance of all species was calculated. Next, the mean values for abundance (n m^{-2}), biomass (g m^{-2}), and diversity (N1) were calculated. Total abundance, biomass, and number of species located from 3 - 10 cm below the sediment-water column interface were also calculated. Species-level data was then organized into trophic guilds as defined by Tenore et al. (2006). The three trophic metrics for the FIBI were designated as the % predators, % deposit feedeers, and % sediment interface feeders with respect to the total abundance of all trophic guilds combined.

Statistical Analyses - FIBI

Values for each of the metrics for each sampling period and station were ranked in groups of five (0 - 4) using the PROC RANK procedure in SAS (1996). Using these ranked values, the metrics were analyzed using the PROC FACTOR procedure in SAS in the principal components (PC) analysis mode. The PROC FACTOR procedure was used in lieu of PROC PCA because the former produces data sets of both vector loads and sample scores, while the latter only outputs vector loads (Carr et. al. 2000, Long et. al. 2003). Therefore, comparisons could be made among the different loading variables, and among stations and sampling periods. The new PC variables represent the FIBI because all the biological metrics are reduced to just two new variables, which represent the underlying structure of the dataset due to environmental variability.

To link the hydrographic data with the biotic data, linear correlations were run between FIBI and hydrographic PC factor 1 and PC factor 2 scores using the PROC CORR procedure in SAS. This method of linking reduced environmental variables with reduced biological variables has been used successfully in previous sediment toxicity studies (Green and Montagna 1996; Carr et al. 2000), and is outlined in detail by Montagna in Porewater Toxicity Testing (Long et al. 2003).

RESULTS

Average salinities ranged from 12.3 to 26.6 psu, increasing continuously from the freshwater-influenced portions of the study area to the marine (Table 1.1, Fig. 1.1). The highest mean salinity occurred at marine station D, while the lowest mean salinities were found at the more freshwater-influenced stations A, B, and F. Average dissolved oxygen levels decreased continuously along the increasing salinity gradient. Mean values ranged from 6.5 to 7.6 mg l^{-1} (Table 1.1). The highest mean values occurred at the freshwater-influenced stations A and F, while the lowest mean values were found at intermediate station E and marine station D. Temperatures and sediment characteristics were uniform across all stations; nutrients (except for PO_4) and chl a declined with increasing salinities. Sediment characteristics were similar across all stations.

Mean abundance, biomass and diversity generally increased along the salinity gradient from fresh to marine. Mean biomass was lowest (0.94 g m^{-2}) at freshwater-influenced station B and greatest at station D (5.76 g m^{-2}), located in the most marine-influenced area (Table 1.1, Fig. 1.2). The lowest mean abundance was also found at station B (4,685 n m^{-2}) and the highest at station D (13,590 n m^{-2}; Table 1.1, Fig. 1.3). Mean diversity (N1) followed the same pattern, with the lowest values occurring at stations A & B (2.76 and 2.98, respectively), and the highest at station D (7.75; Table 1.1, Fig 1.4).

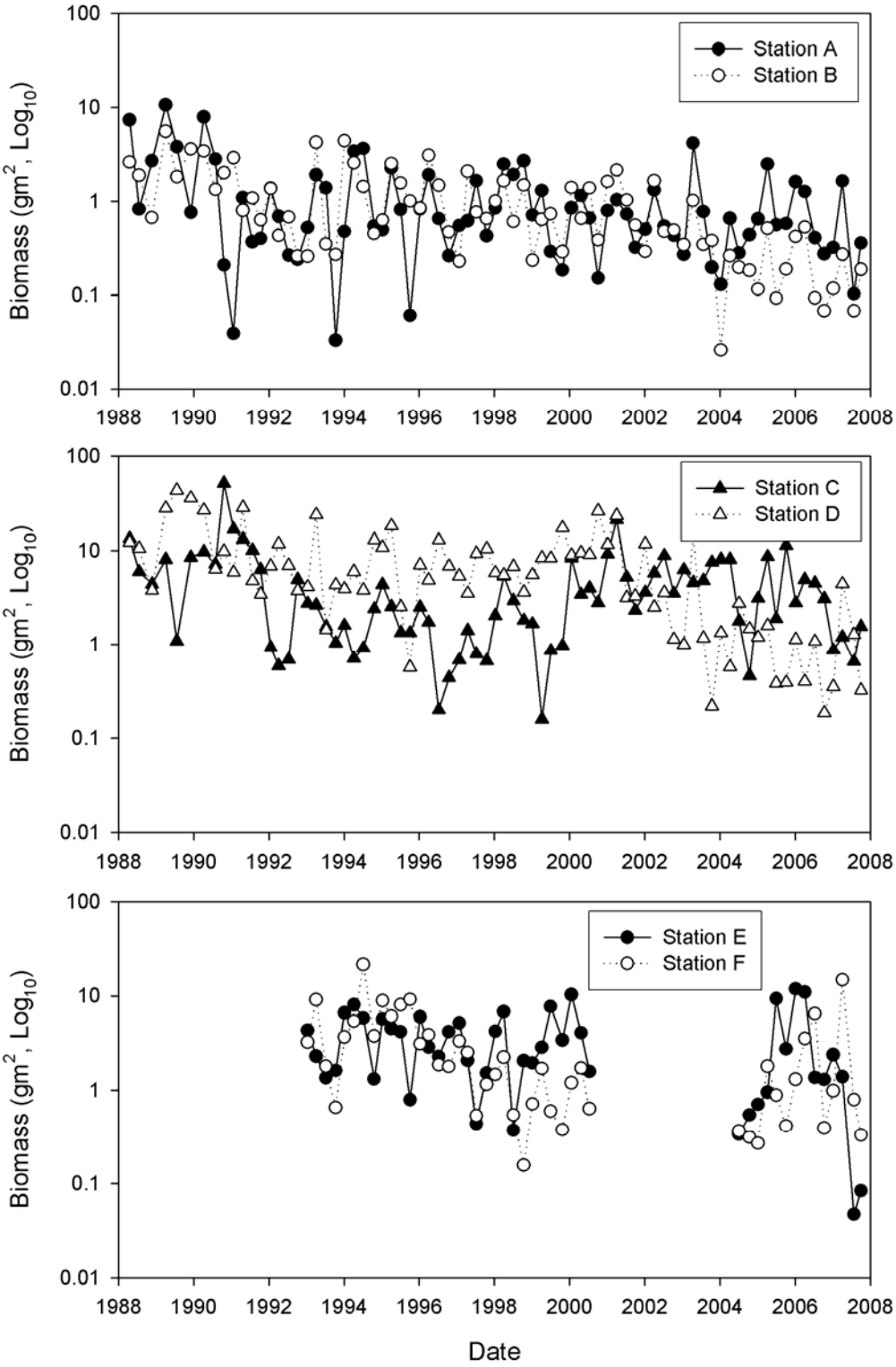

Figure 1.2. Biomass (g m^{-2}, $\log_{10}$) at stations A-F from April 1988 - October 2007.

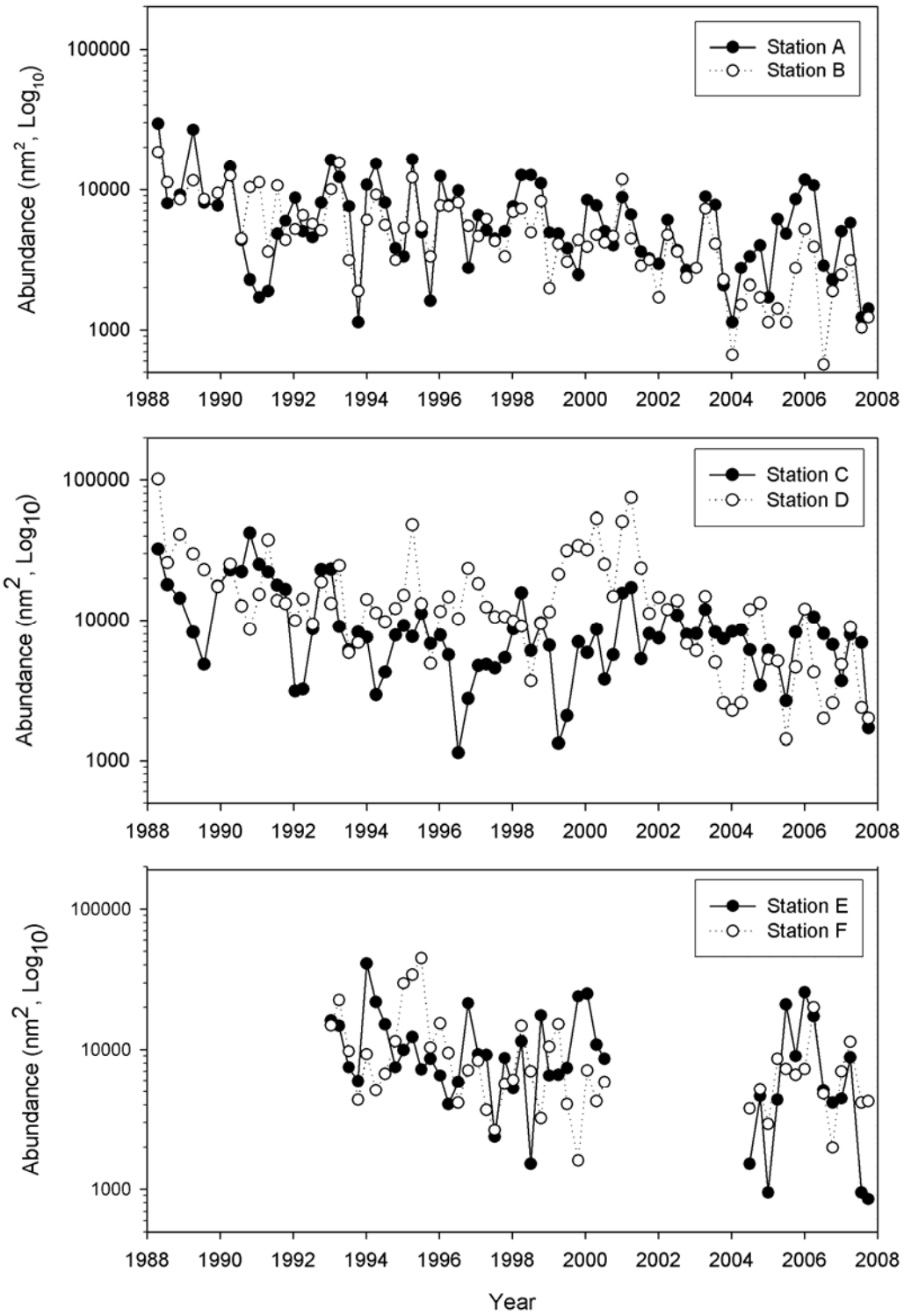

Figure 1.3. Abundance (n m^{-2}, log_{10}) at stations A-F from April 1988 - October 2007.

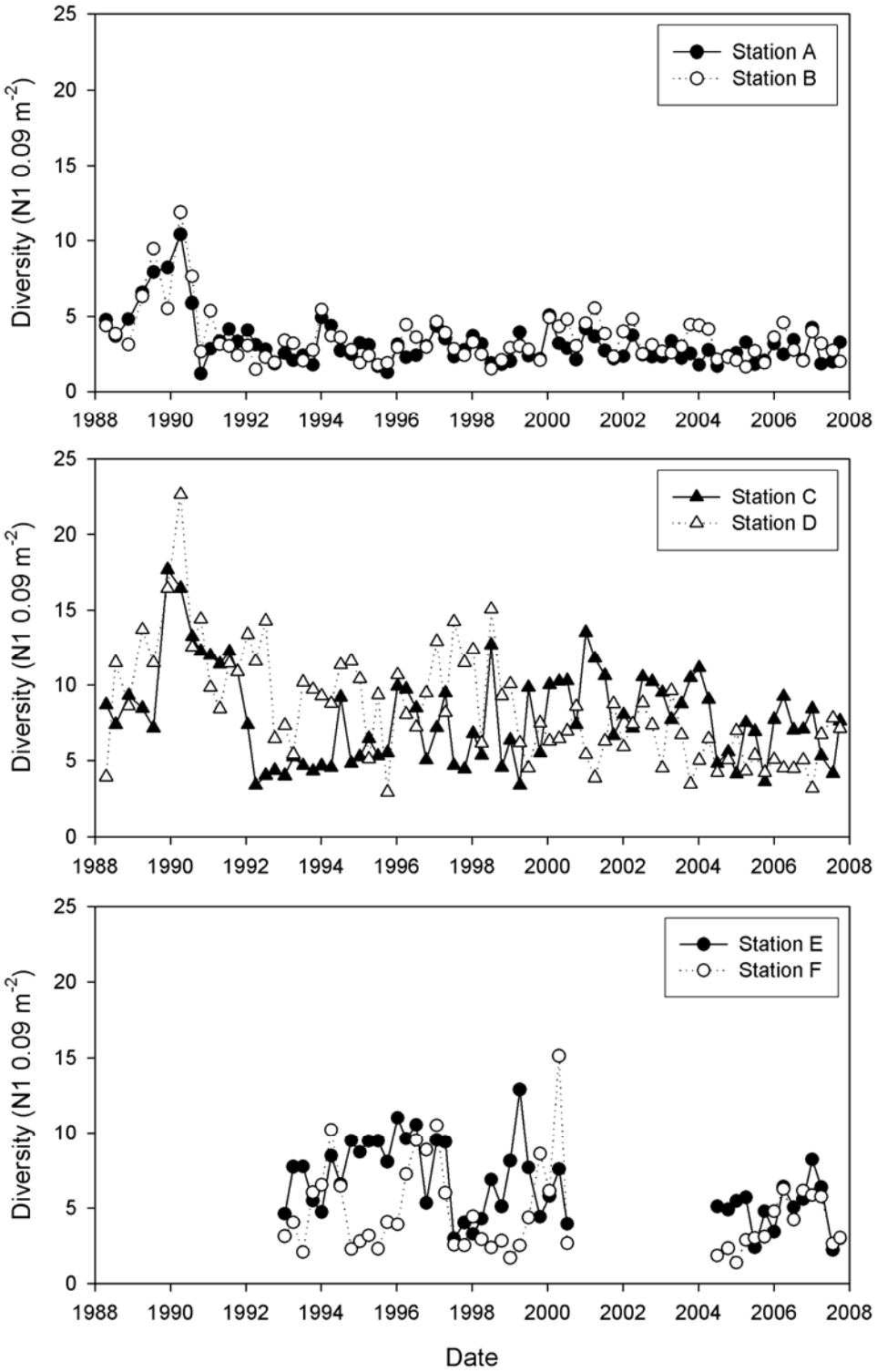

Figure 1.4. Diversity (N1) at stations A-F from April 1988 - October 2007.

Over 70% of all species found comprised polychaetes, while crustaceans constituted 5% and bivalves 7% (Table 1.2). The polychaete *Mediomastus ambiseta* dominated overall, representing 45% of the entire species pool. The next most abundant species were the polychaetes *Polydora caulleryi* at 9% and *Streblospio benedicti* at 8%. Out of a total of 202 species, 22 species represented 91% of all of the individuals found over all stations.

Freshwater inflow indicator species were categorized as freshwater-indicative, brackish-indicative or marine-indicative (Table 1.3). Freshwater indicator species were all aquatic insects. Most of the indicators are chironomid species of different life stages. Although aquatic insects are the best indicator of fresh conditions, these organisms were never dominant, constituting < 0.1% of the total organisms sampled.

Five brackish-water indicator species were found in the species dominance results (Table 1.2, 1.3). The polychaete *Streblospio benedicti* and the crustacean *Ampelisca abdita* both declined continuously from freshwater station A to marine station D, and then increased again from intermediate station E to freshwater station F. The bivalve *Macoma mitchelli* also exhibited the same pattern, with the exception that stations D and E had equal values. The bivalve *Mulinia lateralis* showed a less distinct pattern in that values were relatively similar among the fresh and intermediate stations, but a large decline in individuals occurred at the marine station.

Five marine indicator species were also found in species dominance analysis (Table 1.2, 1.3). The crustacean *Apseudes* sp. A increased continuously along the freshwater to marine salinity gradient. Bivalves *Corbula contracta* and *Periploma* cf. *orbiculare* also exhibited a strong increase along the salinity gradient, as did the polychaete *Minuspio cirrifera* and the brittle star *Amphiodia atra*.

The first and second principal components (PC 1 and PC 2) for sediment variables explained 92% and 7% of the variation within the data set (total 99%; Fig. 1.5). The system was characterized by percent clay, rubble, and sand, thus PC 1 represents a scale of sediment grain size. Variability in $\delta^{13}C$ and C% was largely driven by one sample from station E. PC 1 scores for percent clay and $\delta^{15}N$ were generally opposed to percent sand and rubble, meaning that as the proportions of clay and $\delta^{15}N$ increase, the proportions of sand and rubble decrease. Station A tended to have high proportions of sand and rubble and low levels of $\delta^{15}N$ and percent clay relative to the other stations, and station D displayed no strong relationships with any of the variables. No significant correlations were found between the sediment PCA and biological variables (biomass, abundance, diversity, or evenness; Table 1.4).

Table 1.2. Dominant species for stations A - F. Species are ranked in order of decreasing mean abundance over the analysis period.

		Station								
Rank	Species Name	A	B	C	D	E	F	Mean	%	Cum%
1	*Mediomastus ambiseta*	4,509	3,065	3,292	4,326	3,811	4,238	3,874	44.80%	44.80%
2	*Polydora caulleryi*	0	0	162	756	1,826	2,160	817	9.40%	54.20%
3	*Streblospio benedicti*	1,128	668	416	244	391	1,042	648	7.50%	61.70%
4	*Apseudes* sp. A	0	0	2	2,408	8	0	403	4.70%	66.40%
5	*Cossura delta*	17	137	456	416	599	225	308	3.60%	69.90%
6	*Mulinia lateralis*	290	204	330	29	656	130	273	3.20%	73.10%
7	Oligochaeta (unidentified)	34	11	21	1,006	195	23	215	2.50%	75.60%
8	Nemertea (unidentified)	103	95	202	475	223	134	205	2.40%	77.90%
9	*Minuspio cirrifera*	0	0	32	693	174	6	151	1.70%	79.70%
10	*Paraprionospio pinnata*	2	27	181	118	347	128	134	1.50%	81.20%
11	*Gyptis vittata*	4	15	151	137	357	134	133	1.50%	82.80%
12	*Corbula contracta*	0	0	0	538	4	0	90	1.00%	83.80%
13	*Lepton* sp.	0	0	0	508	2	6	86	1.00%	84.80%
14	*Amphiodia atra*	0	0	57	275	95	15	74	0.80%	85.70%
15	*Glycinde solitaria*	27	38	134	57	113	71	74	0.80%	86.50%
16	*Schizocardium* sp.	0	0	40	82	258	55	72	0.80%	87.40%
17	*Ampelisca abdita*	97	27	11	2	6	277	70	0.80%	88.20%
18	*Macoma mitchelli*	134	105	13	4	4	151	69	0.80%	89.00%
19	*Lumbrineris parvapedata*	0	0	101	67	88	13	45	0.50%	89.50%
20	*Haploscoloplos foliosus*	25	46	76	6	27	48	38	0.40%	89.90%
21	*Periploma* cf. *orbiculare*	0	0	4	206	17	0	38	0.40%	90.40%
22	*Nuculana acuta*	2	0	44	21	118	11	33	0.40%	90.70%
	180 other species	422	246	1,057	1,214	1,097	775	802	9.30%	100.00%
	Total	6,795	4,683	6,780	13,590	10,417	9,644	8,651		

TABLE 1.3: Indicator species for the Freshwater Inflow Biotic Index (FIBI).

Inidicator Species		
Indicator Type	Taxa	Species Name
Freshwater	Insecta	Chironomidae (larvae)
	Insecta	Chironomidae (pupae)
	Insecta	Chironomidae (unidentified)
	Insecta	Diptera (unidentified)
	Insecta	Insecta (unidentified)
	Insecta	*Pentneura* sp. (larvae)
Brackish	Crustacea	*Ampelisca abdita*
	Mollusca	*Macoma mitchelli*
	Mollusca	*Mulinia lateralis*
	Polychaeta	*Parandalia ocularis*
	Polychaeta	*Streblospio benedicti*
Marine	Crustacea	*Apseudes* sp. A
	Mollusca	*Corbula contracta*
	Mollusca	*Periploma* cf. *orbiculare*
	Ophiuroidea	*Amphiodia atra*
	Polychaeta	*Minuspio cirrifera*

Principal components 1 and 2 (PC 1 and PC 2) for the water-column variables explained 47% and 20% of the variation within the data set (total 67%; Fig. 1.6). The eigenvalue for the third component is only 0.36, which is far below the conventional value of 1.0, so only the first two PCs are used in the analysis. The PC 1 variable loads had the highest positive values for oxidized inorganic nitrogen ($N+N = NO_2 + NO_3$), ammonium (NH_4), and phosphate (PO_4), and the highest negative values for salinity. This means that a decrease in salinity (or increase in freshwater inflow) is associated with an increase in nitrogen and phosphorus nutrients; thus PC 1 represents a linear scale of freshwater inflow effects on hydrology. Dissolved oxygen (DO) and temperature exhibited secondary roles; PC 2 thus represents seasonal effects where increases in temperature result in decreases in DO concentrations. Positive PC 2 variable loads for silicate (SiO_4) were likely due to increased sediment re-suspension as a function of seasonal wind patterns. Station loading scores were distributed in a fairly distinct spatial pattern along the salinity gradient, with station A generally exhibiting the most negative relationship with salinity and station D the most positive. PC 1 scores were significantly correlated with diversity (N1, p 0.001, r=-0.289; H′ p 0.001, r=-0.310) and evenness (p 0.023, r=-0.208; Table 1.4). The relatinshipPC 2 scores were significantly correlated with biomass (p 0.001, r=-0.305), and diversity (N1, p <0.001, r= -0.407; H′ p <0.001, r=-0.408; Table 1.4).

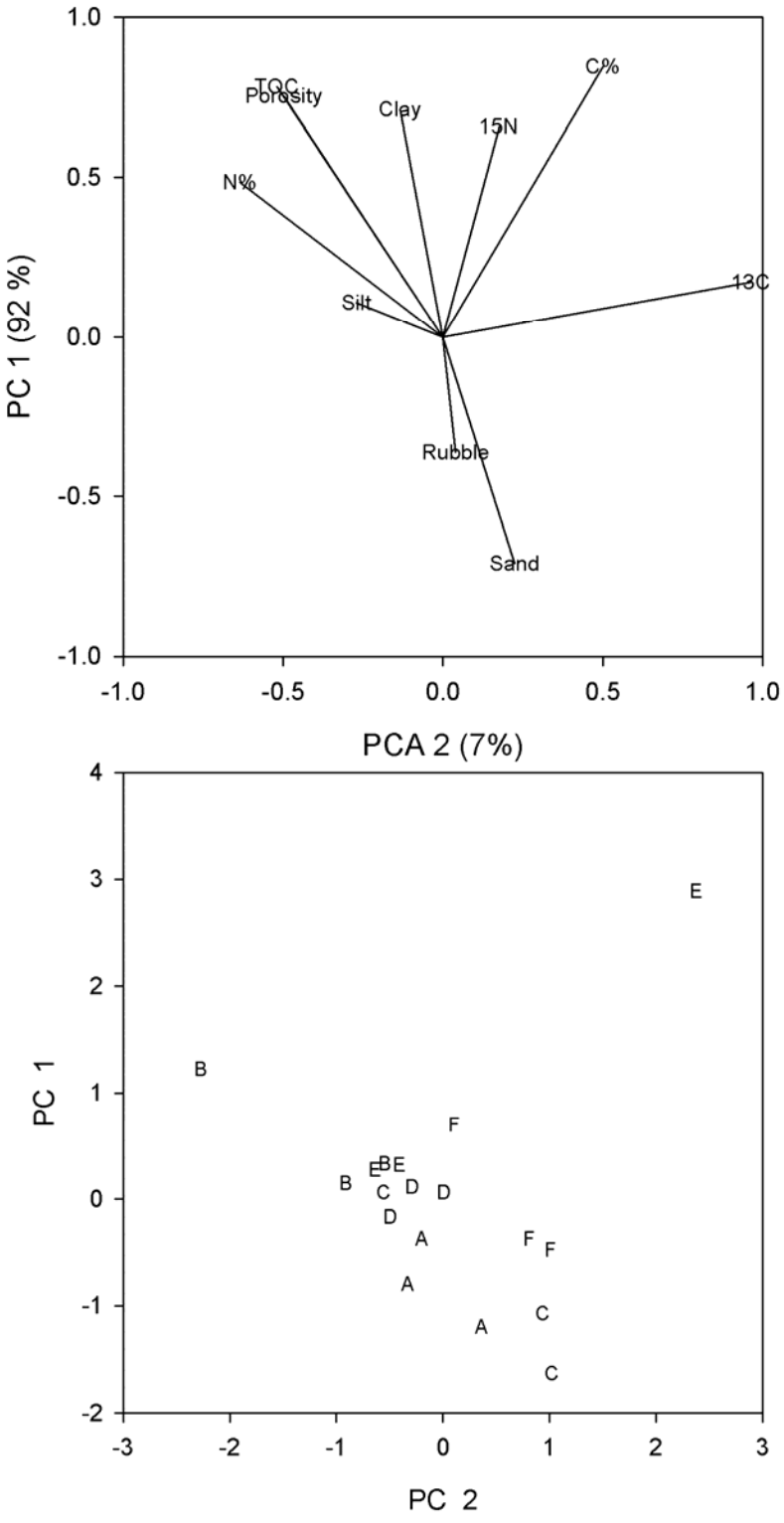

Figure 1.5. PCA variable loads (top) and station-date scores (bottom) for sediment characteristics stations A - F. Abbreviations: TOC = percent total organic carbon, 15N = $\delta^{15}N$, C% = percent carbon, N% = percent nitrogen, 13C = $\delta^{13}C$.

Table 1.4. Linear correlations between sediment (Figure 1.5) and hydrologic PC (Figure 1.6) factor scores and biological variables. Abbreviations: r = Pearson product correlation coefficient, P = probability of the null hypothesis, and n = number of sample pairs.

		Sediment		Water Column	
Benthic Metric		PC1	PC2	PC1	PC2
Biomass (g m^{-2})	r	-0.166	0.406	-0.164	-0.305
	P	0.510	0.094	0.075	0.001
	n	18	18	119	119
Abundance (n m^{-2})	r	-0.290	0.311	-0.055	0.059
	P	0.242	0.209	0.556	0.525
	n	18	18	119	119
Diversity (N1)	r	0.055	0.445	-0.289	-0.407
	P	0.829	0.064	0.001	<.0001
	n	18	18	119	119
Diversity (H′)	r	0.159	0.452	-0.310	-0.408
	P	0.528	0.060	0.001	<.0001
	n	18	18	119	119
Evenness (J′)	r	0.193	0.291	-0.208	0.122
	P	0.442	0.242	0.023	0.187
	n	18	18	119	119
FIBI PC1	r	0.320	0.179	-0.198	-0.068
	P	0.196	0.478	0.033	0.464
	n	18	18	117	117
FIBI PC2	r	-0.302	0.386	-0.150	-0.515
	P	0.222	0.114	0.106	<.0001
	n	18	18	117	117

Multidimensional scaling (MDS) analysis of community structure showed that benthic communities are generally spatially distinct along the salinity gradient of the study area (Fig. 1.7). Macrofaunal communities were divided into 5 zones (1-5) with at least 30% similarity among stations in each zone; stations tended to group left to right with increasing salinities. Zone 1 contained all of the samples collected at freshwater-influenced stations A and B. Zone 1 also overlapped with zone 2, coinciding with a majority of the samples from intermediate stations C and E. Zone 2 also contained most of the samples from marine station D. Station F oscillated between freshwater and intermediate community structure, while station E alternated between intermediate and marine community structure. Samples that grouped into zones 3, 4 and 5 were primarily due to differences in seasonal sampling. Zone 3 comprised samples collected in July and October from station E, while zone 4 included one sample collected in April at station C and Zone 5 included two samples collected in January from station F.

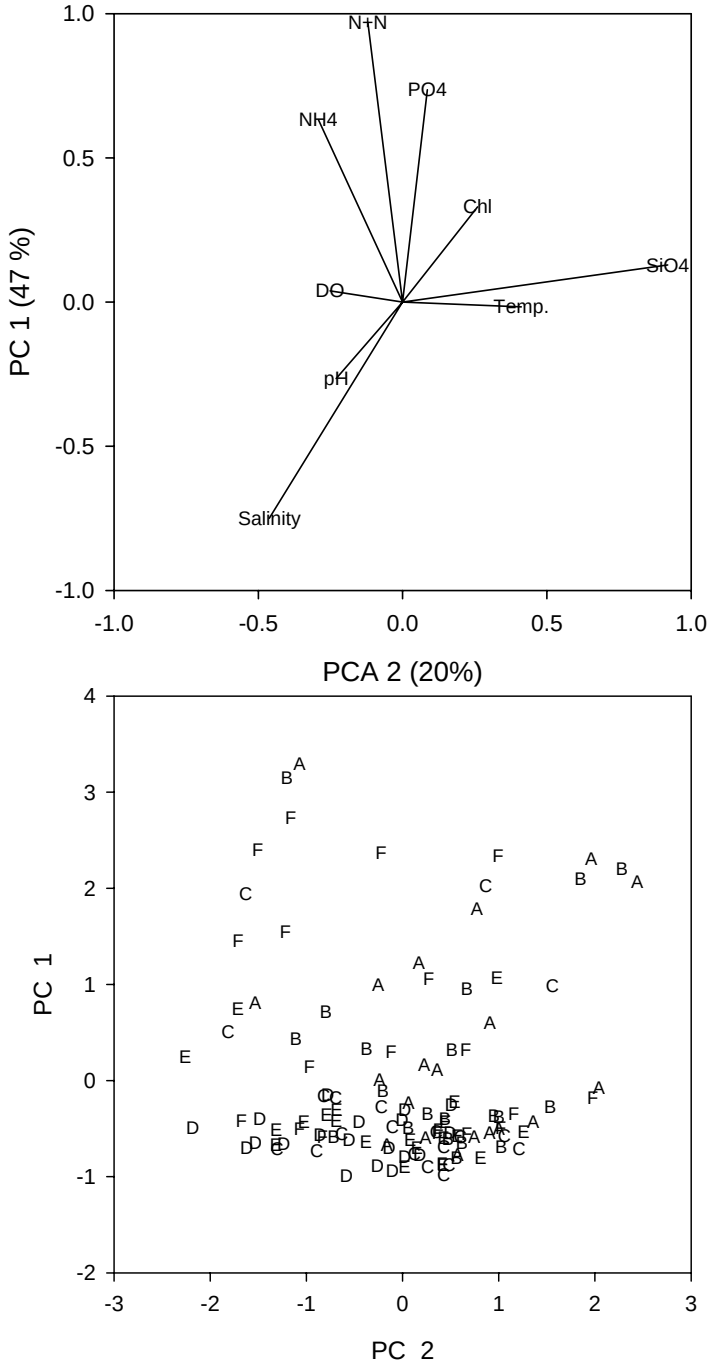

Figure 1.6. PCA variable loads (top) and stations-date scores (bottom) for hydrographic characteristics, stations A-F. DO = dissolved oxygen, Temp = temperature, SiO_4 = silicate, PO_4 = phosphate, NH_4 = ammonium, N+N = total nitrogen, Chl = chlorophyll a.

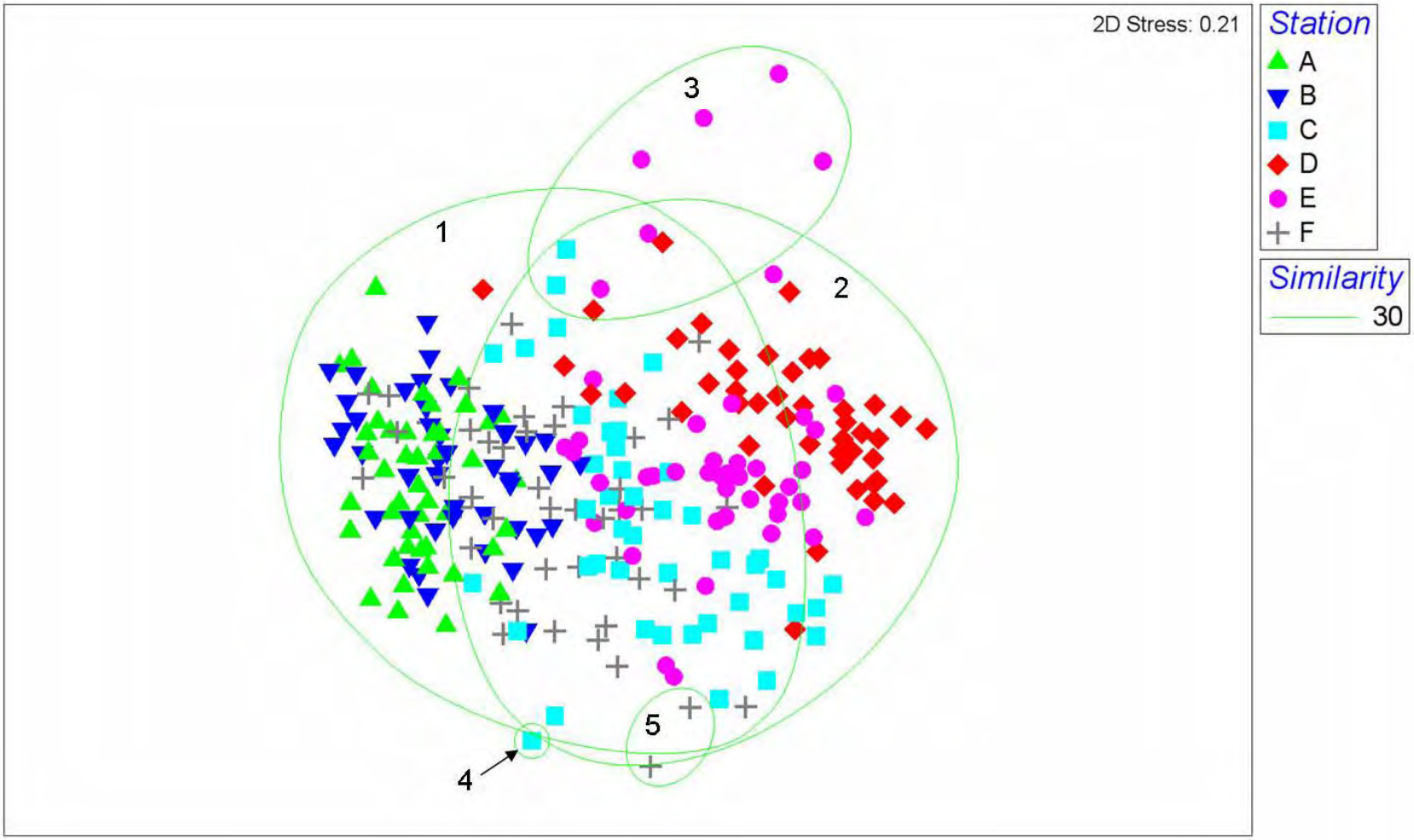

Figure 1.7. MDS analysis of species-level community structure including all sampling periods and stations.

The first and second principal components (PC 1 and PC 2) for the FIBI metric ranks explained 38% and 17% of the variation within the data set (total 55%; Fig. 1.8). The PC 1 variable loads had the highest positive values for abundance, biomass, and diversity 3-10 cm below sediment-water interface (%Abu>3, %Div>3, %Bio>3), and percent marine indicator (%Mar) species; the highest negative value was for percent brackish indicator (%Brk) species. The loads for %Mar, abundance, biomass and diversity (all at 3-10 cm sediment depth) were positive, meaning that an increase in %Brk would result in a decrease in %Mar, and abundance, biomass, diversity at 3-10 cm depth (and vice versa). All PC 1 loads except for percent freshwater indicators (%FW), deposit feeders (%Dep), and N1 diversity were above an absolute value of 0.5, indicating that all of the metrics along PC 1 except these 3 were relatively important. The PC 2 variable loads had the highest absolute values for overall biomass and abundance. All other metrics' PC 2 loads were below the absolute value of 0.5, indicating that those metrics were not as influential along the PC 2-axis.

A significant correlation was found between FIBI PC 1 and the hydrologic PC 1 scores (p 0.005, r =-0.271; Table 1.4), and also between FIBI PC 2 and hydrologic PC 2 loads ($p <$ 0.001, r = -0.448). These significant correlations indicated that the biological variables are directly related to the hydrology of the system. No significant relationships were found between FIBI PC 1 and hydrographic PC 2 loads or among FIBI PC 2 and hydrographic PC 1 loads. There was also no significant relationship between either of the FIBI PC loads and either of the sediment variable PC loads.

The main driver of the water PC 1 is salinity, and the main drivers of the FIBI PC1 is biomass and diversity (Table 1.4). Although the Pearson correlation between biomass and diversity with water PC1 was significant, the relationships are non-linear (Fig. 1.9). For biomass and diversity, there are high diversity values associated with negative water PC1 scores representing a trend with high salinities. But the trend with positive water PC 1 scores is relatively flat.

Table 1.5. Principal Component vector scores for the FIBI metrics. Ranked by PC1

Metric	PC1	PC2	PC3	PC4	PC5
%Abundance > 3 cm	0.857	0.152	0.050	0.005	0.192
%Sp > 3	0.783	0.031	0.246	0.027	0.308
%Marine	0.497	0.499	0.141	0.455	-0.169
%Biomass > 3 cm	0.421	0.215	0.021	0.187	0.803
Biomass	0.122	0.266	0.149	0.862	0.195
Diversity (N1)	0.113	0.816	0.324	0.283	0.029
%Interface Feeders	0.069	0.041	0.957	0.165	0.004
%Predators	0.049	0.911	-0.033	-0.072	0.201
Abundance	0.011	-0.045	0.095	0.933	0.017
%Deposit Feeders	-0.088	-0.343	-0.892	-0.096	-0.037
%Freshwater	-0.108	-0.227	-0.090	-0.034	-0.020
%Brackish	-0.658	-0.444	0.357	-0.287	0.126

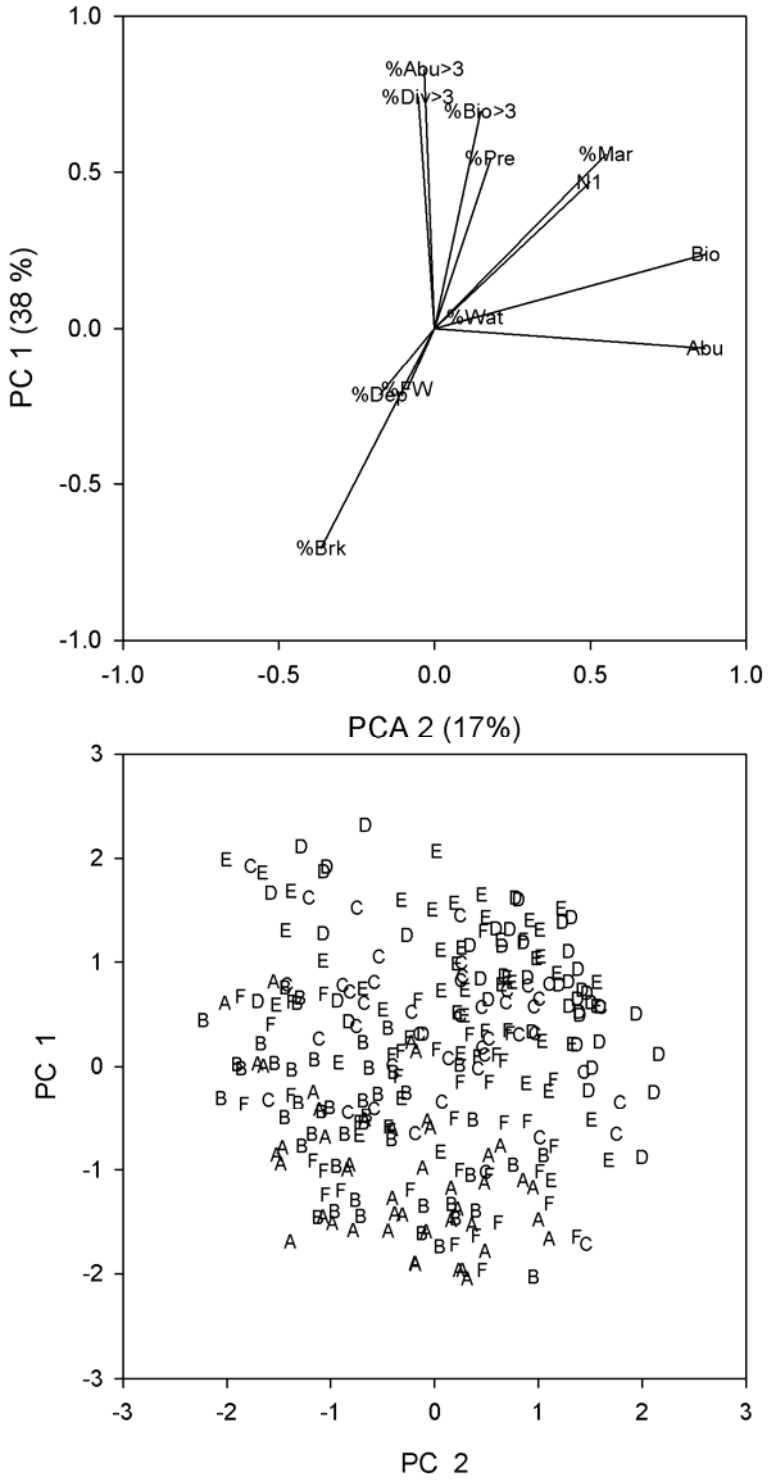

Figure 1.8. PCA variable loads (top) and station-date scores (bottom) for FIBI ranks. Abbreviations: %Brk = brackish indicator species; %Mar = marine indicator species; %FW = freshwater indicator species; %Wat = sediment interface (water column) feeders; %Dep = deposit feeders; %Pre = predator/ omnivore; Abu = overall abundance; Bio = overall biomass; N1 = diversity; %Bio>3 = biomass 3-10 cm in sediment column; %Abu>3 = abundance 3-10 cm in sediment column; %Div>3 = diversity 3-10 cm in sediment column.

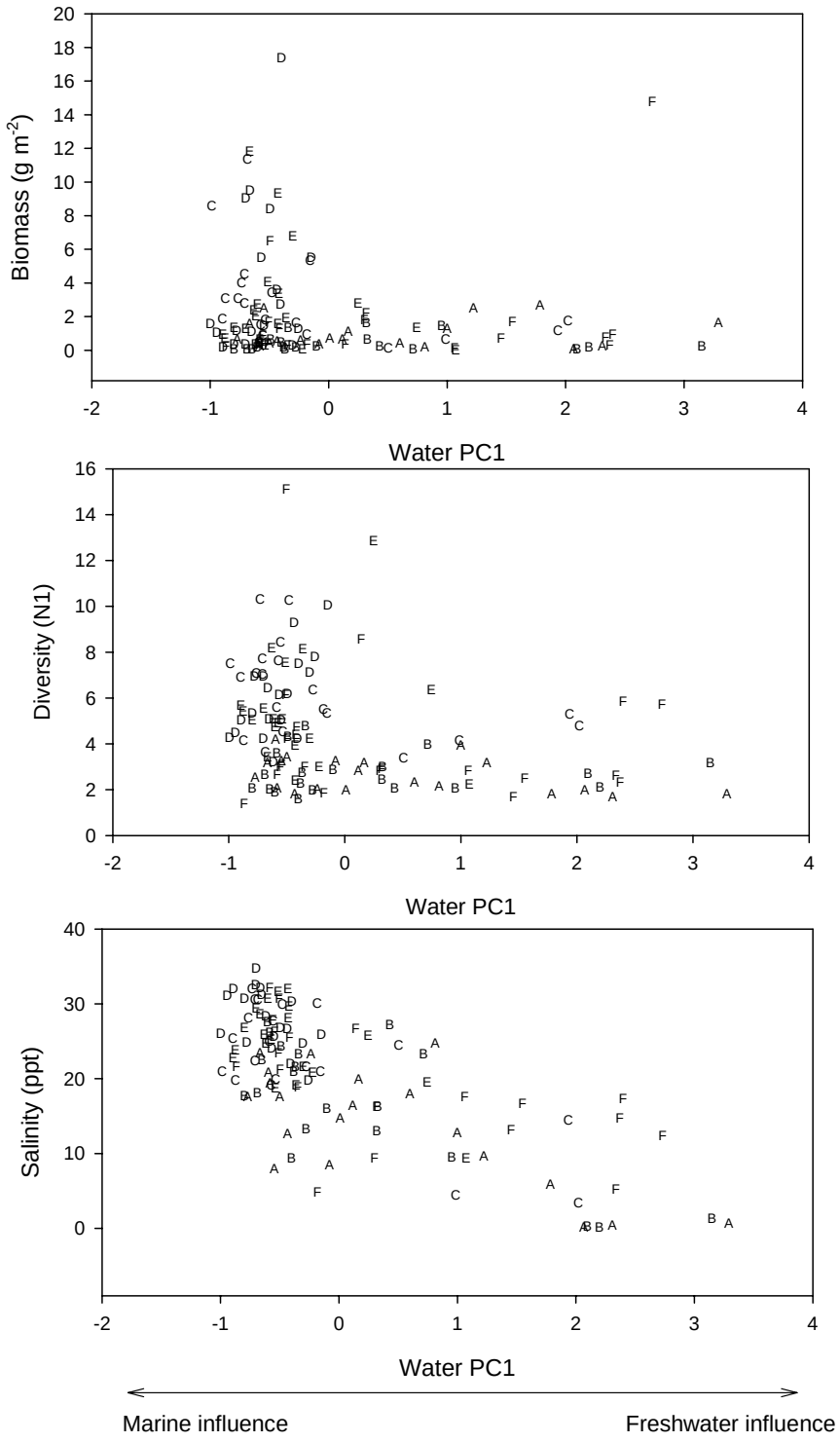

Figure 1.9. Relationship between PC1 scores for hydrographic measurements and biological responses.

DISCUSSION

No single variable can be used to completely characterize the effects that changes in salinity regime (e.g. freshwater inflow) have on a particular system. Changes in salinity gradients (e.g., inflow regimes) in estuaries result in multiple interactions between physical and biological factors, thus a multivariate method to assess which variables are most important is particularly important to predict how a system will change in response to alterations of hydrology. The FIBI was developed and employed to characterize how benthic populations are affected by changes along a salinity gradient in the Lavaca-Colorado estuary.

Field measurements illustrated the variability of benthic macrofaunal community structure and productivity along a salinity gradient. In general, the intermediate and marine-influenced stations were more productive than the freshwater-influenced stations, displaying higher values of abundance, biomass, and diversity. Similarly, Drake et al. (2002) reported that nekton and macroinvertebrate species richness, abundance, and biomass were all positively correlated with the salinity. Sediment composition, which has been found to significantly influence community structure in certain systems (Grebmeier et al. 1989, Mannino and Montagna 1997, Llansó et al. 2002), was not correlated with any of the biological data in this study. However, these findings do agree with earlier studies of the Lavaca-Colorado system and other Texas estuaries, which report that sediment grain size did not affect benthic community spatial distribution (Kalke and Montagna 1991; Montagna and Kalke 1992, 1995; Ritter et al. 2005). In contrast, the PC loads for the FIBI and the hydrographic variables were positively correlated along both the first and second PC axes, allowing for detection of changes in biological attributes due to changes along a salinity gradient (i.e., inflow regime) as well as location and season.

The PC 1 axis for both the FIBI and hydrographic variables represented inflow effects. In the FIBI PC1, freshwater and brackish water species had negative loads while marine species had high positive loading values. In the hydrographic PC 1, salinity had a high negative load while nutrients had high positive loads, representing the decrease in salinities and increase in nutrients (from runoff) associated with increased freshwater inflow. The second hydrographic axis (PC 2) represented seasonal effects by the direct opposition of temperature and dissolved oxygen, representing seasonal decreases in dissolved oxygen with increased summer temperatures (Applebaum and Montagna 2002). Thus, the correlation between FIBI PC 1 and hyrdrographic PC 1 demonstrates a strong link between inflow effects and community structure.

Increases in most of the FIBI index variables (i.e., metrics) were detected along the salinity gradient from fresh to marine, except for those responsive to freshwater inflow. The negative correlation between FIBI 1 and hydrographic PC 1 demonstrated that brackish indicator species, freshwater indicator species, and deposit feeders decreased along the increasing salinity gradient. High biomass, diversity, and marine indicators were the primary variables characterizing the most marine-influenced environment; other important variables included overall abundance as well as abundance and biomass of organisms residing in deeper substrates (from 3-10 cm in the substrate). Predators also exhibited an increase along the salinity gradient from fresh to marine. Deposit feeders were more associated with

freshwater-influenced environments, exhibiting a decrease in marine environments. This was most likely due to the sediment interface feeders in this trophic group that tend to colonize quickly after a disturbance (such as a large inflow event).

The dominance of the polychaete *Mediomastus ambiseta*, an equilibrium species, over all stations is interesting because it implies that the system is generally stable, or returns to stability relatively quickly post-disturbance. *Mediomastus ambiseta* is a subsurface deposit-feeding polychaete that can be found in both the first 3 cm of the substrate, as well as below in the 3 - 10 cm range (Kalke and Montagna 1991, Martin and Montagna 1995). Consistent with other studies, *Streblospio benedicti*, a pioneer species that can respond quickly to disturbances in the environment, was also dominant across all stations (Ritter et al. 2005, Kalke and Montagna 1991). *Streblospio benedicti* is a suspension-feeder generally found in the first 3 cm of the substrate. Both *M. ambiseta* and *S. benedicti* have wide tolerance ranges for changes in salinity such that when other species are stressed or killed by hypersaline conditions, both can survive and dominate. *Streblospio benedicti* can proliferate rapidly after significant disturbances (Ritter et al. 2005), while *M. ambiseta* generally dominates after conditions stabilize. The dominance of both species suggests a cycle of disturbance and stability for this estuary.

The scoring system for the FIBI reflects the inflow effects across the estuary. The lowest PC 1 factor scores were associated with more freshwater influenced portions of the system. Values increased along the salinity gradient from fresh to marine, with the highest values representing more marine-influenced environments. Whereas many indices of pollution or other environmental stressors utilize a system of classification based on threshold values that define specific environments of interest (e.g., degraded vs. pristine environments), for the FIBI, PC loading scores were left continuous and not parsed into distinct categories, allowing for greater sensitivity in detecting small shifts in environmental condition. Further, it was possible to detect trends associated with both season and salinity regime (i.e., inflow events) by comparing the biological PC scores to the corresponding hydrological scores on each axis. This is important because a shift in community structure during a given sampling period might be primarily due to seasonal differences (i.e., temperature and irradiance) rather than inflow effects, or vice versa. These distinctions are necessary to determine causal relationships and appropriately apply the findings to inflow management strategies.

The FIBI can also be used as a quantitative assessment tool. For example, there appear to be three zones in the Lavaca-Colorado estuary separated by about 1.00 to 1.25 PC 1 score units. The freshwater and brackish zone, primarily composed of freshwater-influenced stations A, B and F, ranged from PC 1 values of -1.75 to -0.25. The intermediate zone, composed of station C, ranged from -0.25 to about 0.75, and the marine zone characterized by station D near the Gulf of Mexico pass ranged from 0.75 to 2.0. These distinct value ranges indicate that approximately a 30% change in PC loading score is sufficient to cause a shift in community structure and function.

The results presented here indicate that the FIBI can be effectively used to detect changes in benthic biological measures due to changes along a salinity gradient in the Lavaca-Colorado estuary. The overall methods for such an index should remain relatively similar in other

systems, although some metrics may need to be adjusted. For example, in other systems, the use of different indicator species or the addition of a sediment substrate metric may be useful. If found to be an effective general model, the FIBI could be paired with salinity or inflow models in order to quantitatively predict changes in benthic secondary production (i.e., biomass). This pairing may also allow for the development of models that could predict quantitative trophic changes throughout estuarine food webs. Such models would advance our understanding of how freshwater inflow affects trophic dynamics (which are strong indicators of ecosystem function and health) in these environments.

The maintenance of environmental health or integrity is a growing priority in environmental legislation and regulation for local, national, and international communities. In the past, the terms "environmental integrity" or "environmental health" have been vague and difficult to interpret. However, new definitions and guidelines for interpretation of these terms have emerged over the last two decades and have helped scientists and environmental governing agencies more accurately assess and create management strategies for various ecosystems. "Health" has been defined as a vigorous, resilient to disturbance, and organized ecosystem that is maintaining or supporting a biologically diverse community (Costanza et al. 1992; Mageau et al. 1995). Biological "integrity" is essentially the capacity or capability of the system to sustain its health, which can be understood by characterizing the functional and structural aspects of ecosystems over time (Cairns 1977; Karr and Dudley 1981). The FIBI, which contains components of both structural integrity (e.g., abundance, biomass, and diversity) and functional integrity (e.g., trophic guilds and vertical habitat presence), was designed to be a tool to help simplify the process of determining ecosystem health and detecting important changes that occur within these environmentally-sensitive systems. The index was used successfully to detect changes in benthic biological measures as a function of salinity regime (as a proxy for freshwater inflow) in the Lavaca-Colorado estuary. More research and application of the FIBI to other systems is needed to determine the generality and effectiveness of this new approach.

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Chapter 2

Colorado River Flow Relationships To Bay Health: Modeling Benthic Productivity

By:

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The Lavaca-Colorado Estuary is a major estuarine system along the Texas coast that provides major economic benefit to the region by supporting a variety of agricultural, residential, industrial, and recreational functions. The Matagorda Bay Health Evaluation (MBHE) component of the LCRA/SAWS Water Project (LSWP) Study Plan was created to assess the environmental effects that could result from further changes to inflow patterns in the Matagorda Bay system. To support this assessment, a bio-energetic model, calibrated using a long-term data set of benthic biomass, was run to relate macrobenthic biomass to salinity within the estuary. This model was applied to the current study to assess the role of freshwater inflow in controlling benthic productivity. Benthic productivity was calculated for two groups of macrobenthos, suspension feeders and deposit feeders; in two bays, Lavaca Bay and Matagorda Bay. Simulations of the Lavaca-Colorado Estuary, based on a calibration of data from 1988 - 2005, fit the observed data relatively well. However, following the year 2000, simulations predicted a much higher benthic biomass than in observed data. The increase in benthic biomass is likely explained by the decrease in predator populations, particularly blue crabs, which reduced loss to predation. Simulations of deposit and suspension feeder biomass exhibited responses due to natural and simulated salinity changes in both bay systems. As salinity increased, deposit feeding biomass increased while suspension feeding biomass decreased. Total biomass in Lavaca Bay increases very slightly as salinity increased, which indicates that reduced inflow rates in this bay would not harm benthic community productivity. Total biomass concentration in Matagorda Bay decreased with increasing salinity. Thus, reducing the freshwater inflow may cause the upper river communities to take on downstream community appearance. This effect is probably due to the benthic community acclimating to the elevated salinity or more salt tolerant species populating the area. It is concluded that freshwater inflow plays an important role in maintaining the observed character of estuarine productivity through the combined effects of the frequency, duration, timing, and magnitude of inflow, particularly during droughts or low-flow periods.

INTRODUCTION

Background

The Lavaca-Colorado Estuary is a major estuarine system along the Texas coast that provides major economic benefit to the region by supporting a variety of agricultural, residential, industrial, and recreational functions. As a result of these human endeavors, the estuary has undergone dramatic changes in recent history (even resulting in a name change). Alterations in hydrology, circulation, and freshwater inflows have occurred due to anthropogenic activities, including human modifications of inflow to several major tributaries that supply the estuary, particularly the Lavaca River and the Colorado River. Prior to 1990, the name of the estuary was the Lavaca-Tres Palacios Estuary because the main supply of freshwater inflow came from the Lavaca and Tres Palacios rivers. After a diversion channel was completed in 1990 to redirect water from the Colorado River into Matagorda Bay, the Colorado River surpassed the Tres Palacios River as a major supply of freshwater to the system, and the name changed accordingly to the Lavaca-Colorado Estuary. This is just one example of the magnitude of change that has occurred with respect to inflow since 1990.

Another change occurred in 1991 when the Colorado River was diverted into the eastern arm of Matagorda Bay via a flood diversion channel. In 1992, a dam was built in the river channel below the point of diversion. This project diverted Colorado River water that might have flowed into the Gulf of Mexico into the eastern arm of Matagorda Bay.
The lower Colorado River basin supports a diverse ecological community that relies heavily on the quality and quantity of water moving through the system. The wide range of ecosystem components and ecological conditions associated with communities in the lower Colorado River and Matagorda Bay means that understanding its processes is quite challenging. The LCRA-SAWS Water Project (LSWP) has the potential to alter the flow regime for the lower Colorado River and, consequently, Matagorda Bay. The Matagorda Bay Health Evaluation (MBHE) has been established to assess the potential impact of these flow regime modifications. The present study is an integral part of the MBHE's objective to assess potential impacts/benefits on the aquatic resources of Matagorda Bay with and without the project and also quantify the condition of the aquatic environment under different flow scenarios to satisfy federal and state permitting requirements and ensure that the environmental principles set forth for this project satisfied.

Benthic Studies

Historical studies have stressed the importance of freshwater inflow to estuarine system, and that inflow is a major factor driving estuary functioning and health (Chapman 1966, Kalke 1981). Inflows serve a variety of important functions in estuaries, including the creation and preservation of low-salinity nurseries, sediment and nutrient transport, allochthonous (outside) organic inputs, and movement and timing of critical estuarine species (Longley 1994). Benthic macrofauna (> 0.5 mm) are especially sensitive to changes in inflow, and can be useful in determining its effects on estuarine systems over time (Kalke and Montagna 1989).

Benthic macroinvertebrates are established indicators of water quality in both freshwater and marine systems, and can highlight different aspects of the environment, including pollutant levels, hypoxia/anoxia, turbidity, and salinity changes (Oglesby 1967, Merritt and Cummins 1984). Relatively sessile and long-lived, benthic macroinvertebrates can reveal temporal changes in the environment that simple hydrographic measurements and chemical analysis can either not determine, or are impractical to use because of sizeable monetary and time constraints. Ubiquitous and relatively inexpensive to collect and analyze, bnethic macrofauna are excellent tools in assessing both short- and long-term environmental conditions (Montagna and Kalke1992, 1995).

While many early studies in Texas estuaries focused on oyster reefs, recent benthic work has concentrated on the soft-bottom dwelling macrofauna and their relationship to several key environmental variables, including freshwater inflow, salinity, dissolved oxygen, depth, and sediment type (Kalke and Montagna 1991, Engle and Summers 2000). While grain size and type, as well as temperature, play significant roles in benthic macroinvertebrate community structure, freshwater inflow and corresponding changes in salinity are the primary factors controlling the distribution of marine and freshwater organisms within an estuary (Mannino and Montagna, 1994, Kalke and Montagna 1989, 1991, 1992, Attrill et al. 1996, Montagna et al. 2002). While sediment type and depth are important to benthic organisms, they are relatively minor factors controlling benthic community structure in most Gulf of Mexico estuaries when compared with salinity (Engle and Summers 2000). Depth, in particular, is rarely a key factor because most estuaries in Texas are relatively shallow (Baird et al. 1996, Engle and Summers 2000).

Benthic fauna are critical intermediaries in estuarine food webs and affect substrate structure, sediment, and water chemistry as well. Thus, a change in their community composition has the potential to affect many other biological and physical processes in estuarine habitats. Because freshwater inflow and salinity play such major roles in benthic community structure, it is important to understand the effects of inflow and other external factors to evaluate (and prescribe management options for) estuarine ecosystem health.

Using models to predict bay health is dependent upon knowing what is meant by “health.” Ecological health can be defined as the determination that indicators of specific ecological conditions are in an acceptable range. Indicators are metrics for which sufficient information exists on the acceptable range of responses across broad spatial and temporal scales. For the current project, the metric being examined is benthic productivity responses to changes in freshwater inflow.

Ecosystem Modeling

Ecosystem models are representations of underlying mechanistic relationships among ecological components and processes. Ideally, they reduce ambiguity and describe complexity with maximum parsimony. Models in ecology are useful because of the inherent complexity of ecological relationships, the characteristic variability in ecological systems, and the apparently unpredictable effects of deliberate modification of systems by man. It is difficult to understand benthic dynamics from empirical or static modeling analyses alone

(e.g., multivariate statistical methods). The concept that benthos are an isolated subsystem, governed by internal interactions and "key species" is not sufficient to explain the heterogeneity of benthos community dynamics in closely related sites. However, a model can incorporate spatial variability to provide insights into the dynamics and interactions of benthic populations within an ecosystem, or to predict long-term effects of those interactions.

Modeling of an ecosystem can start from a qualitative conceptual model. The conceptual model is largely theoretical and heuristic. The purposes for modeling ecosystems can range from developing simple conceptual models to provide a general understanding of system behavior, to detailed realistic applications aimed at evaluating specific policy proposals. It is not possible to judge this whole range of models by the same criteria. At least three criteria are necessary: realism, precision, and generality. Unfortunately, no single model can maximize all three. The conceptual model has high generality for Texas estuarine systems, but low realism and low precision. A quantitative model, however, can provide the realism and precision, and test hypotheses drawn from the conceptual model.

A quantitative model requires a long-term data set to calibrate the model. In addition, independent data sets are also needed for model corroboration or validation before these models are used for extensive predictions. Long-term macrobenthos data for the Lavaca-Colorado Estuary was available, which made it possible to perform a modeling experiment on this system. A bioenergetic model was developed within and among four Texas estuaries (Lavaca-Colorado, Guadalupe, Nueces and Laguna Madre), which related macrobenthic productivity to salinity (Montagna and Li 1996). A five year data set from 1990-1995 of macrobenthic biomass was used to calibrate the model. The benthos were divided into two trophic groups: deposit feeders (that consume detritus or sediment organic matter) and suspension feeders (that filter phytoplankton or graze on benthic diatoms). Simulations for the eight Texas bays did fit the data well, indicating that the structure of Texas estuaries is strongly influenced by inflow and Gulf exchange. Within estuaries, the production to biomass ratio (P/B), with units of 1/year, increased with proximity to the freshwater inflow source. The P/B ratio for deposit feeders generally increased with water residence time, i.e., inflow volume adjusted by the estuary volume, but declined with water residence time for suspension feeders. This trend is consistent with the hypothesis that suspension feeders are good indicators of the importance of freshwater inflow on maintaining secondary production. Thus, regression coefficients determined from the earlier eight-bay study were used to compare the long-term data on benthic productivity within the Lavaca-Colorado Estuary.

Study Focus

Long-term macrobenthos data for the Lavaca-Colorado Estuary was used to perform a modeling experiment to determine the effects of alterations in freshwater inflow. The estuary is characterized by a primary and secondary bay, which experience a salinity gradient from sea to river because the primary bay is tidally influenced by connection to the Gulf of Mexico, and the secondary bay is connected to a freshwater source. Comparison of these bay types allows for an examination of effects over a large range salinity, which acts as a surrogate of freshwater inflow effects.

The goal of the present study was to characterize the salinity-benthos and salinity-ecological relationships within Lavaca Bay and Matagorda Bay to provide a means of assessing biological impacts or benefits of altering various flow regimes. Data generated from past studies was combined with recently collected field data to simulate dynamics of benthos with an existing bioenergetics model (Montagna and Li 1996) and to compare productivity over space and time. The model provides prediction capabilities necessary to evaluate the full range of potential flows from low to moderate to high on two ecological components of the lower Colorado River system throughout the annual hydrologic cycle.

METHODS

Study Sites

Six stations were chosen in the Lavaca-Colorado Estuary (Fig. 2.1 and Table 2.1). Two replicate stations (A and B) are in the secondary bay, where freshwater influences are greatest, and two other replicate stations (C and D) are in the primary bay, where marine influences are greatest. Using two stations in the freshwater-influenced zone and two stations in the marine-influenced zone replicates the effects at the treatment level and helps to avoid pseudoreplication. A diversion of the Lower Colorado River into the east arm of Matagorda Bay added two additional stations (E and F) in that lagoon. The data from Stations E and F were combined with data from Stations C and D to characterize the primary bay. Data from Stations A and B were pooled to characterize the secondary bay.

Figure 2.1. Study area and station locations.

Table 2.1. Sampling stations and sampling periods in the Lavaca-Colorado Estuary.

Stations	Bay Name	Bay Type	Sampling Period
A	Lavaca	Secondary	1984 - 2007
B	Lavaca	Secondary	1988 - 2007
C	Matagorda	Primary	1988 - 2007
D	Matagorda	Primary	1988 - 2007
E	East Matagorda	Lagoon	1993 - 1995 2004 - 2007
F	East Matagorda	Lagoon	1993 - 1995 2004 - 2007

Modeling Procedure

The model consisted of several mathematical equations that calculate the variation of benthos biomass in response to the variation in environmental data. The model's input was the observed long-term environmental data, and output was simulated benthos biomass over time. When the data input and benthos model input is fixed, simulation of observations can be improved by changing the mathematical equations. To model an ecosystem, several repeated cycles of structure, calibration, and simulation are required until the simulation of observations is satisfactory (Fig. 2.2). Each time, sensitivity analysis is performed before the calibration to determine if the output range of the simulation will cover the range of all observations. A parameter should be sensitive enough to change the model output, if not then the parameter can be deleted. In the present study, the simulation of the observations is based on newly calibrated parameters.

For the current study, a database was assembled that includes data for salinity, benthos biomass, and other environmental variables. The data may be used as either observations for model calibration, input to the model, or as a forcing function to drive the model. Statistical analyses can be used to determine significant environmental factors or to simplify the model by reducing the unimportant variables in the model. The data collection is also used to set up initial parameter ranges. It is more efficient when the initial parameter ranges are as narrow as possible, because it reduces calibration effort and limits the possibility of a wrong calibration direction. When the ranges are unknown, the range of parameter values must be as large as possible to include all possibilities. Previous modeling studies are the best source of information for developing the model structure, calibration, and validation techniques and the present study referred to Montagna and Li (1996) for the initial range of parameter values.

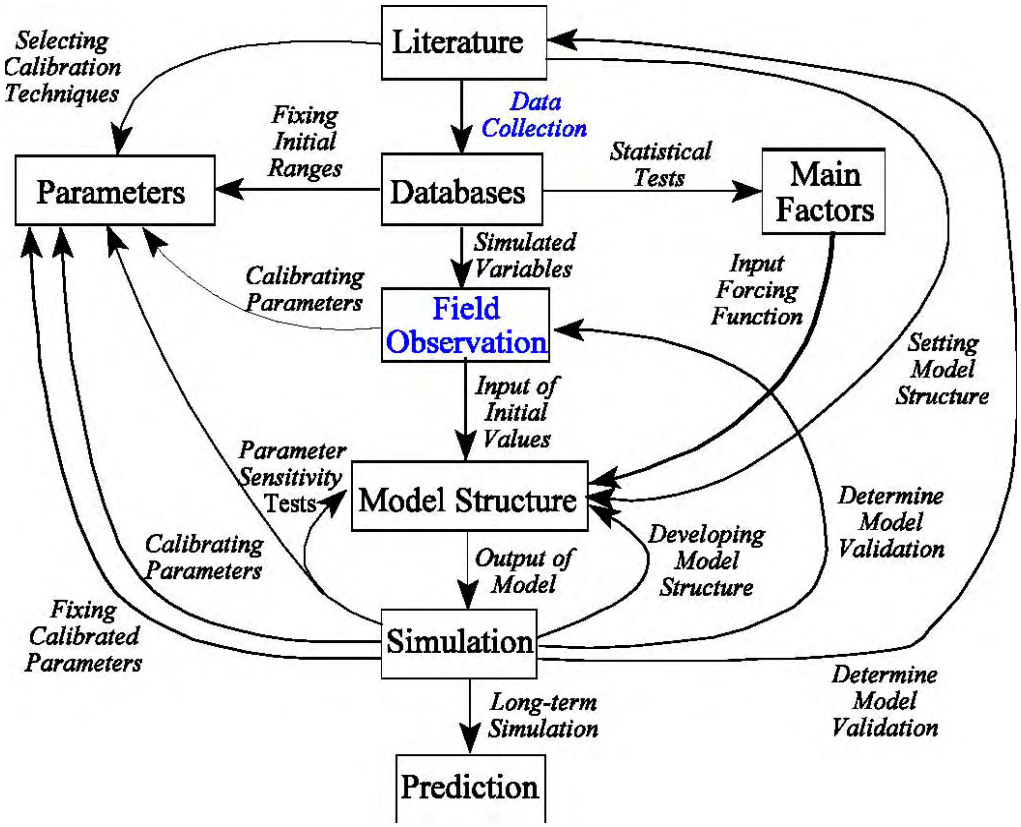

Figure 2.2. Flow diagram of the steps used to develop the model.

Databases

The availability of a database is the most important component for a modeling study, as it can determine the success of a modeling project. A good database provides a high number of observations, information from the literature, and low error rates due to high sample numbers. A good database can also provide independent data that can be used for model validation, a very crucial step in the modeling process. These characteristics of the database ensure the correct determination of parameters.

Benthic Macrofauna

Sampling of the Lavaca-Colorado Estuary began in November 1984 (Table 2.1). From previous studies, it was learned that long-term changes in benthos within these estuaries could be characterized by sampling on a quarterly basis (Kalke and Montagna 1991). Starting in July 1988, a sampling program to compare the Lavaca-Colorado and Guadalupe Estuaries was initiated, which was funded by the Texas Water Development Board (TWDB). Since then, the program has been expanded to include the eastern arm of Matagorda Bay with funding from the Lower Colorado River Authority (LCRA). The goal for establishing these stations was to assess the effect of the Colorado River diversion on estuarine productivity. Funding for various other projects by the Coastal Bend Bays Foundation, Texas A&M Sea

Grant Program, Corpus Christi Bay National Estuary Program, and the Texas Advancement Technology Program has contributed to the development of a long-term, coast-wide database on benthic biomass, abundance, and community structure.

During each sampling event, hydrographic measurements were also made, which included chlorophyll a nutrient concentrations, salinity, temperature, and water depth. Once each year (usually in October), sediment grain size, total nitrogen, and organic carbon content were also measured in sediments.

Predators

Fisheries data from 1988-present were obtained from Texas Park and Wildlife Department (TPWD) (Dailey et al. 1991). The Coastal Fisheries Division samples monthly in the Lavaca-Colorado Estuary using a shrimp trawl and bag seine. In a study of mercury bioaccumulation in different food chains, Montagna (unpublished) determined that black drum, red drum, and blue crab are the main predators on benthic infauna. Therefore, the average value for density of each of these three main predators was used.

Other Environmental Data

The salinity, temperature, and water depth data used in this study were also recorded by TPWD. Nutrient data for many of the same stations and periods are available (Whitledge 1989). There is not enough primary production data to form a time series, however, a range of values for primary production from previous studies is available (Table 2.2). Monthly day-length for the Texas coastal area was obtained from Tony Amos at UT Marine Science Institute. Table 2.3 summarizes the data available in the continuous long-term database assembled in order to model the Lavaca-Colorado Estuary.

Table 2.2. Available data for primary production from previous studies.

Bay	Previous record of primary production (g C m^{-2} d^{-1})	References
Lavaca	0.5 - 2.4	Brock (1994)
Matagorda	0.5 - 2.4	Brock (1994)

Table 2.3. Summary of data variables assembled for the continuous long-term database for modeling the Lavaca-Colorado Estuary. Period is given as the year and month for each estuary.

Variable	Dates (year/month)
Temperature	1988/04 - 2005/10
Salinity	1988/01 - 2005/10
Water Depth	1988/04 - 2005/10
Nutrients (N, P, Si)	1991/10 - 2005/10
Predator Density	1987/01 - 2005/7
Benthos Biomass	1988/04 - 2006/10

Model Structure

The long-term, benthic macrofaunal data set from the Lavaca-Colorado Estuary was used to calibrate the temporally dynamic model of biological processes. The two principle environmental factors associated with freshwater inflow are salinity and nutrient concentrations; therefore, the relationship between biomass of benthic macrofauna and these environmental factors was incorporated into the model. To test for inflow effects, the ideal input to the model would be freshwater inflow as the basic forcing function. However, inflow rates have variable effects depending on the hydrological and physiographic characteristics of each estuary. Therefore, a physical model that predicts salinity change under varying inflow scenarios would be needed to provide input to the biological model. To avoid this level of complexity, the empirical salinity values were used as input; thus, salinity was used as a surrogate for inflow. Salinity values represent the integration of all the physical characteristic of the estuary, e.g., size, inflow, outflow, tidal exchange, and climatic variability. Other inputs to the model included fish and crabs as predators, temperature, water depth, day length, and nutrient concentrations.

Odum (1971 and 1983) energy circuit language is used to present the model for simulating the Lavaca-Colorado benthic biomass (Fig.2.3). The current model includes four forcing functions: salinity, temperature, food sources, and predators. They drive the model mainly through four environmental limitations: salinity, temperature, food availability, and predation. Other forcing functions (e.g., nutrient concentrations, day length, and water depth) drive the model through the estimation of food source availability by the calculation of primary production.

There are two main trophic guilds in benthic sediments: the grazing food-chain and the detrital food chain. Grazers utilize autotrophic production and detritivores utilize heterotrophic production. To simplify the model, all macrobenthic animals were separated into one of two groups: the suspension feeders and deposit feeders. Suspension feeders are

defined as those who obtain their food sources through capturing suspended particles from the sediment surface or water column, filtering phytoplankton from the water column, or grazing benthic diatoms on the sediment surface. Suspension feeding taxa include the Mollusca, Crustacea, and Chironomid larvae. Deposit feeders are defined as those organisms that obtain their food through ingestion of the sediment, predation, or omnivory. The deposit feeders include the Hemicordata, Nemertinea, Ophiuroidea, Polychaeta, and Sipunculida. Many benthic organisms, e.g., mollusks and polychaetes, can alternate between being suspension feeding and deposit feeding. This simplification allows suspension feeders to be defined as organisms limited by autotrophic food sources, and deposit feeders as organisms limited by heterotrophic food sources.

Modeling benthic secondary production is not as simple as modeling primary production. The benthic food web is complex, and secondary production rates are not a function of physical-chemical variables. Primary producers, whose growth is based on irradiance and nutrient concentrations, are the main food source for suspension feeders. Therefore, it was necessary to predict the food sources for suspension feeders in the study model. Deposit feeders primarily consume particulate organic matter (POM), and this can be approximated by the concentration of total organic carbon (TOC) in sediments, which is empirically derived. The accumulation of POM, as well as the variation of nutrient concentrations and salinity due to temporal variations in freshwater inflows, were not simulated in this model. Instead, the measured concentration of nutrients and salinity were used as inputs. The mathematical formulae were based on known bioenergetic mechanisms of invertebrates.

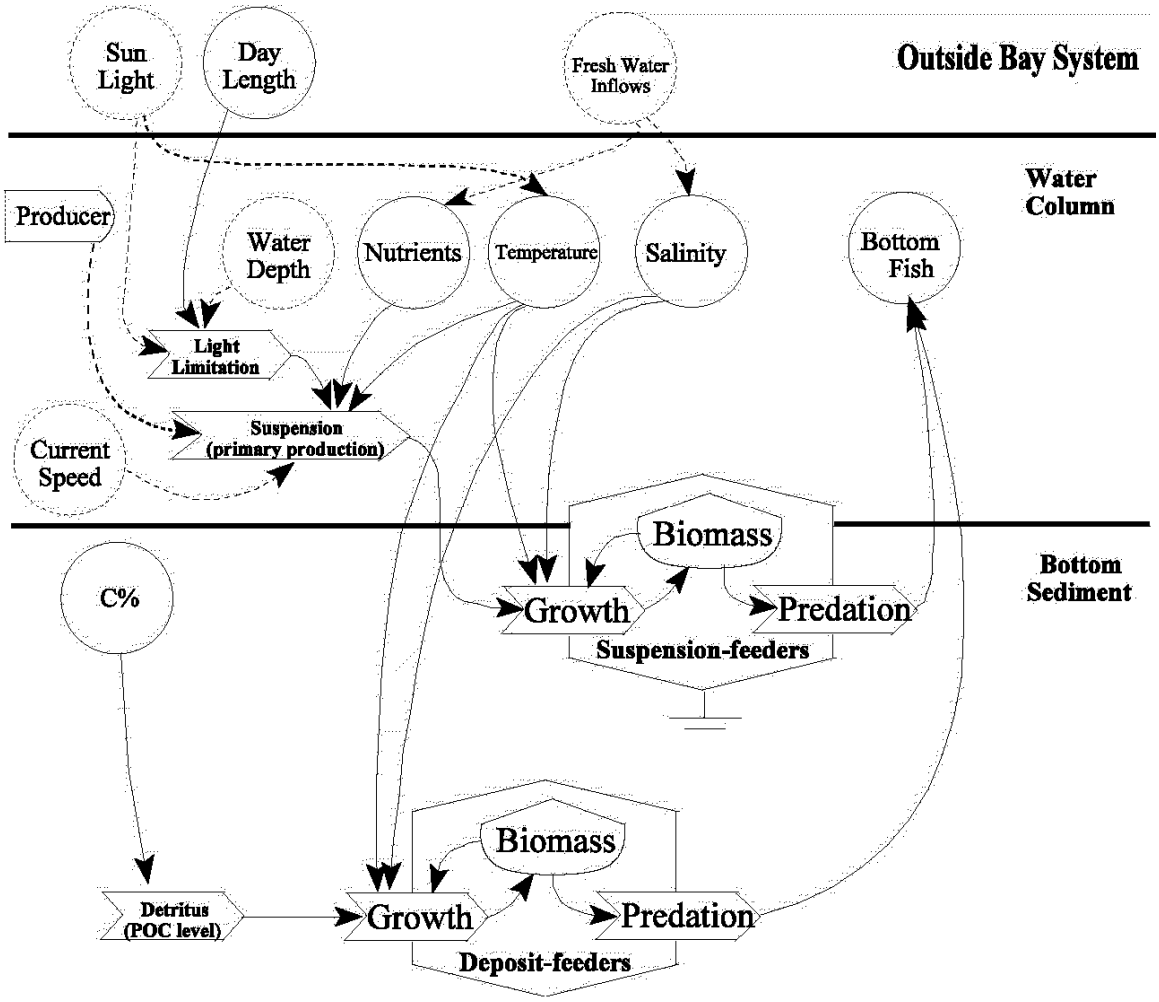

Figure 2.3. Energy circuit diagram for the structure of the benthic macrofauna biomass model. Dashed lines represent variables not included in the study model.

Development of Mathematical Formulae

The basic formula that describes the change of benthic biomass over time (Li et al. 1996) is based on the law of conservation of mass, and in the form:

$$\frac{d(B)}{d(t)} = I \cdot A - L - D \tag{1}$$

where B is the benthic biomass, t is time, I is the total intake of food by benthic infauna, A is average assimilation efficiency of benthic infauna, L is the total loss due to respiration, excretion and age-related mortality, D is total mortality caused by predators. Unfortunately, it is not possible to simulate B in terms of I and L, because available field data on food source standing stocks and respiration of benthic organisms are rare and incomplete in the study area. The approach used in this study replaces net growth rate in place of I and L.

Growth Rate of Benthic Biomass

The net growth rate is used in place of the intake rate, assimilation efficiency, respiration rate, aging mortality and excretion rate. The formula becomes a Lotka-Volterra growth rate model (Lotka 1925) in the form:

$$\frac{d(B)}{d(t)} = r \cdot B - g \cdot F \tag{2}$$

where r is the net growth rate without predation pressure. The predation loss is calculated by feeding rate of predators, g, and the density of predatory fish, F.

A logistic limitation term to growth rate is suggested by Brown and Rothery (1993), and takes the form:

$$\frac{d(B)}{d(t)} = r \cdot B \cdot \left(1 - \frac{B}{c}\right) - g \cdot F \tag{3}$$

where c is the biomass carrying capacity for a population that is limited by space. The c in equation (3) is only a limitation for the capacity of biomass. The limitation of a population and its biomass is also due to many other environmental effects. The study model is based on equation (3), and has been modified to include environmental limitation. The new equation contains a parameter to reduce the maximal growth rate (r) and maximal predation rate (g) by the effects of environmental limitation (E). The values of E are between 0 and 1. When E = 1, there is no environmental limitation, and the benthic population reaches maximal growth rate or the predators reach maximal feeding rate. When E = 0, environmental factors reach maximal limitation, benthic populations do not grow, or predators do not consume benthos. As there is more than one predator, the final equation for the model becomes:

$$\frac{d(B_{(i,j)})}{d(t)} = \frac{r_{(i)}}{12} \cdot E_{ben(i,j)} \cdot B_{(i,j)} \cdot \left(1 - \frac{B_{(i,j)}}{c_{(i)}}\right) - 30 \cdot E_{fish(i,j)} \cdot \sum_{k} g_{(i,j,k)} \cdot F_{(j,k)} \tag{4}$$

where i = 1 or 2 for deposit feeders or suspension feeders; j = 1 - 2 for the two bay systems; k = 1 - 3 for three different predators: red drum, black drum and blue crab. The annual net growth rate is $r_{(i)}$, $E_{ben(i,j)}$ is the environmental limitation for benthic biomass growth, $c_{(i)}$ is the biomass carrying capacity levels for the two feeding groups, $g_{(i,j,k)}$ is the predation rate by fish k in bay j to prey benthos i, and $E_{fish(i,j)}$ is the environmental limitation for predation. There are two constants in equation 4. The parameter $r_{(i)}$ and is divided by 12 to convert annual growth to a monthly rate. The parameter $E_{fish(i,j)}$ is multiplied by 30 to convert daily densities to a monthly rate. The different benthic species have different biomasses in each bay, computed by their r, E, c and g. Predator abundances are also different in both bays. Benthic organisms should have the same r and c in each bay, however the dominant species in both the deposit-feeding group and suspension feeding group were different in each bay. For this research it was necessary to run the model separately for the different bay systems.

The term $E_{ben(i,j)}$ includes three effects: temperature limitation ($E_{tem(j)}$), salinity limitation ($E_{sal(i,j)}$), and food concentration limitation ($E_{food(i,j)}$):

$$E_{ben(i,j)} = E_{tem(j)} \cdot E_{sal(i,j)} \cdot E_{food(i,j)} \tag{5}$$

Temperature Limitation

An exponential equation was used for the temperature effect (Carrada, 1983):

$$E_{tem(j)} = \frac{1}{e^{\frac{|T_{(j)} - T_{\max}|}{p_{(1)}}}} \quad (6)$$

where $E_{tem(j)}$ is the temperature limitation, $T(j)$ is the temperature, and $Tmax$ is the most suitable temperature, which is fixed at the highest temperature recorded at each location. When T is close to $p_{(1)}$, $E_{tem(j)} = 1$, and there is no temperature limitation. Therefore, $p_{(1)}$ is a parameter that describes the weighing due to temperature limitation. The higher $p_{(1)}$ is, the higher the sensitivity to temperature (Fig. 2.4).

Salinity Limitation

Salinity is one of the most influential environmental variables affecting benthic communities and is directly correlated with freshwater inflow. All invertebrates have optimal salinity ranges at which population growth is maximal, i.e., the highest metabolism rates (Wohlschlag et al. 1977). Because of these salinity effects, an exponential equation is used to model salinity limitation. The equation is similar in form to that used for temperature limitation:

$$E_{sal(i,j)} = \frac{1}{e^{\frac{|S_{(j)} - p_{(i,3)}|}{p_{(2)}}}} \quad (7)$$

where $E_{sal(i,j)}$ is the salinity limitation, $S_{(J)}$ is salinity, $p_{(i,3)}$ is the optimal salinity for a population, and $p_{(2)}$ is a parameter that describes the weight of the salinity limitation. There is no salinity effect when $p_{(2)} = \infty$. Salinity limitation has a centralized optimum, with greater effects at high and low salinities (Fig. 2.5). The greater the salinity tolerance range, the higher the $p(2)$ value (Fig. 2.5).

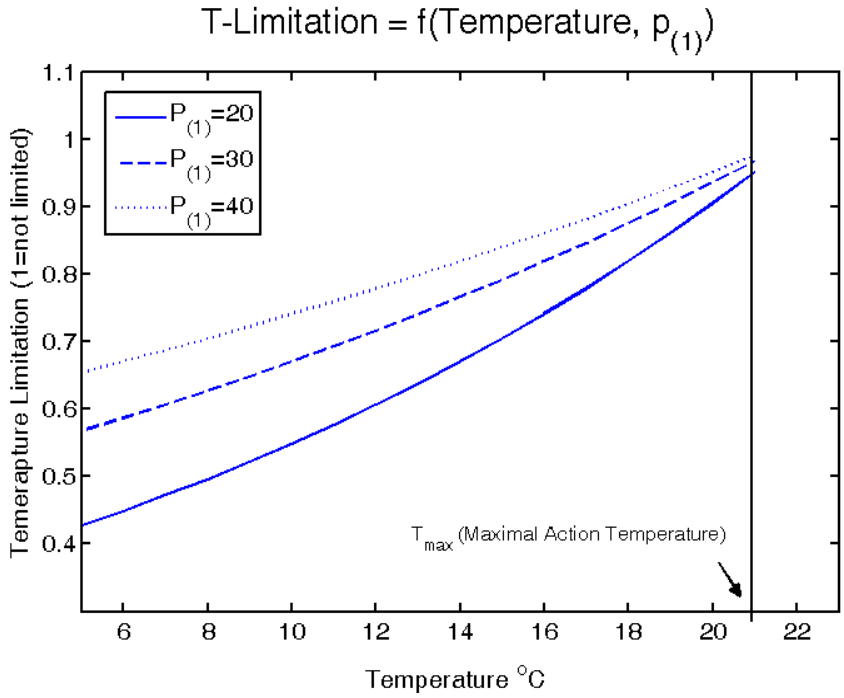

Figure 2.4. Effect of the weight of the parameter ($p_{(1)}$) on temperature limitation calculated via equation (6).

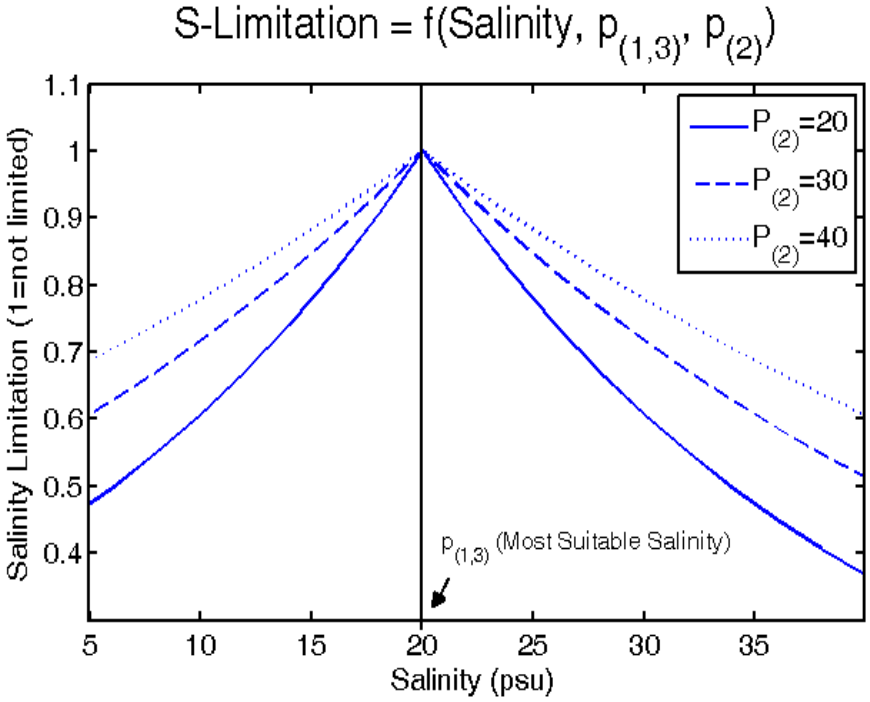

Figure 2.5. Effect of the weight of the parameter ($p_{(2)}$) on the salinity limitation calculated via equation (7).

Food Source Limitation

Michaelis-Menten kinetics is used to describe the food source limitation (Keen and Spain 1992):

$$E_{food(i,j)} = \frac{F_{(i,j)}}{F_{(i,j)} + p_{(i,4)}} \tag{8}$$

where $E_{food(i,j)}$ is the food limitation, $F_{(i,j)}$ is the concentration of the food source for benthic organisms (sedimentary POC for deposit feeders and primary production for suspension feeders), and $p_{(i,4)}$ is a parameter at which the food concentration is at half the maximum level of the population growth rate.

As two feeding groups were simulated in the model (deposit feeders and suspension feeders), there were two different food sources: detritus in sediment and organic matter in the water column. Sedimentary POM was used as a food source for deposit feeders, and expected primary production was used for suspension feeders. Increased consumer biomass ($B(i,j)$) can increase food limitation, therefore, equation (8) transforms the ratio as a function of benthic biomass:

$$E_{food(i,j)} = \frac{\frac{F_{(i,j)}}{B_{(i,j)}}}{\frac{F_{(i,j)}}{B_{(i,j)}} + p_{(i,4)}} \tag{9}$$

Food Sources for Deposit Feeders

The sedimentary POM is expected to remain constant in each bay. Two parameters were used as POM levels, one specific to each bay. The POM levels were pre-calibrated by the observed carbon concentration ($C\%_{(j)}$) in the sediment (j = 1 - 2 for two bays):

$$C\%_{(j)} = \frac{p_{pom(j)}}{p_{sed}} \cdot 100\% \tag{10}$$

Where $P_{pom(j)}$ is the sedimentary POM level for each bay, and P_{sed} is a parameter for the average dry weight of the whole sediment. The POM levels for each bay represent the food sources for deposit-feeders ($F_{(i,j)}$) in each bay:

$$F_{(1,j)} = p_{pom(j)} \tag{11}$$

Food Sources for Suspension Feeders

Primary production is expected to be the most important food source for suspension feeders. Primary production is simulated as a function of day length, temperature, nutrient concentration and water depth. Primary production was pre-calibrated using data from previous studies (Stockwell 1989, Armstrong 1985) using the following formula:

$$F_{(2,j)} = D_{(j)} \cdot p_{mic(2)} \cdot \frac{1}{e^{\frac{|T_{(j)} - T_{\max}|}{p_{mic(3)}}}} \cdot L_{(t)} \cdot E_{nut(j)} \tag{12}$$

Where $F_{(2,j)}$ is the available food for suspension feeders, $D_{(j)}$ is the water depth adjustment described in equation (13), $P_{mic(2)}$ is the maximal monthly primary production rate, $P_{mic(3)}$ is the temperature limitation for primary production, $L_{(t)}$ is the day length that represents light limitation, and $E_{nut(j)}$ is the nutrient limitation for photosynthesis that includes concentrations of nitrogen (N), silica (Si), and phosphorus (P). Because suspension feeders only use the available food source 10 cm above the sediment surface, the following adjustment must be taken into consideration:

$$D_{(j)} = \frac{1000 \cdot 30}{0.42 \cdot 10 \cdot d_{(j)}} \tag{13}$$

where $d_{(j)}$ is the water depth and the final unit for suspension feeder food availability is measured in mg dw m^{-2} 10 cm^{-1} $month^{-1}$ where dw stands for dry weight. The constants are used to convert from 1000 µg to mg, 30 days per month, 42% Carbon content per dry weight, and 10 cm depth of sediment.

Nutrient Limitation

Nutrient limitation ($E_{nut(j)}$) for photosynthesis was modeled according to the Redfield ratio of 106:16:15:1, which assumes that producers use carbon, N, Si, and P proportionally by weight (Redfield, 1934):

$$E_{nut(j)} = MIN\left(\frac{\frac{[N]_{(j)}}{[N]_{(j)} + p_{mic(1)}}}{16}, \frac{\frac{[P]_{(j)}}{[P]_{(j)} + p_{mic(1)}}}{1}, \frac{\frac{[Si]_{(j)}}{[Si]_{(j)} + p_{mic(1)}}}{15} \right) \tag{14}$$

where *[N](j), [P](j),* and *[Si](j)* are concentrations of inorganic nitrogen, phosphorus and silica.

Day Length Limitation

Photosynthesis is limited by light, which varies seasonally. A sine function is used to simulate the seasonal cycle of day length:

$$L_{(t)} = p_{avg} + p_{amp} \cdot \sin\left(\frac{2\pi \cdot (t)}{12} - p_{pha}\right) \quad (15)$$

where $L_{(t)}$ is day length at time t, P_{avg} is the average day length over a year, P_{amp} is the amplitude of the seasonal fluctuation, and P_{pha} is the correction factor for the beginning phase of the sine cycle at a given time.

Prey Density Limitation

Predation can be limited by temperature, salinity, benthic biomass, and predator density. In this study, we considered only the benthic prey biomass, because predation rates on benthic organisms are strongly related to benthic biomass. A complete ecosystem model would also include fish bioenergetics. Predator limitation, $E_{fish(i,j)}$, may be different for different predators, but in this study the same $E_{fish(i,j)}$ was used for all predators, including black drum, red drum and blue crab. However, $E_{fish(i,j)}$ is different for deposit feeders and suspension feeders because of the different vertical distribution of these two groups within the soft-bottom habitat.

In addition to standing stock, a second characteristic of prey is its distribution. The feeding rate of predators is expected to increase exponentially when prey are aggregated in time or space. A logistics-type curve is used to simulate this effect (Montagna et al. 1993):

$$E_{fish(i,j)} = 1 - e^{-p_{(5)} \cdot B_{(i,j)}} \quad (16)$$

where $B_{(i,j)}$ is the biomass of the benthic prey (i = 1 or 2 for deposit feeders or suspension feeders, and j = 1 - 2 for both bays), and $p_{(5)}$ is a new parameter for the aggregation effect. When biomass ($B_{(i,j)}$) is at a very low level, the value of term $E_{fish(i,j)}$ is close to 0, and limitation due to the aggregation effect is nil. When the predator reaches its maximal grazing rate and $B_{(i,j)}$ is very high, the term $E_{fish(i,j)}$ is close to 1, and the limitation due to aggregation is at the maximal level.

Goodness of Fit

To evaluate the model performance, the percent root mean square (%RMS) difference was calculated between model outputs and observations over the period 1988 - 2005 (Eq.1).

$$\%RMS = \sqrt{\frac{\sum \frac{(X_{mod} - X_{obs})^2}{N}}{\sum \frac{(X_{obs})^2}{N}}} \times 100 \qquad (17)$$

where X_{mod} and X_{obs} are model simulations and data, respectively. N is the size of the sample (number of individual data points).

Modeling Tool

The study model has been previously run and calibrated for the years between 1988 and 1996 (Montagna and Li 1996). This was done using the FORTRAN 77 language and facilitated by the PC software package SENECA (Simulation Environment for Ecological Application) (de Hoop et al. 1989).

Model Calibration

The Lavaca-Colorado Estuary was divided into two bays: the primary bay, Matagorda Bay, and the secondary bay, Lavaca Bay. The model was then calibrated individually for each of these bays (Montagna and Li 1996). Sampling of the Lavaca-Colorado Estuary began in November of 1984 (Kalke and Montagna 1991). The data set is incomplete, however, and periodically non-existent between 1984 and 1988 (Table 2.1). Sampling took place from 1988-1990, but less than four times per year. Unfortunately, there is a period from 1989 to 1990 when observations for nutrients are missing. A 15-year period, beginning in January 1991 and ending October 2005, is the best data series in terms of completeness, with four continuous observations for all variations per year in both bays. However, the calibration parameters were derived from the eighteen-year series (1988-2005) and used in order to perform a long-term simulation. A newly-derived model parameter set was used in order to simulate benthic macrofauna biomass in the Lavaca-Colorado Estuary from April 1988 to October 2005.

Day Length

Table 2.4 presents the results of calibration for equation (15) that simulates the day length (Fig. 2.6), and were determined by Montagna and Li (1996). The day length simulation is very close to the observed data (Fig. 2.6). These results are used as the parameters in equation (15) and became a forcing function for the main model.

Table 2.4. Calibration parameters for Equation (15) for the simulation of day length.

Parameter	Definition	Best Fit Value	Reduced Ranges		Initial Ranges	
			Minimum	Maximum	Minimum	Maximum
p_{avg}	Average value of the harmonic function	12.15849	12.15849	12.1585	11	13
p_{amp}	Amplitude of sinus function	1.755811	1.755809	1.755815	1	2
p_{pha}	Phase of sinus function at reference time	0.2244535	0.2244535	0.2244536	0	1

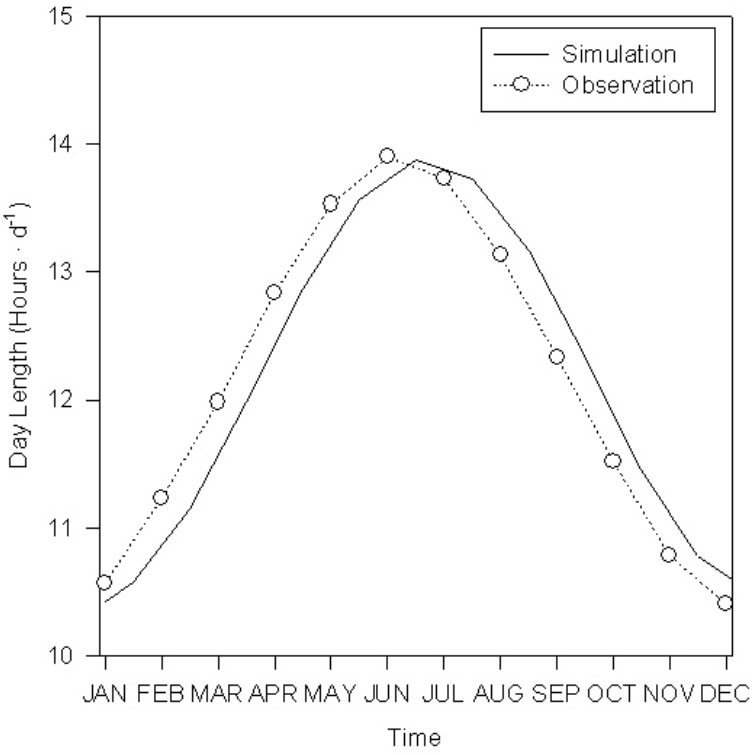

Figure 2.6. Simulation of day length over period of one year.

Primary Production

Table 2.5 presents results of the calibration of parameters for simulation of primary production that were formulated by equations (12) and (14) (Montagna and Li 1996).

Table 2.5. Calibration of parameters for primary production.

Parameter	Definition	Best Fit Value	Calibrated Ranges		Initial Ranges	
			Minimum	Maximum	Minimum	Maximum
$p_{mic(1)}$	Nutrient limitation	1.909248	1.856677	1.914145	0.5	2
$p_{mic(2)}$	Maximal primary production (g C • m • h)$^{-2\ -1}$	0.5093191	0.5076211	0.5105351	0.5	2
$p_{mic(3)}$	Temperature Limitation	11.39727	11.29732	11.39727	10	50

POM Level

To calibrate the POM level, $P_{(sed)}$ was set at 18 (Li et al. 1996) in equation (10) and the carbon concentration was simulated (Montagna and Li 1996). The results of the calibration are listed in Table 2.6, which estimates the carbon concentration levels that were fit to the observations.

Table 2.6. Parameters for the calibration of POM levels (g dw • m^{-2} •10 cm^{-1}) for both bays.

Parameters	Definition	Best Fit Value	Calibrated Ranges		Initial Ranges	
			Minimum	Maximum	Minimum	Maximum
p_{sed}	Sediment Weight (g dw • m •10 cm)$^{-2}$ $^{-1}$	18[a]				
$p_{pom(1)}$	POM level for Lavaca Bay	10136.14	9973.851	10279.69	1000	30000
$p_{pom(2)}$	POM level for Matagorda Bay	10007.6	9589.807	10364.56	1000	30000

[a]Calibrated by Li et al.(1996)

Data Series

The model was calibrated individually for each bay system using the period from April 1988 to October 2005. The initial ranges for the seventeen parameters were set within the same values for each bay. There were fifteen thousand calibration runs, and all parameter ranges were reduced to less than 50% of the initial ranges. The results of the calibration are presented in the Table 2.7.

Table 2.7. Best fit parameter values from the calibration of the Lavaca-Colorado Estuary for the continuous seventeen-year database: 1988 - 2005. Lavaca Bay did not have red drum and black drum observations, so the parameters $g_{(1,j,1)}$ and $g_{(1,j,2)}$ were not computed. The parameters are defined in equations (4-16).

Parameter	Best fit values for each estuary		Initial ranges
	Lavaca Bay	Matagorda Bay	
$p_{(1)}$	44.95681	40.33941	20, 45
$p_{(2)}$	17.22897	17.28709	20, 45
$p_{(1,3)}$	36.68249	36.37392	20, 40
$p_{(1,4)}$	24.01417	66.51145	0, 100
$g_{(1,j,1)}$	-	3.944138	0, 5
$g_{(1,j,2)}$	-	4.975888	0, 5
$g_{(1,j,3)}$	1.037885	1.228183	0, 5
$p_{(5)}$	1.27698E-3	3.77488E-3	0.001, 0.01
$r_{(2)}$	6.230073	7.617166	5, 20
$c_{(1)}$	31.06771	31.89199	30, 70
$p_{(2,3)}$	8.028702	8.270572	20, 40
$p_{(2,4)}$	55.46861	82.03805	0, 100
$g_{(2,j,1)}$	-	0.8315539	0, 5
$g_{(2,j,2)}$	-	0.5318332	0, 5
$g_{(2,j,3)}$	2.075487	3.731529	0, 5
$r_{(2)}$	5.555974	18.62822	5, 20
$c_{(2)}$	99.30896	54.33509	50, 100

RESULTS

The simulations of benthic biomass are based on the best fit parameters from the calibration of the period (1988 - 2005). All of the simulations were run from April 1988 until October 2005, and were compared to observed benthic macrofauna biomass data (Figs. 2.7 - 2.10).

Simulation of Benthos Biomass

The biomass variance was determined in Lavaca Bay and Matagorda Bay for deposit feeders and suspension feeders. Benthic biomass variance is defined as the average difference between observed and simulated biomass for a given time period (Table 2.8). The simulations for both bays and each feeding group fit the observed data relatively well during the calibration period, 1988 - 2005 (Figures 2.7 – 2.10). The most important part of the fit is that in general, the trends in the prediction over time fit the trends in the observed biomass over time. The worst fit was for deposit feeders in Lavaca Bay where the model predicts rather constant biomass from 1991 to 1994 when the biomass actually decreased. The model predicts increasing biomass from 2000 to 2005 when the observed biomass data showed decreasing trend. Although the variance was high for Matagorda Bay deposit feeders also, the trends actually fit the data well. The high variance for Matagorda Bay is explained by the higher average biomass. As in most biological data, the variance increases with the mean.

Table 2.8. Biomass variance in the Lavaca-Colorado Estuary.

	Lavaca Bay		Matagorda Bay	
Period	Deposit Feeders	Suspension Feeders	Deposit Feeders	Suspension Feeders
1991 - 1995	4.497	1.871	5.255	0.617
1996 - 2000/07	-	-	5.488	1.552
1996 - 2001/04	1.512	0.583	-	-
1996 - 2006	7.879	2.036	8.993	6.814
2001/04 - 2006	14.56	3.57	-	-
2000/07 - 2006	-	-	13.322	13.063
1991 - 2006	6.692	1.978	7.552	4.561

Unfortunately, three of the four simulations predict much higher biomass than those observed after the year 2000, and one predicts higher biomass after the year 2003. This increase in macrofauna biomass in the simulation is caused by a decrease in predator density (particularly blue crabs) in both bays at that time. There was a pronounced decrease in blue crab populations from 2000 to 2006 in Lavaca Bay (Figure 2.11) and Matagorda Bay (Figure 2.12).

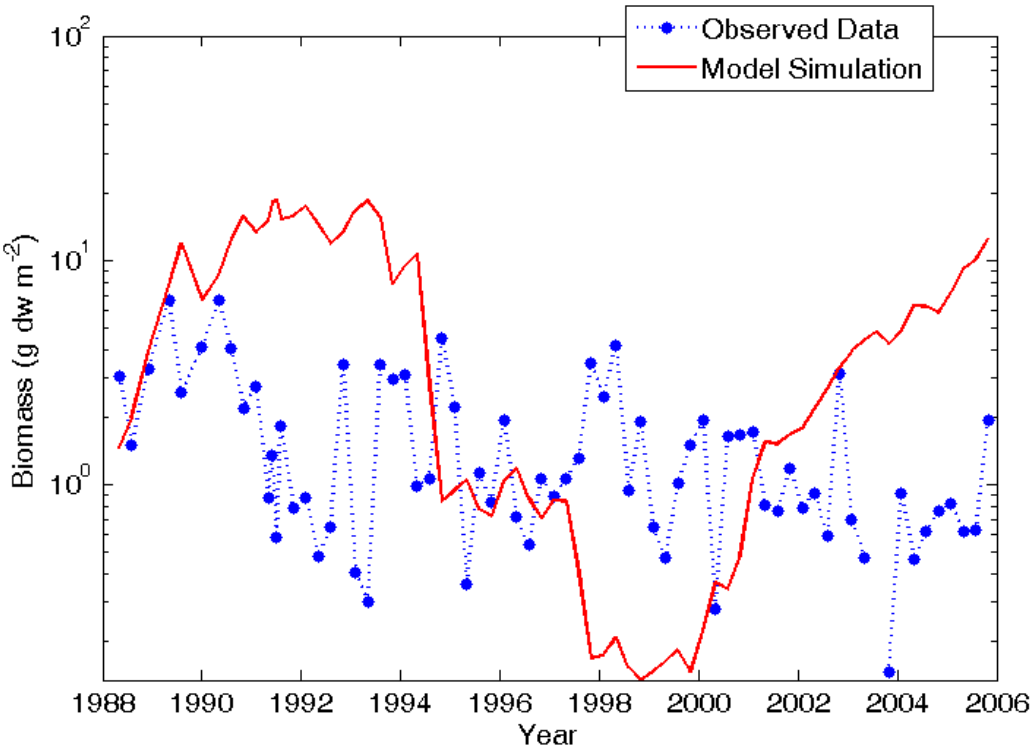

Figure 2.7. Simulation of deposit feeder biomass in Lavaca Bay for the period 1988 - 2005.

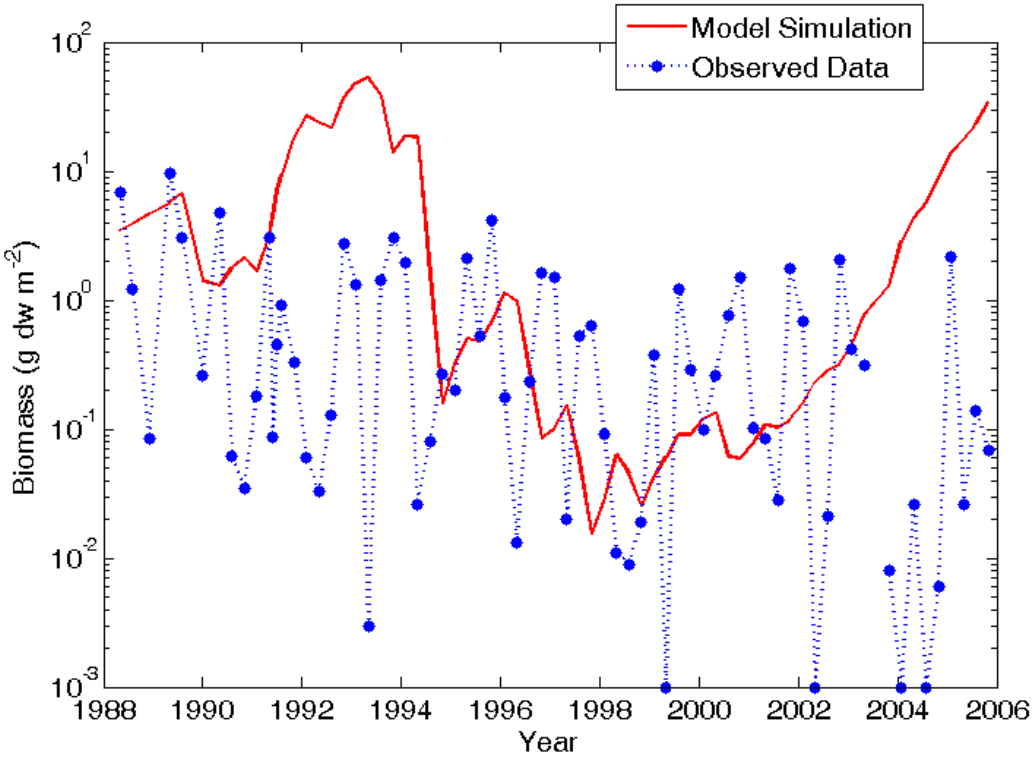

Figure 2.8. Simulation of suspension feeder biomass in Lavaca Bay for the period 1988 - 2005.

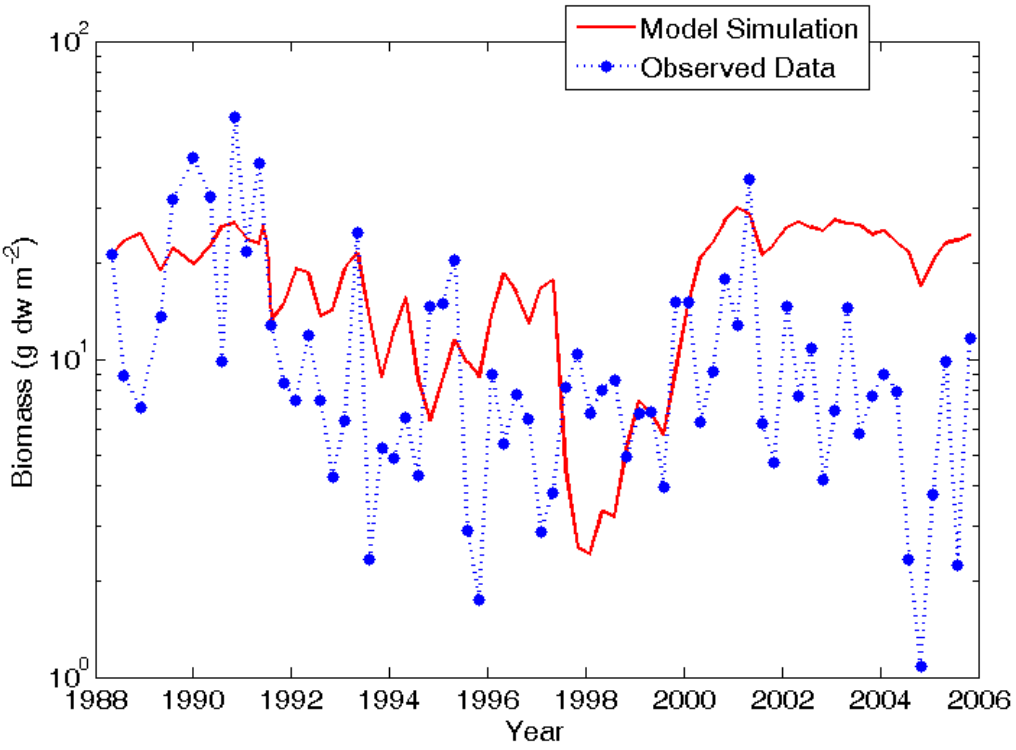

Figure 2.9. Simulation of deposit feeder biomass in Matagorda Bay for the period 1988 - 2005.

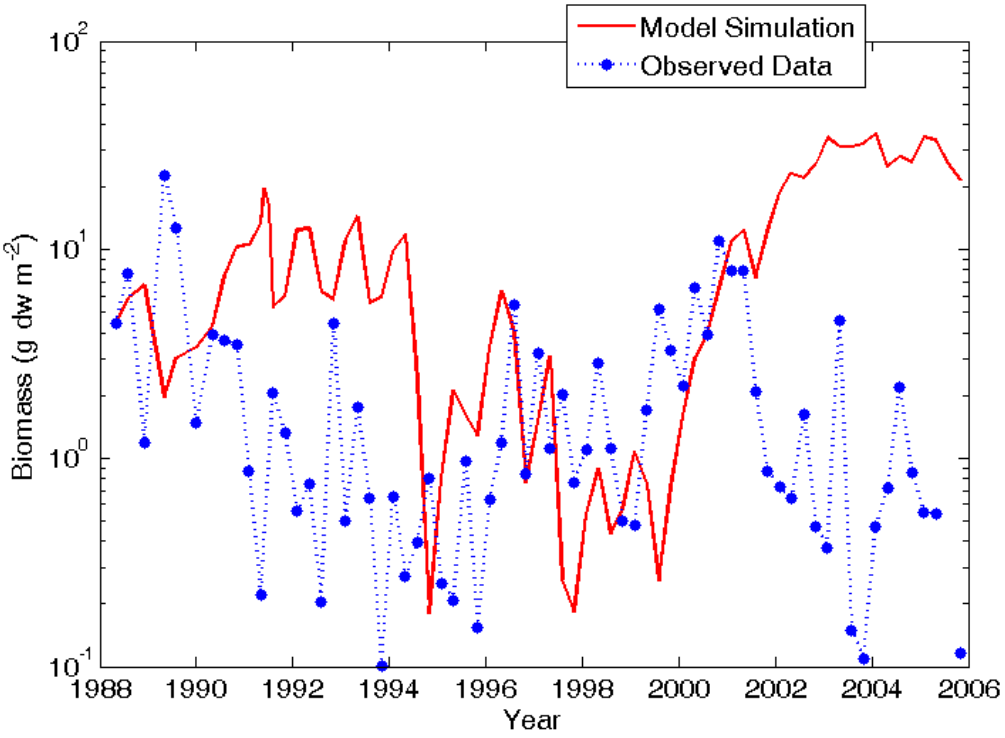

Figure 2.10. Simulation of suspension feeder biomass in Matagorda Bay for the period 1988 - 2005.

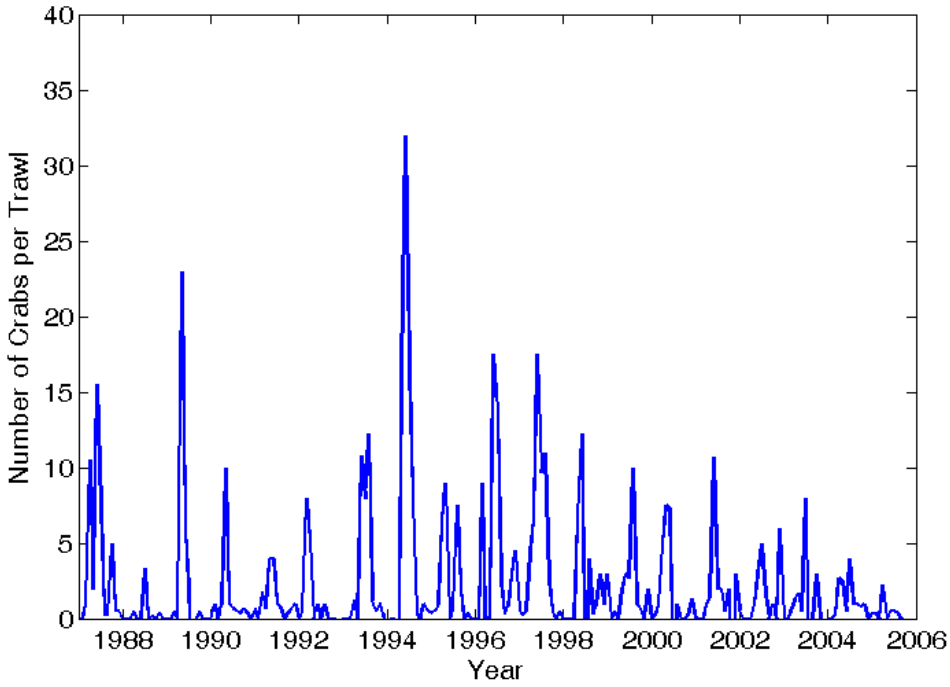

Figure 2.11. Crab population in Lavaca Bay from 1987 - 2006 (TPWD data).

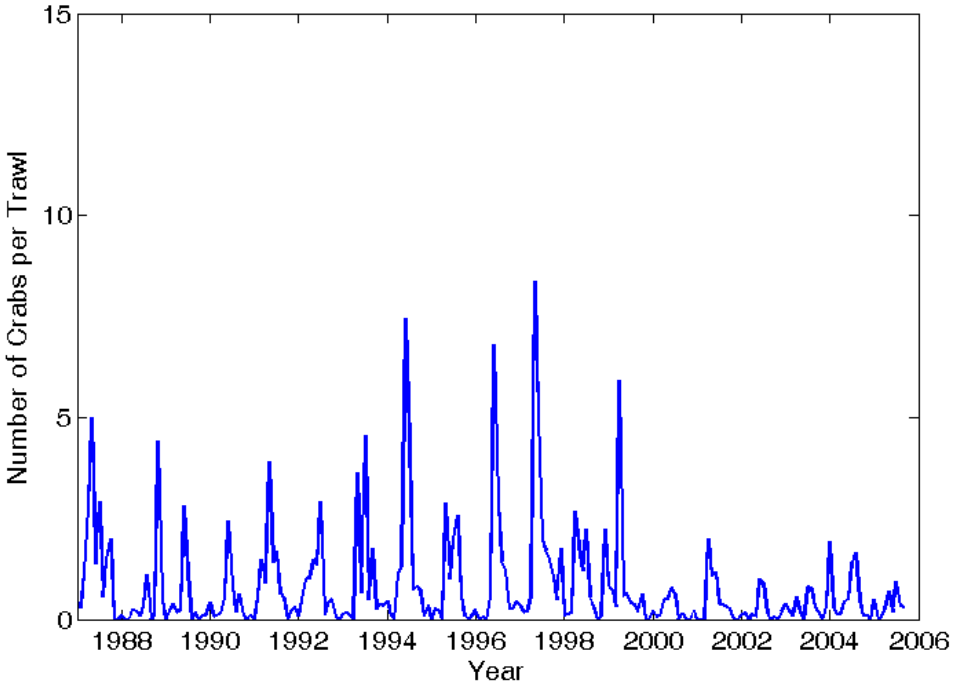

Figure 2.12. Crab population in Matagorda Bay from 1987 - 2006 (TPWD data).

Error Analysis

The percent RMS differences between the simulated and observed biomass for deposit and suspension feeder in Lavaca Bay were 408% and 1014%, respectively. Similar comparison between the model simulation and observation in the Matagorda Bay shows 81% difference for the deposit and 308% for the suspension feeder biomass, respectively. The deposit feeder biomass simulation in Matagorda Bay was the best result among the four simulations and the suspension feeder simulation in Lavaca Bay had the worst fit. The overall model performance, however, did not seem very successful. The major deviation is ascribed to the misfit for the two periods of declining biomass due to predation: 1994-1994 and 200-2005. The model could not capture the decline in biomass properly, because the model had an increasing trend in simulations. This indicates the governing equation for predation needs improvement. Improvement of the model structure is left to further study.

Benthic Biomass Change Due to Salinity Increase and Nutrient Decrease

The deposit feeder and suspension feeder biomass were simulated to reflect the response to a potential increase in salinity and decrease in nutrients in both bays (Figs 2.13 - 2.20). Because principal components analysis showed that low inflow was correlated to high salinity and low nutrients (Chapter 1), the combined effects of salinity increase and nutrient decrease was investigated in the current study. The sensitivity test was performed at 1% intervals up to a 30% increase from original salinity observations, combined with 30% decrease from original nutrients level with same intervals. The simulations predict benthos biomass based on the following scenario: what if the salinity had been higher and the nutrients had been lower than they used to be over the historical period? Simulations of deposit feeders followed a pattern of gradual increases followed by a dramatic drop in biomass about once a year, with the lowest biomass concentration occurring around 1998 and 1999 (Figures 2.13 and 2.14). After the year 2000, the biomass concentrations showed signs of slowly increasing and leveling off. Simulations of suspension feeder biomass followed the same pattern as the deposit feeder biomass, and had a trend of a slower increase in biomass after the year 2000 (Figs. 2.15 and 2.16).

As salinity increased and nutrients decreased, the model predicted an increase in deposit feeder biomass for both bays (Figs. 2.14 and 2.18). The scale of the increase was higher in Lavaca Bay (about 6 g dry weight m^{-2}) than in Matagorda Bay (about 4 g dry weight m^{-2}). However, as salinity increased, it was correlated to a decrease in suspension feeder biomass in both bays (Figs. 2.16 and 2.20). The scale of the decrease was small in Lavaca Bay (about 4 g dry weight m^{-2}) compared to Matagorda Bay (about 10 g dry weight m^{-2}).

Total biomass concentration was calculated by adding deposit and suspension feeder biomasses. Lavaca Bay total biomass slightly increased with percent change in salinity and nutrient (Fig. 2.21) because the decrease in suspension feeders was small compared to the increase in deposit feeders. In Matagorda Bay, biomass initially started decreasing rapidly, and then showed little change in biomass above 25% level of salinity increase because the initial phase of increasing trend in deposit feeders was not as rapidly as that of decreasing trend in suspension feeder biomass (Fig. 2.22).

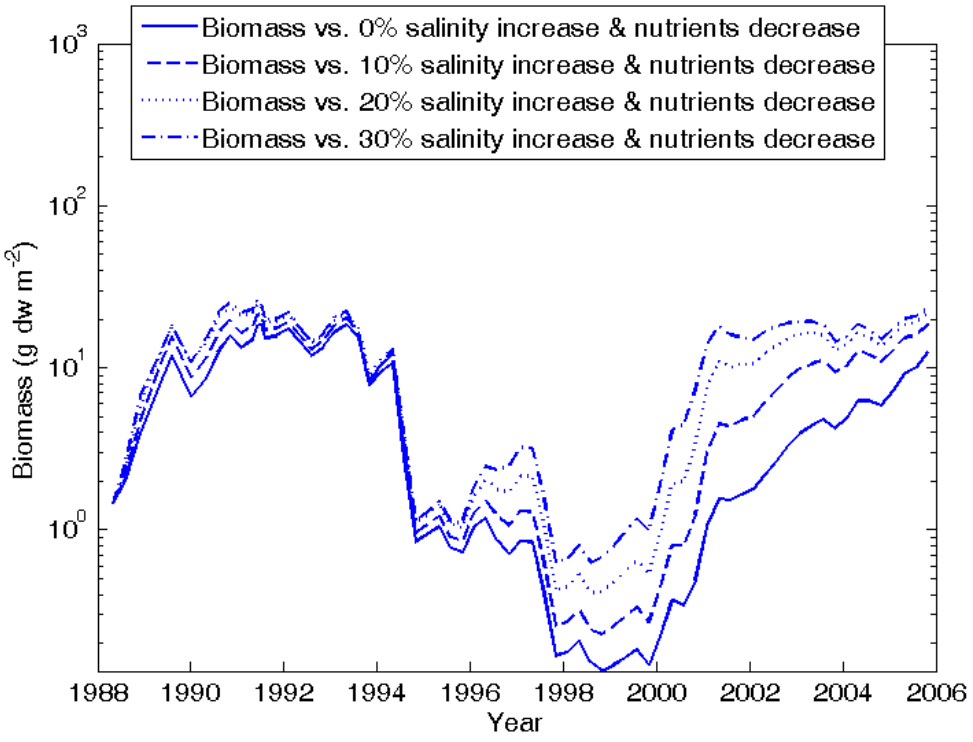

Figure 2.13. Simulations of deposit feeder biomass reflecting different salinity increases in Lavaca Bay.

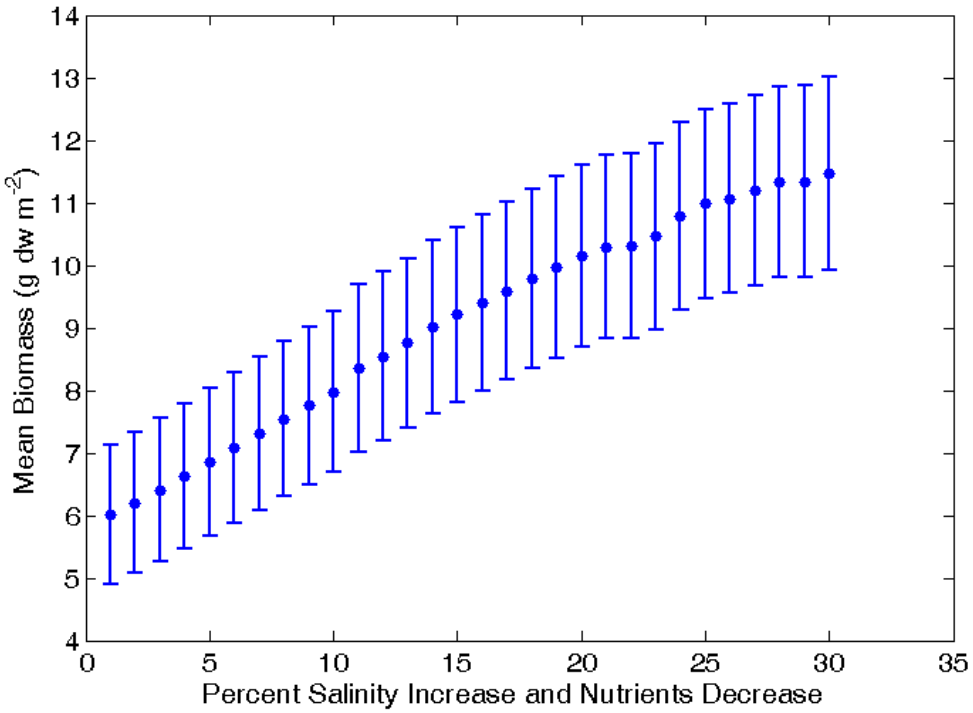

Figure 2.14. Mean (± std error) biomass concentration of deposit feeders in Lavaca Bay in response to changes in salinity.

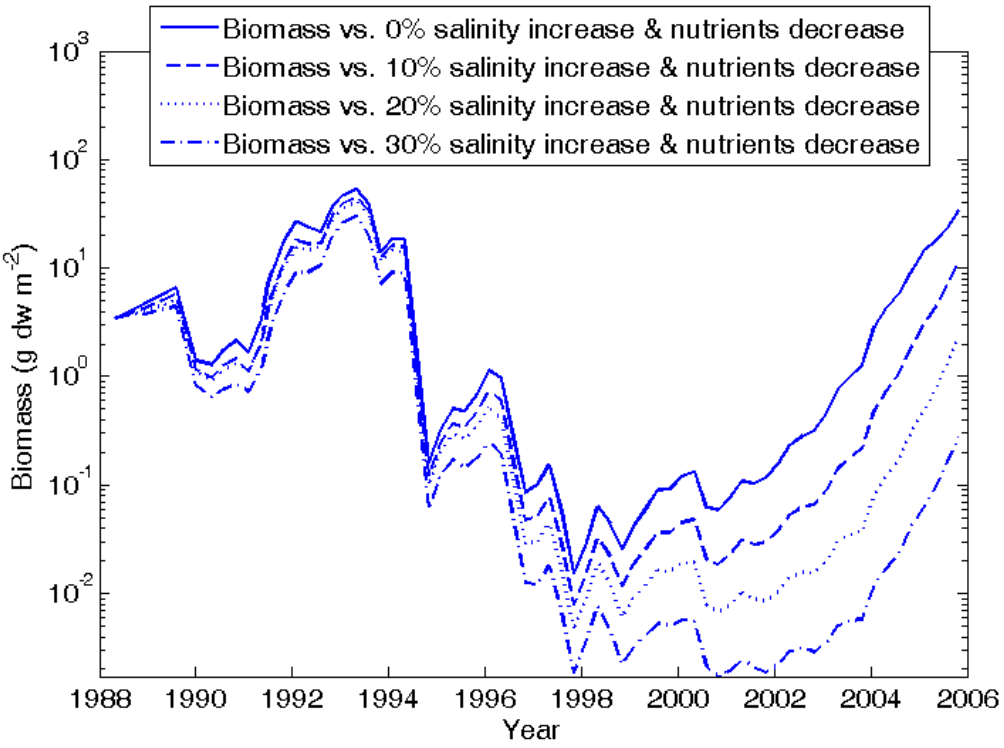

Figure 2.15. Simulations of suspension feeder biomass reflecting different salinity increases in Lavaca Bay.

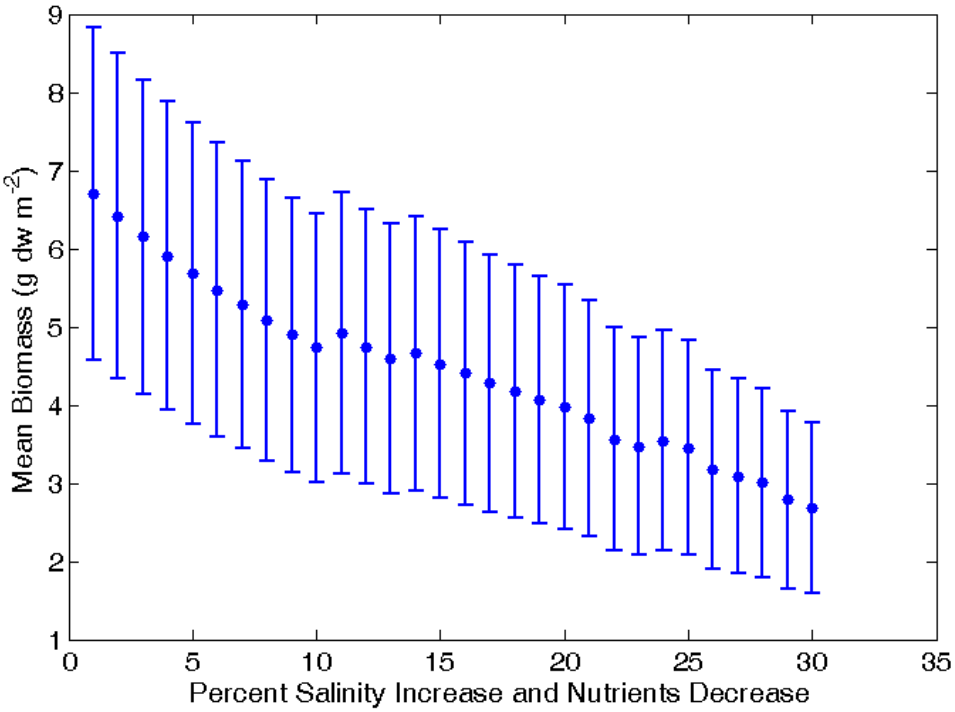

Figure 2.16. Mean (± std error) biomass concentration of suspension feeders in Lavaca Bay in response to changes in salinity.

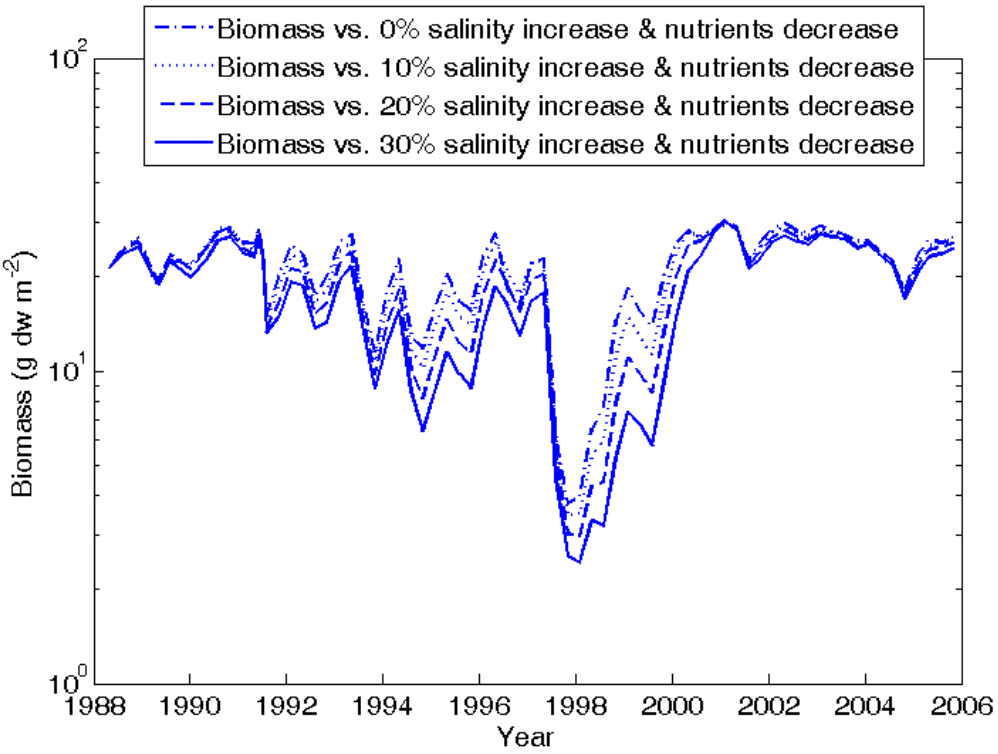

Figure 2.17. Simulations of deposit feeder biomass reflecting different salinity increases in Matagorda Bay.

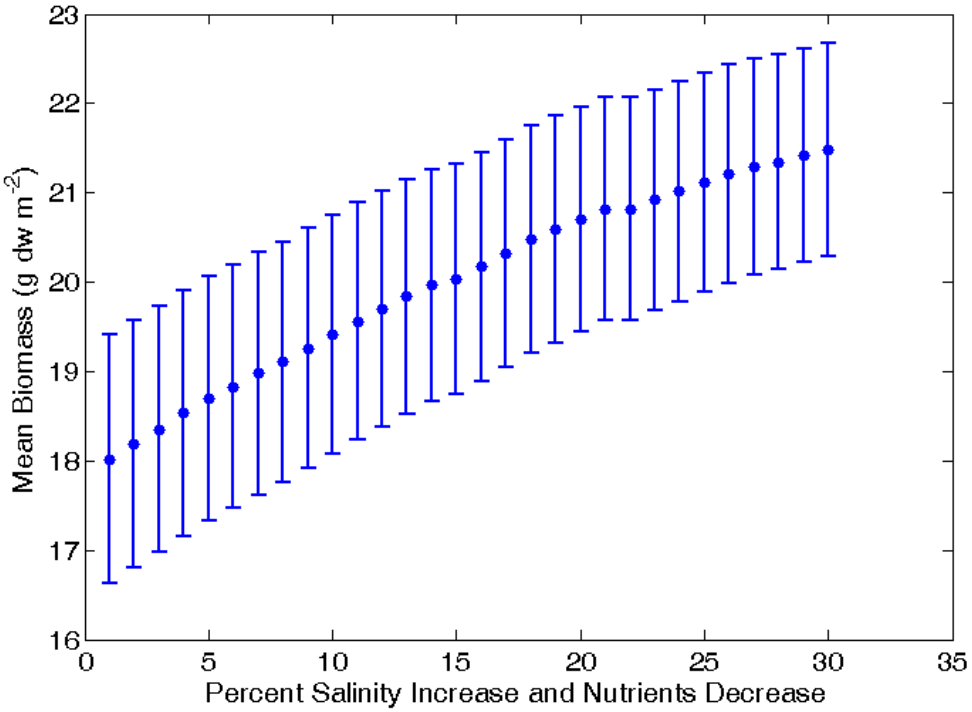

Figure 2.18. Mean (± std error) biomass concentration of deposit feeders in Matagorda Bay in response to changes in salinity.

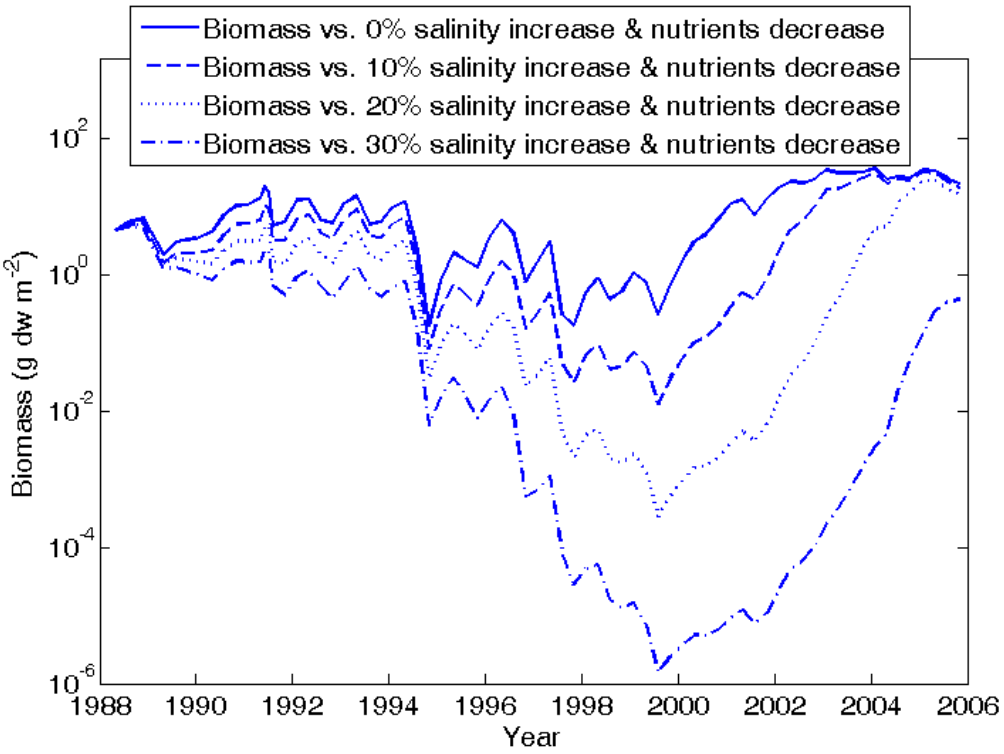

Figure 2.19. Simulations of suspension feeder biomass reflecting different salinity increases in Matagorda Bay.

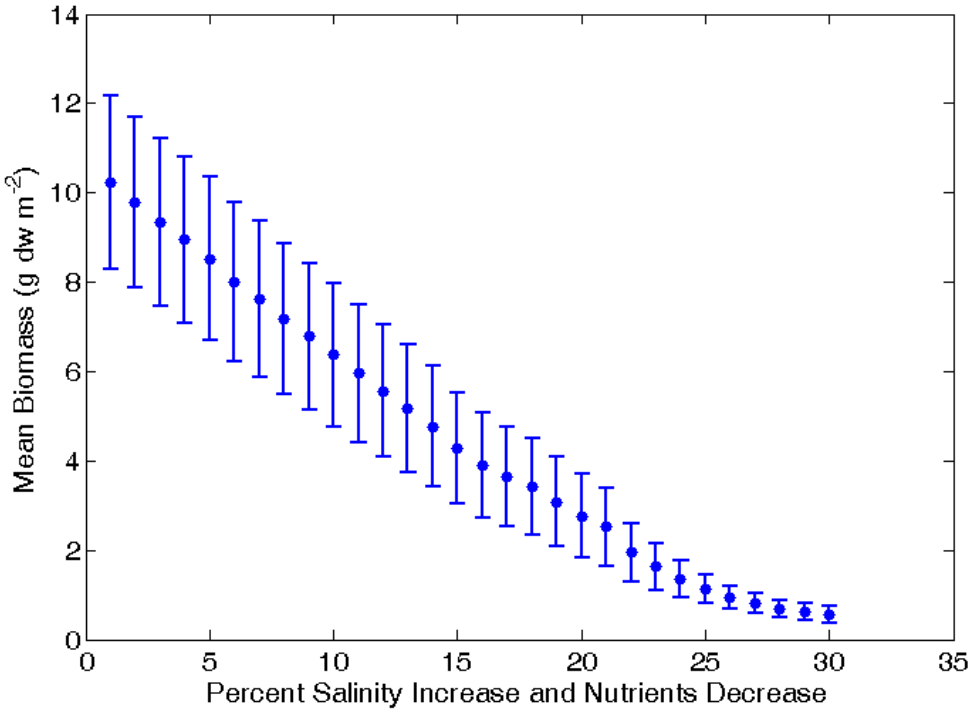

Figure 2.20. Mean (± std error) biomass concentration of suspension feeders in Matagorda Bay in response to changes in salinity.

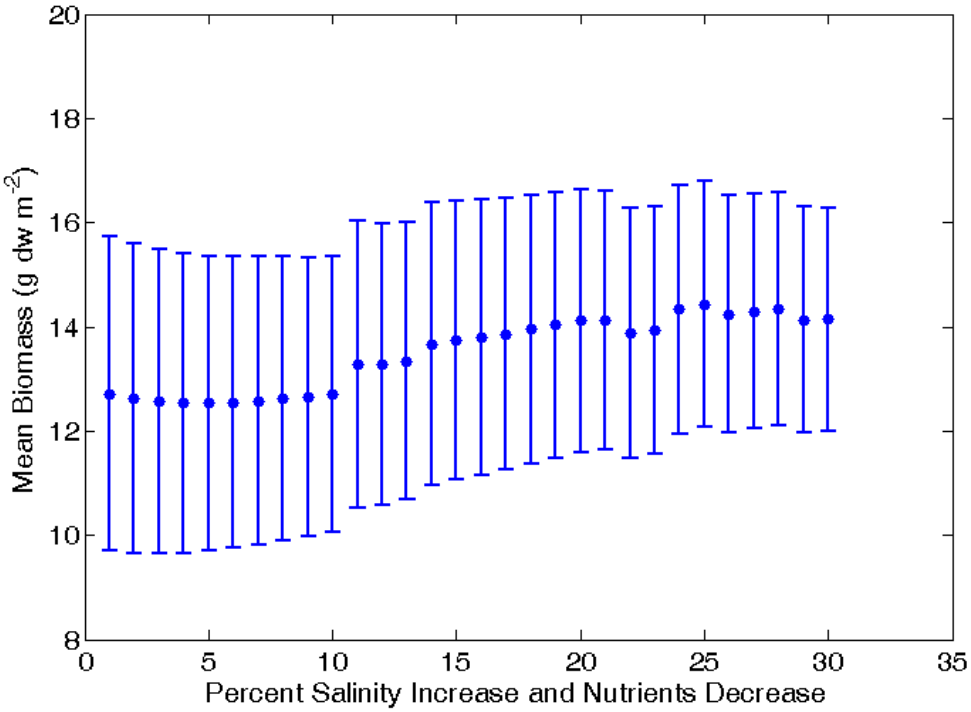

Figure 2.21. Mean (± std error) total biomass concentration in Lavaca Bay in response to changes in salinity.

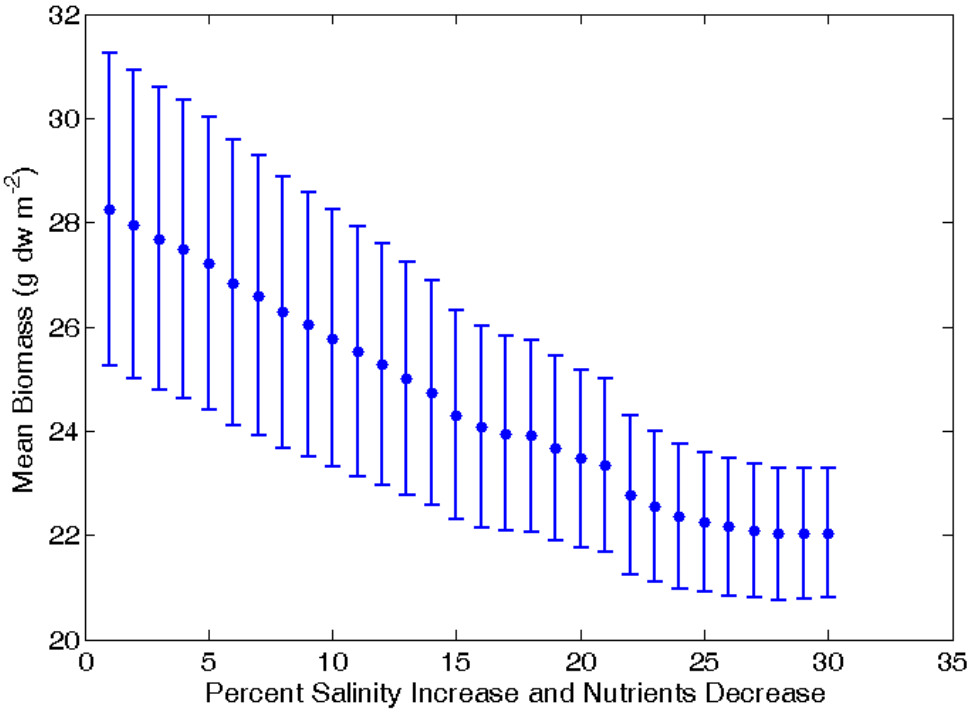

Figure 2.22. Mean (± std error) total biomass concentration in Matagorda Bay in response to changes in salinity.

DISCUSSION

Colorado River Diversion

A diversion channel from the Colorado River was built in 1991 to mitigate the effects of reduced freshwater inflow to Matagorda Bay from past water development projects. In 1992, a dam was added to divert the river into the channel. At that time, alterations in the exchange of seawater from the Gulf of Mexico also occurred, which included closure of Parker's Cut Dam near the mouth of the Colorado River to prevent saltwater from entering Matagorda Bay to help reduce the salinity.

A study following the alterations found diversity, abundance, and biomass to be higher in Matagorda Bay versus Lavaca Bay (Montagna 1994). It was suggested that as a result of the diversion project, Matagorda Bay became a healthier and more productive ecosystem than Lavaca Bay as indicated by increased diversity and biomass. This conclusion, however, is confounded by much larger influences of human activities and development in Lavaca Bay, which should be taken into consideration when comparing the two ecosystems (Montagna 1994).

The Matagorda Bay Health Evaluation (MBHE) component of the LCRA/SAWS Water Project (LSWP) Study Plan was created to assess the environmental effects that could result from further changes to inflow patterns in the Matagorda Bay system.

A bio-energetic model was developed that related macrobenthic biomass to salinity within the Lavaca-Colorado Estuary (Montagna and Li 1996). This model was applied to the present study to assess the role of freshwater inflow in controlling benthic productivity. The existing model was used to simulate the effects due to a potential change in freshwater inflow, which might alter salinity and nutrient concentrations in the model to simulate how benthic productivity might respond.

Modeling Benthic Biomass

Simulations of the Lavaca-Colorado Estuary fit the observed data relatively well for the periods 1988 – 1991 and 1995 – 2001 (Figs. 2.8 - 2.10). Specifically, model simulations overestimated biomass of both deposit and suspension feeders at both Lavaca Bay and Matagorda Bay for two simulation periods: years between 1991 and 1995 and years between 2002 and 2005. The higher predicted benthic biomass can be explained by a decrease in predator populations, particularly blue crabs, which releases predation pressure and could cause prey populations to increase. Numbers of blue crabs caught in the individual bays have been decreasing with a drop in numbers in the year 2000 (Figs. 2.11 and 2.12). In the model, predator density and benthic biomass are inversely correlated, which demonstrates that the model behavior is strongly driven by predation. Actual observations in benthos biomass, however, show no indication of increase despite the decrease in predator density. This trend can be explained by three possibilities: 1) another environmental factor is causing both predator density and benthic prey density to decrease, 2) that the low level of benthic biomass between 2002 and 2005 may have influenced predator density to decrease, or 3) top-

down control is not important to benthos in this ecosystem. More investigation is needed to better explain the trophic relationship between prey and predator in this ecosystem, and this will guide modification of the model structure to achieve more realistic behaviors. These issues must be left to further studies in the future.

Benthic Biomass Response to Salinity and Nutrient Changes

Simulations of deposit and suspension feeder biomass exhibited responses caused by changes in salinity and nutrients in both bay systems (Figs. 2.13-2.20). In general, when salinity increased with decreasing nutrient concentrations, deposit feeding biomass increased while suspension feeding biomass decreased.

Kalke and Montagna (1991) studied sites in the upper portion of Lavaca River and Bay from 1984 - 1986 to determine the effects of freshwater inflow on macrobenthos. A high freshwater inflow rate caused low salinity species to populate the area, and, therefore, it was determined that freshwater is necessary in the upper portion of the bay to induce recruitment of low salinity species. Following an inflow event, Chironomid larvae (suspension feeders) and *Hobsonia florida* (polychaete) increased in density, as both prefer lower salinity environments. In contrast, the mollusks, *Mulinia lateralis* and *Macoma mitchelli* (suspension feeders), and S*treblospio benedicti* and *Mediomastus californiensis* (deposit feeders) increased in benthic biomass during periods of low freshwater inflow (Kalke and Montagna 1991). These findings from the previous studies are consistent to the results of sensitivity tests conducted in the present study, which shows that salinity increase combined with nutrient decrease caused deposit feeder biomass to increase and suspension feeder to decrease (Figs. 13, 15, 17, and 19). Although the effects of salinity and nutrients was not presented separately, the sensitivity tests showed small discrepancies between the result of salinity-only and that of salinity-nutrient combined case (data not shown). The small difference between the two cases in the sensitivity analyses is related to small sensitivity of the nutrient limitation term in the model structure (equations 12 and 14).

It was also found that chlorophyll *a* concentrations increased in the Lavaca River and Lavaca Bay during high inflow rates, and this indicates that primary production is stimulated by increased nutrients loadings in discharged freshwater (Kalke and Montagna 1991). As salinity increased during periods of low inflow, the chlorophyll a concentration decreased due to reduced nutrient loadings and elevated grazing pressure by the growing population of mollusks (Kalke and Montagna 1991). Again, these findings confirm that the results of sensitivity tests in this study are consistent with field observations.

Rozas et al. (2005) investigated the effect of freshwater inflow in the Breton Sound Estuary, Louisiana, and found releases of freshwater from the Caernarvon diversion structure lead to an increase in submerged aquatic vegetation (SAV) and dissolved oxygen concentrations. Macrofauna populations also increased in density and biomass with increasing inflow, which was probably due to the growing SAV coverage creating more habitat (Rozas et al. 2005).

Total benthic biomass in Lavaca Bay was found to slightly increase in concentration as salinity increased (Fig. 2.21), which indicates that reduced inflow rates to this bay would not

likely harm benthic community productivity. Total benthic biomass in Matagorda Bay at first decreased rapidly with increasing salinity but then declined more gradually after 21 % increases (Fig. 2.22). Previous studies examined how benthic macrofaunal community structure varied over space and time in response to changes in inflow in Lavaca and Matagorda Bay (Kinsey and Montagna 2006). The results found Matagorda Bay to be a healthier ecosystem in general compared to Lavaca Bay. There was also a direct relationship between freshwater inflow and salinity on benthic communities. Distinct station differences were found in community structure along salinity gradients, which implies that reduced flows will cause upstream communities to take on characteristics of downstream communities (Kinsey and Montagna 2006).

Community Responses

The ecology of south Texas estuaries has been studied for many years. Montagna and Kalke (1995) observed how freshwater inflow benefits estuaries in south Texas. *Mulina lateralis* was one of the dominant species found and is important as the predominant food source for black drum. This species was frequently found in secondary bays along the Texas coast where freshwater inflow has a large impact. It was concluded that recruitment events for *M. lateralis* are likely initiated by a significant change in salinity (Montagna and Kalke 1995). Freshwater inflow has obvious benefits to estuaries along the Texas coast. Only estuaries with high freshwater inflow rates support productive shellfish industries (Montagna and Kalke 1995).

The present study combined long-term data sets with an energetic model to predict how altering salinity in the Lavaca-Colorado Estuary might affect productivity of the macrobenthos population. Based on observed long-term patterns and model predictions, it appears that reducing freshwater inflow may cause the upper river communities to take on downstream community appearance, i.e., more salt tolerant species would dominate the community. The Lavaca Bay benthic community appears to benefit from reduced freshwater inflow by increasing in biomass. The macrobenthos in Matagorda Bay appear to decrease in biomass concentration rapidly at first, and then gradually approach the constant level in numbers. This effect is probably due to the benthic community acclimating to the higher salinity, or more salt tolerant species populating the area.

Freshwater inflow into an estuary is recognized as an important factor in estuary productivity, affecting physical, chemical, and biological aspects of the system (Montagna et al. 2002). River inflow drives increased circulation, salinity gradients, and sediment transport as well as enhancing the productivity of coastal fisheries (Powell et al. 2002). Nutrients from freshwater inflow become incorporated into the estuarine food web, can increase vegetation, and enhance the secondary production in the area (Rozas et al. 2002). It is clear that freshwater inflow is important in maintaining estuarine productivity. Management studies should consider not only the quantity of inflow required, but also seek to determine the regime of timing and magnitude of inflow that is needed to maintain functional, healthy ecosystems.

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Chapter 3

Organic Matter – Productivity Assessment

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INTRODUCTION

The recently completed study on "Bay Productivity" completed by Paul Jensen (PBS&J 2006) discovered a very large data gap. It has been found that dissolved organic matter (DOM) is an important component of materials transported to bays by inflow, yet there is no information on this complex of compounds for Matagorda Bay, or any other Texas estuary. Also, recent work in Lavaca, Tres Palacios, and Matagorda Bays shows changes in dissolved oxygen caused by respiration, and thus net ecosystem metabolism increases dramatically during freshwater inflow events and returns to normal values after just a few days (Russell et al., 2006; 2007). This burst of productivity can only be explained by bacterial use of DOM.

Dissolved oxygen concentrations in estuarine waters can be directly related to the characteristics and reactivity of soluble organic substances within the estuarine water column. This collection of soluble organic material is known as the DOM pool. The relationship between dissolved oxygen and DOM is primarily regulated by the water column bacterial population that utilizes DOM during respiration, a process that can reduce in dissolved oxygen levels. Key sources of DOM to estuarine waters include freshwater discharge and primary productivity. Typically, DOM delivered by river flow is less bioavailable than DOM produced directly within estuarine waters by biological activity due to the more complex chemical structure of terrestrially-derived substances that comprise freshwater DOM (Leff and Meyer 1991). However, while riverine DOM may not be readily consumable by bacteria upon initial entry to the estuary, the potential does exist for this material to provide important fuel for microbes during its residence within estuarine waters if it can be broken down into smaller more labile (i.e., consumable) compounds. The primary mechanism by which complex DOM can be broken into bioavailable compounds is through the absorption of solar radiation (Moran and Zepp, 1997; Moran et al. 2000).

Within the DOM pool, there exists a subset of organic substances that are capable of absorbing photons of light energy and undergoing a variety of photoinduced chemical reactions. This pool of light absorbing material is called chromophoric dissolved organic matter (CDOM). As part of the larger DOM pool, CDOM sources can either be primarily terrestrial in nature formed during the degradation of upland freshwater vegetation or primarily marine originating from phytoplankton or marine bacterial processes. Terrestrially derived CDOM is typically more photoreactive than marine CDOM. Upon exposure to sunlight, the larger CDOM structure can be a source of labile organic compounds as smaller

'pieces' of its original complex structure are released into the water. For estuaries with substantially long residence times like many of those along the Texas coast including the Lavaca-Matagorda system, the photochemical production of bioavailable compounds may be an important factor in controlling respiration throughout the system. In contrast, since CDOM strongly attenuates the penetration of light through the water column, it can limit photosynthesis reducing the production of labile DOM by in situ processes (Granskog et al. 2007).

The primary purpose of the research presented in this section was to assess the potential impact of CDOM photodegradation on the production of bioavailable organic material and subsequent contribution to bacterial respiration within the Lavaca-Matagorda system.

METHODS

Field Locations

The study was performed in two locations: Lavaca Bay near the mouth of the Lavaca River, and Eastern Matagorda Bay near the mouth of the Colorado River (Figure 3.1). Three of the stations (A, E and F) were the same as those used in the long-term benthic indicator study (Chapter 1). A new station (FD) was established between station A and the mouth of the Lavaca River. The station locations were chosen based on salinity at the time of sampling in April 2007.

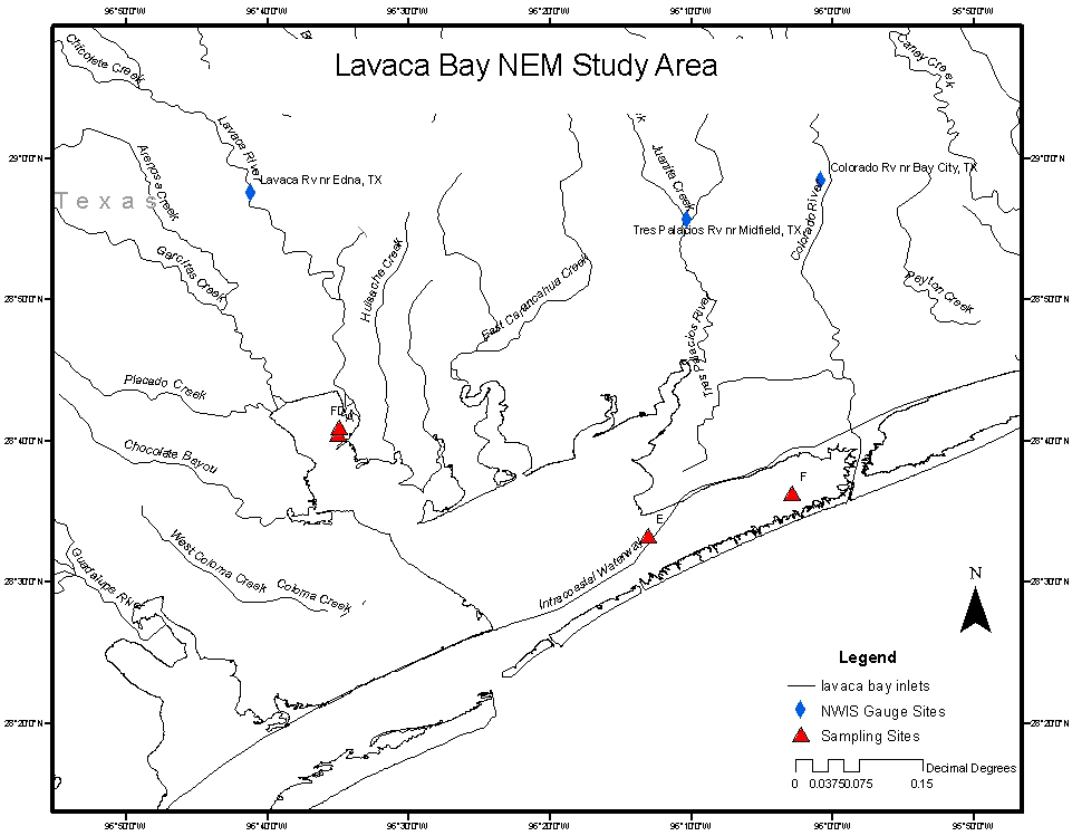


Figure 3.1. Station locations and locations of USGS gages.

Samples were taken twice: April 2-5, 2007 and July 23-26, 2007. The daily flow rates from three USGS gages (Figure 3.1) were summed over the 10-day period prior to sampling. About 3.6 times more flow occurred in July (651 million m^3) than April (177 million m^3).

Water Sampling

Water samples were collected in glass vials or acid cleaned 1 L polycarbonate bottles at each sampling station and filtered on site using 0.2 µm polycarbonate Meissner cartridge filters using a peristaltic pump and Teflon tubing. All filters and tubing were acid cleaned prior to use and glass vials were baked at high temperature in a muffle furnace to remove any potential for organic contamination. Samples were stored in the dark at 4 C until analysis.

DOC analysis

Dissolved organic carbon analysis was performed using high temperature catalytic oxidation on a Shimadzu TOC-V analyzer based on the method described in Alvarez-Salgado and Miller (1998). To ensure accuracy, a seawater reference standard was incorporated into each analysis. For this study, DOC concentrations are used as a proxy for the magnitude of the larger DOM pool.

CDOM analysis

CDOM is not measured by mass because it is not currently possible to extract all CDOM from the larger DOM pool. CDOM concentrations are evaluated by measuring a filtered sample's ability to absorb light over a spectrum of wavelengths assuming that organic substances within the sample are solely responsible for absorbing the light. For this study, CDOM measurements were made using a Perkin Elmer Lambda 35 dual beam scanning spectrophotometer. Absorbance measurements were conducted from 200-800 nm with a slit width of 1 nm and null corrected at 700 nm as described in Shank et al. (2005). Absorption coefficients (a_λ in units m^{-1}) were then calculated from absorbance values (A_λ) using the following equation: $a_\lambda = 2.303*A_\lambda L^{-1}$ where L is the light pathlength (m). CDOM is typically expressed as the magnitude of an absorption coefficient at 300 nm (a_{300}), 305 nm (a_{305}), or 350 nm (a_{350}). For this study, CDOM levels will be described by a_{305}. For reference, a_{305} values typically range from >100 m^{-1} in organic-rich blackwater systems to <0.2 m^{-1} in clear offshore ocean waters. Coastal waters typically have a_{305} values of 1-5 m^{-1}. Another parameter that is used to describe the characteristics of the CDOM pool is the spectral slope coefficient (*S*). The plot of the absorption coefficient as a function of wavelength for any CDOM sample can be described by the exponential equation:

$$a_\lambda = a_{\lambda o} e^{-S(\lambda - \lambda o)}$$

where *S* is termed the spectral slope coefficient. *S* values are typically smaller for freshwater CDOM than for marine CDOM so they can be used as indicators of the source of CDOM to a coastal system (Tzortziou et al. 2007).

Photochemical Experiments

CDOM photoreactivity was measured by exposing filtered water samples to simulated solar sunlight using an Atlas Suntest CPS solar simulator. The solar simulator produces approximately the same radiation spectrum that reaches the Earth's surface during summer months. Approximately 1 hour of exposure in the solar simulator is equal to 1 hour of summer noontime sunlight. For the photochemical experiments, filtered samples were placed in sealed quartz tubes and submerged in a temperature controlled water bath (25˚C). A dark control sample was wrapped thoroughly in aluminum foil and also placed in the solar simulator exposure bath. All quartz tubes were sealed with Teflon stoppers and Teflon tape to prevent leaking and contamination. Samples were collected at three different time points and total exposure times were either 24 or 48 hours. DOC and CDOM concentrations were determined as described above.

DOM Biological Availability Experiments

Two DOM bioavailability experiments were conducted on water samples collected at station E during the July sampling trip. The experiments were designed to 1) evaluate the bioavailability of Matagorda Bay DOM in the absence of sunlight and 2) evaluate whether solar radiation could produce biologically labile organic matter and stimulate bacterial activity in Matagorda Bay. For the first experiment, an aliquot (750 ml) of filtered Station E water was irradiated in the solar simulator for 48 hours as described above. Next, a series of samples/mixtures were placed in 500 ml flasks on stir plates in the dark and sampled for DOC and CDOM over a 4 day period. These samples/mixtures included:

A. 100% filtered and irradiated station E water
B. 100% unfiltered station E water (not irradiated)
C. 100% filtered non-irradiated station E water
D. 50% unfiltered station E water + 50% filtered non-irradiated station E water
E. 50% unfiltered station E water + 50% filtered and irradiated station E water

For the second experiment, the same setup was used with the exceptions that the experiment was allowed to run for 8 days and the unfiltered sample was collected fresh from the Aransas Pass Inlet (salinity = 25) to ensure a viable bacterial community. Filtered station E sample had been stored in the dark at 4 C since the time of collection.

Sonde Deployment

Dissolved oxygen and other water quality parameter measurements were measured at mid-depth using YSI series 6 multiparameter data sondes. Models 6920-S and 600XLM data sondes with 610-DM and 650 MDS display loggers were used. The series 6 parameters have the following accuracy and units: temperature (± 0.15^{o}C), pH (± 0.2 units), dissolved oxygen (mg l^{-1} ± 0.2), dissolved oxygen saturation (% ± 2%), specific conductivity (± 0.5% of reading depending on range), depth (± 0.2 m), and salinity (± 1% of reading or 0.1 ppt, whichever is greater). Salinity is automatically corrected to 25^{o}C.

Net Ecosystem Metabolism

Net ecosystem metabolism was calculated using open water diurnal methods (Russell et al. 2007, 2006). Dissolved oxygen concentrations were taken every 15 minutes and converted to a rate of change in dissolved oxygen concentration. These rates of change were then adjusted to control for diffusion of oxygen between the water column and the atmosphere by using percent saturation of dissolved oxygen in the water column and the wind dependent diffusion coefficient K (g O_2 m^{-2} h^{-1}) at 0% saturation proposed by D'Avanzo et al. (1996) using the equation:

$R_{dc} = R - ((1-((S_1 + S_2) / 200)) * K / 4)$; where

R_{dc} = diffusion corrected oxygen concentration rate of change per 15 minutes,

R = observed oxygen concentration rate of change,

S_1 and S_2 = dissolved oxygen percent saturations at time one and two respectively,

K = diffusion coefficient at 0% dissolved oxygen saturation.

To calculate daily net ecosystem metabolism the 15-minute diffusion corrected rates of dissolved oxygen change were then summed over a 24-hour period, starting and ending at 8 AM.

Statistical Analysis

Principal Component Analysis (PCA) was used to assess relationships between dissolved organic variables and hydrologic variables. PCA reduces a multivariate data set and creates new variables by extracting variance in order of importance. Results of the analysis are (1) new PC variables, which represent variable loading scores, and (2) a matrix of sample scores, which represent the sample contribution and allows for spatial and temporal comparisons. The higher the absolute values of the PC loading scores, the more influence the variable has in the new PC variable. Results are presented in plots as factor patterns and as loading scores; the plot of factor patterns allows for visualization of the similarities and variability among the factors, and the plot of loading scores allows for station comparisons.

RESULTS

CDOM/DOC concentrations at Lavaca and Matagorda Bay sampling sites

CDOM levels measured for the April 2007 sampling period ranged from 8-50 m^{-1}, with the highest levels measured in the upper reaches of Lavaca Bay at Stations A and FD (0 salinity) and the lowest values in the mesohaline waters of Matagorda Bay (Table 3.1). CDOM concentrations rose at Stations A and FD within the April sampling period (April 2 - April 5) from approximately 40 m^{-1} to 50 m^{-1} indicating that rising freshwater discharge was delivering increased quantities of CDOM to the system. Our sampling period also nicely captured the infiltration of organic-rich freshwater from the Colorado River to Matagorda Bay as CDOM levels at Station F more than doubled from April 2 to April 5 (Table 3.1).

Absorption coefficients (a_{305}) were greater in July as compared with April ranging from 15-77 m^{-1}, with the highest values once again measured at stations A and FD in Lavaca Bay. Higher CDOM levels in July likely result from a combination of factors that include higher CDOM production during the summer months in upland wetlands followed by increased delivery of organic-rich substances to the river flow during upland flooding.

Table 3.1. CDOM data and DOC concentrations for Lavaca and Matagorda Bay sampling sites. Abbreviations: a_{305} = absorption coefficient at 305 nm, and S = spectral slope coefficient for CDOM components.

				S (μm^{-1})			
Date	Station	Salinity	CDOM a_{305} (m^{-1})	290-320 nm	320-400 nm	290-400 nm	DOC (mg C/L)
2-Apr	Lock	0	32.18	14	16	16	6.70
2-Apr	A	0	37.35	14	17	16	7.19
2-Apr	FD	0	41.65	14	16	15	7.73
2-Apr	E	19	8.10	19	18	18	3.05
2-Apr	F	16	11.81	17	17	17	3.63
3-Apr	C	12	15.06	17	17	17	4.53
3-Apr	D	20	8.54	18	17	17	3.16
5-Apr	A	0	47.55	14	13	13	7.37
5-Apr	FD	0	50.48	14	15	14	8.21
5-Apr	F	4	28.76	15	16	16	6.58
23-Jul	A	0	62.33	13	15	14	9.03
23-Jul	FD	0	76.50	13	14	14	10.05
23-Jul	E	3	17.73	16	15	15	4.14
23-Jul	F	1	25.38	14	13	14	4.63
24-Jul	A	0	60.62	14	14	14	8.34
24-Jul	B	0	65.58	14	14	14	8.77
24-Jul	C	2	43.71	14	14	14	7.17
24-Jul	D	8	15.92	16	16	16	3.99
25-Jul	E	4	16.83	16	16	16	3.83
25-Jul	F	0	19.71	16	16	16	4.24
26-Jul	A	0	77.31	13	12	13	7.95
26-Jul	FD	0	74.02	13	14	13	8.82

There was not a strong correlation between a_{305} and salinity in the Lavaca and Matagorda Bay system during our sampling periods (Figure 3.2) only because of large variability at low salinities. Spectral slope coefficients calculated for absorption spectra from 290-320 nm ($S_{290\text{-}320}$) exhibited a strong relationship with salinity (Figure 3.3) indicating *S* values may be useful in determining freshwater vs. marine CDOM sources to the Lavaca-Matagorda system. Another interesting facet of the data in Table 3.1 is that industrial discharge from the Formosa Plant outfall located in close proximity to Station B appears to contribute measurable quantities of CDOM to Lavaca Bay as this site nearly always exhibited higher CDOM levels (in both April and July) than the upstream Station A site located closer to the Lavaca River mouth.

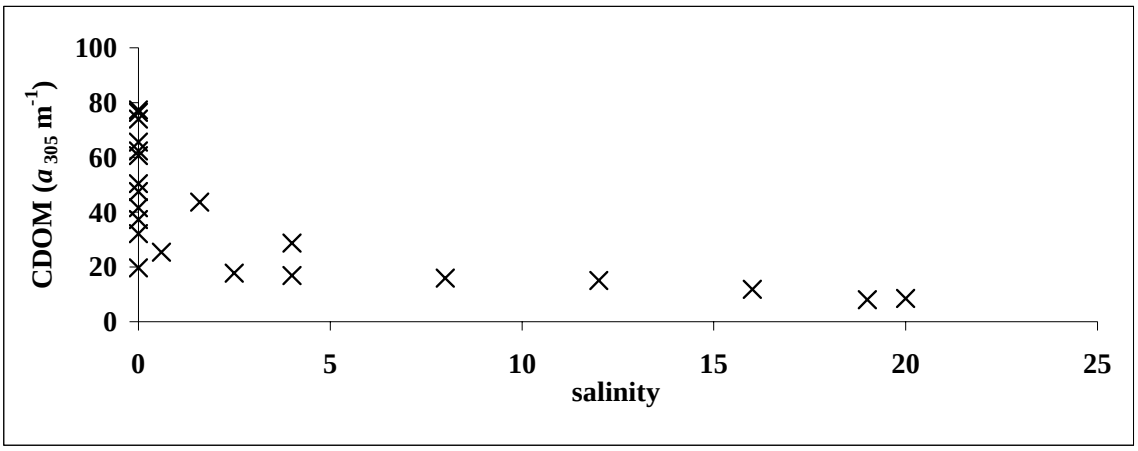


Figure 3.2. Plot of CDOM vs. salinity for all April and July 2007 samples.

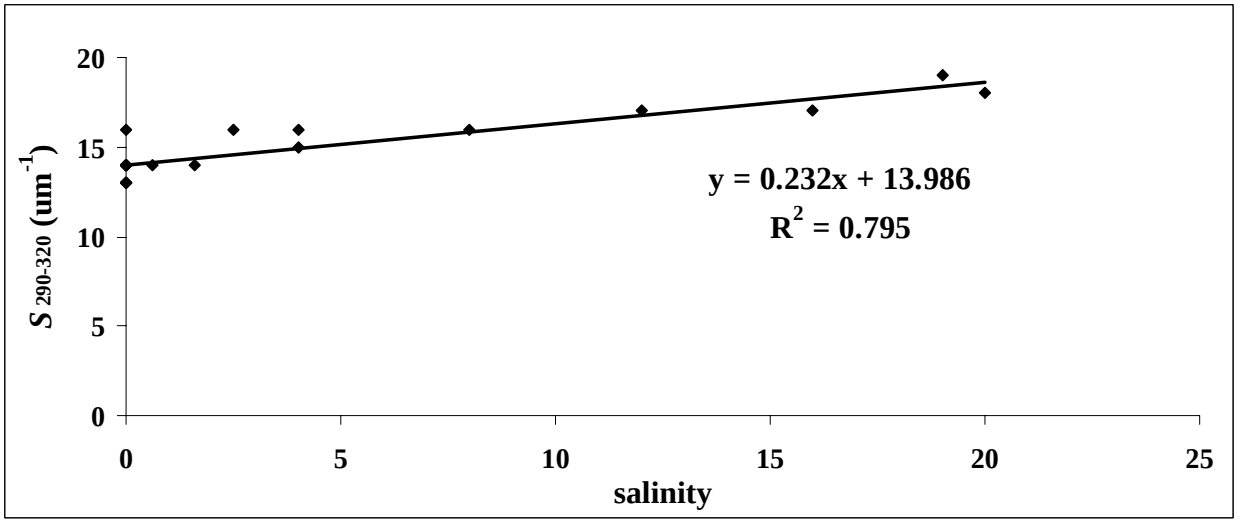


Figure 3.3. Plot of spectral slope coefficients (290-320 nm) vs. salinity for all April and July 2007 samples.

DOC concentrations (Table 3.1) ranged from 3-10 mg C/L with slightly higher values measured during July. DOC concentrations for April exhibited a very strong correlation with salinity (r^2=0.95), but when July data is included, this correlation decreases markedly (Figure

3.4). DOC and CDOM concentrations were very strongly correlated for both sampling sets (Figure 3.5). However, the slope of the linear regression of CDOM vs. DOC was much larger during July than April. As indicated earlier, higher relative CDOM levels in July likely resulted from increased CDOM production in freshwater wetlands during warmer summer months and the subsequent release of this CDOM to the rivers during the large scale flooding of organic-rich inland river banks that occurred just prior to our sampling trip.

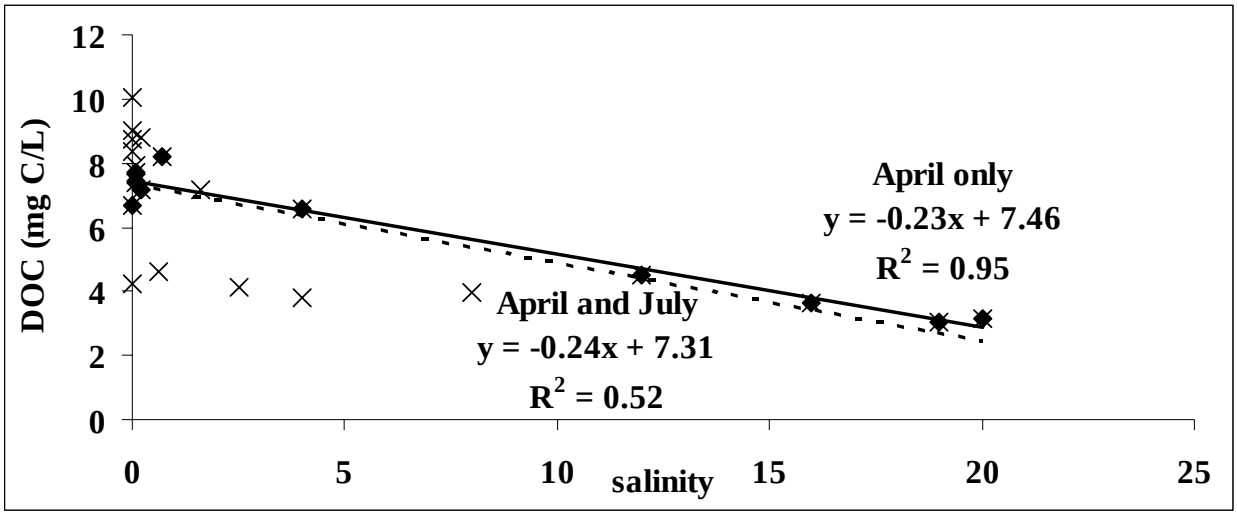


Figure 3.4. Plot of DOC concentrations vs. salinity for April (♦) and July (X) 2007 samples.

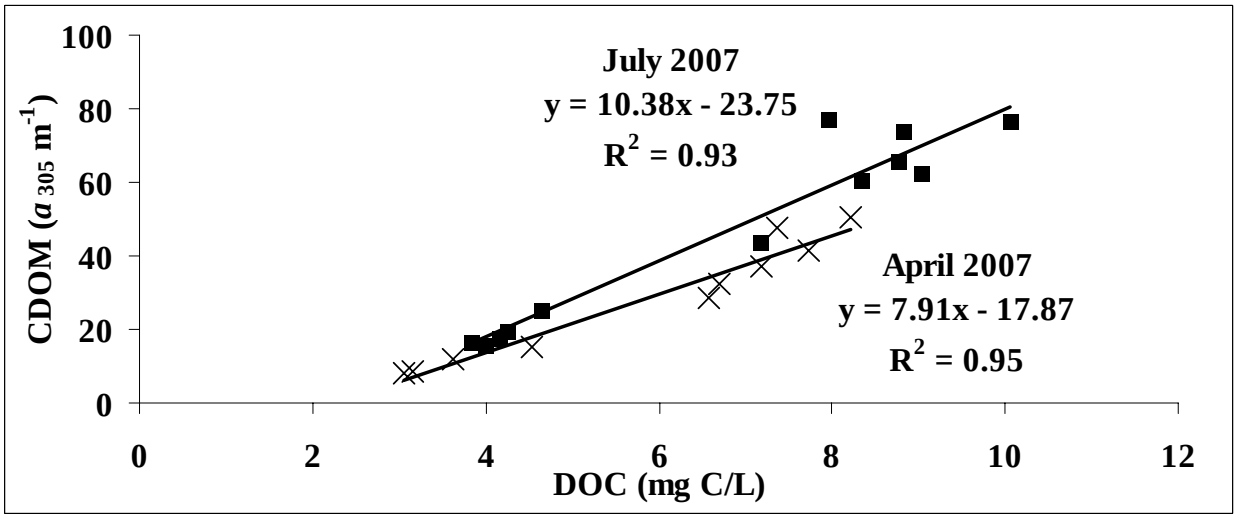


Figure 3.5. Plot of CDOM vs. DOC concentrations for April (■) and July (X)2007 samples.

CDOM Photodegradation Experiments

Photobleaching refers to the loss of CDOM (i.e., decrease in *a*) that occurs when natural waters (or water samples) absorb solar radiation. For this study, photobleaching rates were measured on filtered water samples placed in a solar simulator for either 24 or 48 hours. During April (Table 3.2 and Figure 3.6), a_{305} loss rates within the Lavaca-Matagorda Bay boundaries were very similar and ranged from 0.014 hr^{-1} to 0.021 hr^{-1}, with the highest value

measured for a Station C sample. A sample taken just below the Locks on the Colorado River was especially photoreactive with a photobleaching rate of 0.028 hr^{-1}. Using measured photo-decay rates, it is possible to calculate the photobleaching half-life for a water sample, or the length of time required for 50% of the CDOM to be degraded during exposure. For the April Matagorda Bay samples, half-lives ranged from 33-50 hours. Half-lives were shortest in upper Lavaca Bay indicating this material is slightly more photoreactive than CDOM in lower portions of Matagorda Bay. Assuming approximately 8 hours of equivalent sunlight each day during this time of year, the data in Table 3.2 indicate that 50% of the CDOM in upper surface waters will be degraded in approximately 4-6 days. There were no significant changes to the spectral slope coefficients during exposures. It has been reported that *S* values should increase during exposure to solar radiation due to the breakdown of higher molecular weight CDOM components that absorb longer wavelength light (Del Vecchio and Blough 2002), but this was not evident for our experiments.

Photobleaching rates for the July samples (Table 3.3 and Figure 3.7) were nearly equivalent to those measured during April ranging from 0.015-0.020 hr^{-1}, corresponding to photobleaching half-lives of 35-46 hrs. However, in contrast to April, the shortest half-lives were measured for Matagorda Bay Stations E (35 hrs) and F (37 hrs). Salinities were much lower throughout Matagorda Bay in July suggesting that the organic material had been very recently discharged to the system. But, it is unclear why CDOM in Matagorda Bay (Stations E and F) would be more photoreactive than CDOM in upper Lavaca Bay (Stations A and B) unless there were slight differences in source material. As approximated for April, the time required for 50% of the CDOM to be lost from surface waters during July in the Lavaca-Matagorda Bay system is 4-6 days.

DOC loss by photobleaching of unfiltered Station E water mixed with filtered and irradiated Station E water was at a rate of 0.035 hr^{-1} (Figure 3.8). Loss rates fro Station E water were similar 0.031 hr^{-1} when diluted with Gulf of Mexico unfiltered water from Aransas Pass inlet (Figure 3.9).

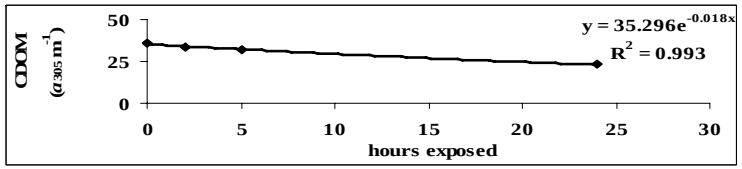
DOC concentrations were also measured during photobleaching experiments for April and July samples (Tables 3.4 and 3.5). However, DOC levels remained virtually constant even though CDOM levels typically decreased by 30-40%. This data gives clear indication that the DOM pool is not being destroyed upon absorption of solar radiation, but rather its structure is being chemically altered.

Table 3.2. Photobleaching data for April 2007 samples

Date	Station	Salinity	hours exposed	CDOM a_{305} (m^{-1})	S (μm^{-1}) 290-320 nm	S (μm^{-1}) 320-400 nm	S (μm^{-1}) 290-400 nm	CDOM a_{305} decay rate (hr^{-1})	CDOM a_{305} half life (hrs)	DOC (mg C/L)
2-Apr	FD	0	0	39.43	14	16	16	0.018	39	7.01
			2	36.48	14	15	15			7.25
			5	34.11	14	15	15			7.23
			24	24.78	15	15	15			7.07
			dark	39.41	14	16	16			7.40
2-Apr	Lock	0	0	31.71	14	16	16	0.028	25	6.11
			2	29.00	14	15	15			6.29
			5	26.42	14	15	15			6.63
			24	15.84	16	16	15			6.03
			dark	31.35	14	16	16			6.10
2-Apr	A	0	0	36.09	14	16	16	0.018	39	6.92
			2	33.58	14	15	15			6.95
			5	31.92	14	15	15			7.14
			24	23.09	15	16	15			6.89
			dark	36.24	14	16	16			6.97
3-Apr	A	0	0	33.72	15	17	16	0.016	43	7.13
			2	32.10	14	15	15			7.08
			5	31.33	14	16	15			7.08
			24	22.90	15	16	15			6.72
			dark	34.32	14	16	15			7.20
2-Apr	E	19	0	7.92	19	18	18	0.014	50	2.9
			5	6.96	18	18	18			3.2
			24	4.94	19	18	18			3.0
			48	3.93	19	18	18			3.1
			dark	8.46	19	18	19			4.5

3-Apr	C	12	0	15.09	17	17	17	0.021	33	4.6
			2	14.27	16	17	17			4.5
			5	12.76	16	18	17			4.7
			24	8.93	17	17	17			4.2
			dark	14.86	18	18	18			4.8
2-Apr	F	16	0	11.28	17	17	17	0.015	46	3.5
			5	9.58	17	18	17			3.4
			24	6.86	18	17	18			3.7
			48	5.42	18	17	17			3.4
			dark	11.21	18	18	18			3.5
3-Apr	B	1	0	34.28	14	16	16	0.018	39	6.7
			2	31.14	15	17	16			7.1
			5	27.82	15	17	16			6.9
			24	21.47	16	16	16			6.5
			dark	33.52	15	17	16			6.7
3-Apr	D	20	0	7.59	18	18	18	0.016	43	2.7
			5	7.18	17	18	18			2.7
			24	5.24	18	18	18			2.8
			48	3.57	20	19	20			4.7
			dark	8.20	19	18	18			3.1

2-April Station A



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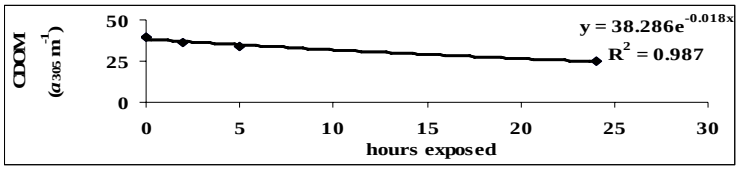

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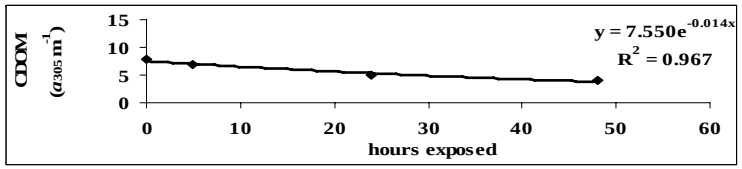

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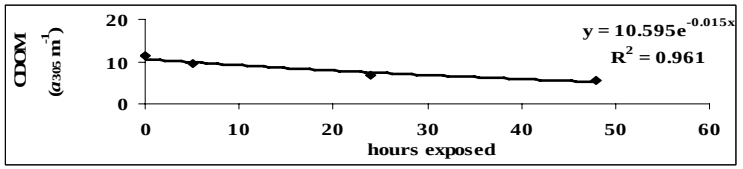
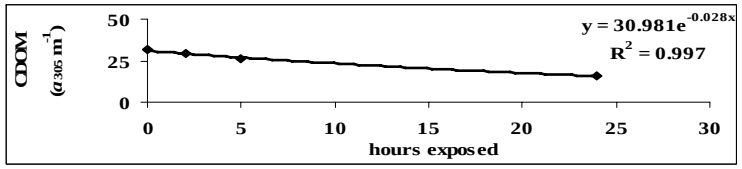
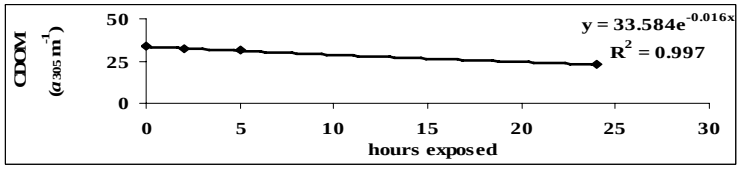


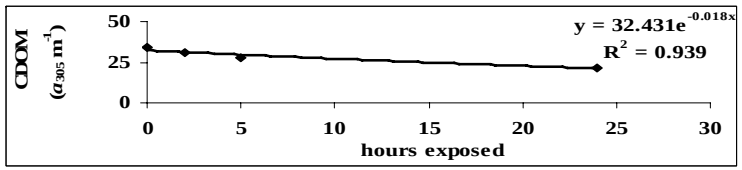
2-April Colorado River Lock



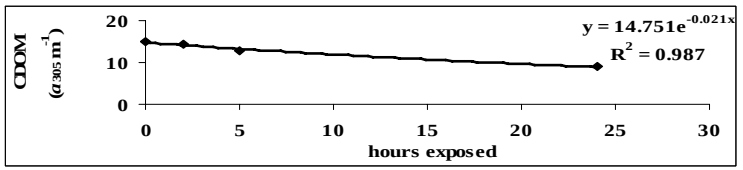
3-April Station A



3-April Station B



3-April Station C



3-April Station D

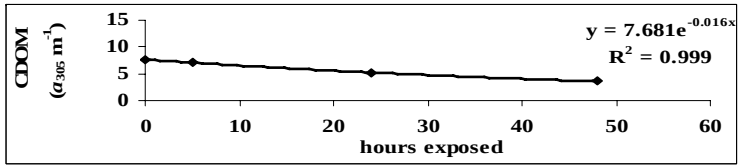
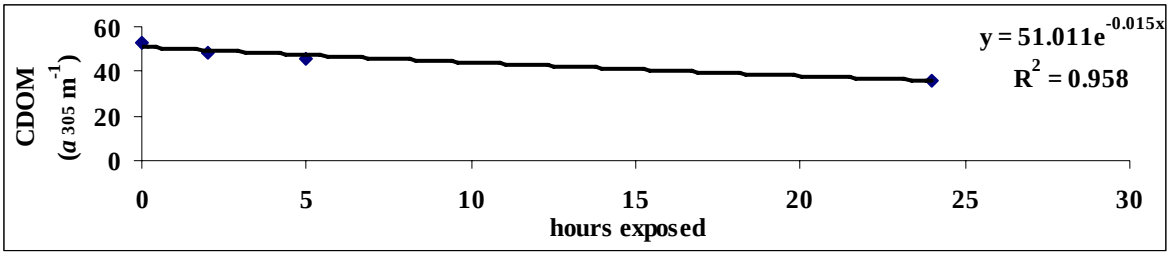


Figure 3.6. Results of photodegradation experiments for April 2007 samples.

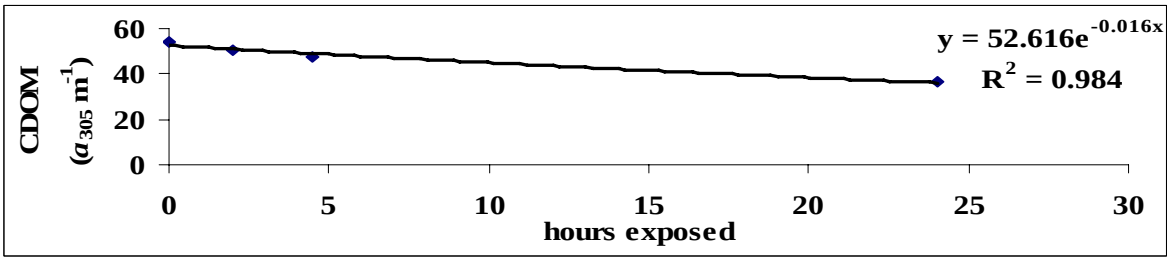
Table 3.3. Photobleaching data for July 2007 samples.

Date	Site	Salinity	hours exposed	CDOM a_{305} (m^{-1})	S (μm^{-1}) 290-320 nm	S (μm^{-1}) 320-400 nm	S (μm^{-1}) 290-400 nm	CDOM a_{305} decay rate (hr^{-1})	CDOM a_{305} half life (hrs)	DOC (mg C/L)
23-Jul	A	0	0	53.26	14	16	16	0.015	46	9.36
			2	48.69	14	16	15			9.43
			5	45.79	14	16	15			9.55
			24	36.16	15	16	15			9.23
			dark	52.44	14	16	16			9.39
23-Jul	E	3	0	16.45	17	17	17	0.020	35	4.09
			2	15.19	16	18	17			4.20
			5	(analysis error)						4.10
			24	10.07	17	18	18			4.06
			dark	16.58	17	17	17			4.15
23-Jul	F	1	0	18.93	16	17	17	0.019	37	4.69
			2	18.13	15	17	16			4.65
			5	(analysis error)						4.69
			24	12.12	17	17	17			4.60
			dark	19.08	16	17	17			4.53
23-Jul	FD	0	0	54.11	14	16	15	0.016	44	8.75
			2	50.49	14	16	15			9.28
			4.5	47.89	14	16	15			9.29
			24	36.30	15	16	16			8.93
			dark	52.79	14	16	16			8.96

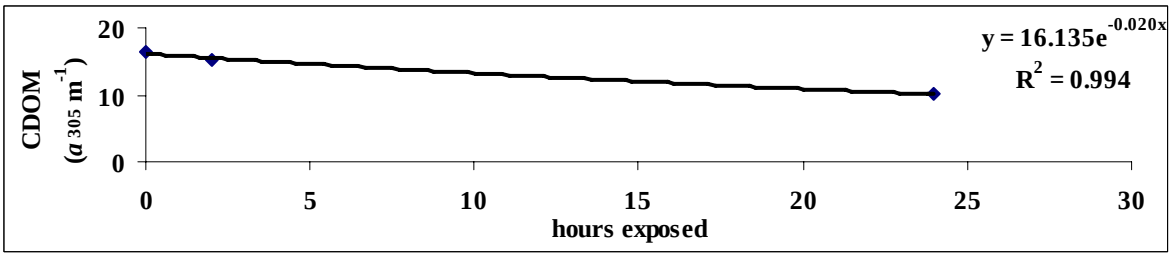
23-July Station A



23-July Station FD



23-July Station E



23-July Station F

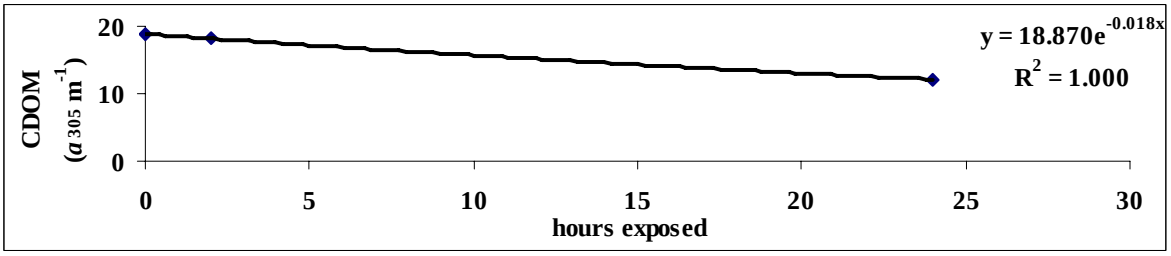


Figure 3.7. Results of photodegradation experiments for July 2007 samples.

Table 3.4. Results of biological availability experiment #1 using filtered, unfiltered, and filtered + irradiated Station E water (collected 7/25/07).

Sample	Treatment hours	CDOM a_{305} (m^{-1})	S (μm^{-1}) 290-320 nm	S (μm^{-1}) 320-400 nm	S (μm^{-1}) 290-400 nm	DOC (mg C/L)
A	0	6.05	17	18	18	3.27
irradiated	24	6.33	17	17	17	3.18
	48	6.09	17	17	17	3.17
	96	6.07	17	18	18	3.35
FD	0	14.80	16	17	17	4.16
unfiltered	24	15.57	16	16	16	4.11
	48	15.32	16	17	17	4.01
	96	15.02	17	17	17	4.05
C	0	11.86	16	17	17	3.28
filtered	24	12.03	17	17	17	3.23
	48	11.69	16	17	17	3.16
	96	11.68	17	17	17	3.25
D	0	13.68	16	17	17	3.70
50% unf. + 50% filt.	24	13.58	17	17	17	3.68
	48	13.38	16	17	17	3.52
	96	13.34	17	17	17	3.72
E	0	11.11	17	17	17	3.57
50% unf. + 50% irr.	24	10.65	17	17	17	3.43
	48	10.55	17	17	17	3.34
	96	10.53	17	17	17	3.27

Table 3.5. Results of biological availability experiment #2 using unfiltered Aransas Pass tidal inlet water (salinity = 25) and filtered and filtered + irradiated Station E water (collected 7/25/07).

Sample	Treatment hours	CDOM a_{305} (m^{-1})	S (μm^{-1}) 290-320 nm	320-400 nm	290-400 nm	DOC (mg C/L)
A	0	5.69	17	17	17	3.24
irradiated	48	5.69	17	18	18	3.30
	120	5.72	17	18	18	3.33
	192	5.73	17	18	18	3.32
FD	0	5.16	22	18	19	2.99
unfiltered	48	5.01	21	18	19	2.85
	120	4.97	21	18	19	2.82
	192	4.92	22	18	19	2.80
C	0	11.92	16	17	17	3.41
filtered	48	11.86	16	17	17	3.46
	120	11.82	16	17	17	3.45
	192	11.60	16	17	17	3.45
D	0	8.43	18	17	17	3.18
50% unf. + 50% filt.	48	8.39	18	18	18	3.17
	120	8.23	18	18	18	3.20
	192	8.21	18	17	17	3.08
E	0	5.35	20	18	18	3.20
50% unf. + 50% irr.	48	5.25	19	18	18	2.93
	120	4.96	19	18	18	2.76
	192	4.90	19	18	18	2.70

DOM Biological Availability Experiments

For the first DOM bioavailability experiment, a combination of filtered irradiated (48 hours in solar simulator), filtered unirradiated, and unfiltered unirradiated Station E Matagorda Bay samples were incubated in the dark for a 4 day period. Results presented in Table 3.6 indicate that DOC levels remained virtually constant over the course of the experiment for samples A-D. However, DOC levels showed a consistent downward trend falling from 3.57 mg C/L to 3.27 mg C/L for sample E that contained mixture of filtered irradiated and unfiltered water. This pattern was also observed during the second 8 day DOM bioavailability experiment when the unfiltered sample was freshly collected Aransas Pass tidal inlet water. For this second experiment (Table 3.5), DOC levels in samples A-D also changed very little, although it could be argued that DOC levels dropped slightly (6%) in sample B (unfiltered only), while a DOC loss of 16% (3.20 to 2.70 mg C/L) was observed in the mixture of filtered irradiated and unfiltered water. The rate of DOC loss for the filtered irradiated and unfiltered mixture in both experiments was 0.001 hr^{-1}. Results from both experiments strongly suggest while much of the dissolved organic material that entered the Lavaca-Matagorda Bay system during our study was largely unavailable to the ambient microbial population initially, production of labile compounds during CDOM photochemical reactions within Matagorda estuarine boundaries can substantially stimulate bacterial respiration.

Table 3.6. Results from 24-hour sonde deployments. Net ecosystem metabolism (NEM) calculated over the day in units of mg O_2 m^{-2} d^{-1}, and mean ± standard deviation for temperature, dissolved oxygen (DO), salinity, pH, and sample depth.

Date	Station	NEM	Temp. (°C)	DO (mg/L)	Salinity (ppt)	pH	Depth (m)
04/03/2007	A	-2.94	23.94 ± 0.46	7.56 ± 0.12	0.67 ± 0.25	7.79 ± 0.03	0.70 ± 0.08
	F rep 1	-4.67	24.42 ± 0.45	6.34 ± 0.67	14.11 ± 1.47	7.84 ± 0.07	0.79 ± 0.08
	F rep 2	-4.34	24.40 ± 0.49	6.46 ± 0.74	14.20 ± 1.02	7.94 ± 0.07	0.77 ± 0.08
	FD rep 1	-3.40	23.90 ± 0.70	7.12 ± 0.62	0.54 ± 0.23	7.75 ± 0.10	0.45 ± 0.08
	FD rep 2	-3.39	23.86 ± 0.66	7.17 ± 0.62	0.54 ± 0.23	7.77 ± 0.10	0.50 ± 0.08
07/24/2007	A	-3.18	28.93 ± 0.50	6.46 ± 0.19	0.17 ± 0.06	7.57 ± 0.06	0.51 ± 0.08
	E	4.32	29.39 ± 0.52	9.17 ± 1.50	5.19 ± 0.91	8.45 ± 0.13	1.25 ± 0.10
	F	-2.17	28.59 ± 0.61	6.91 ± 0.23	0.70 ± 0.38	7.90 ± 0.07	0.74 ± 0.12
	FD rep 1	-.300	28.80 ± 0.64	6.56 ± 0.14	0.11 ± 0.02	7.44 ± 0.04	0.35 ± 0.08
	FD rep 2	-3.32	28.68 ± 0.65	6.42 ± 0.14	0.11 ± 0.02	7.51 ± 0.06	0.36 ± 0.09

Net Ecosystem Metabolism

The average value of net ecosystem metabolism (NEM) for stations A, F and FD was -3.6 mg O_2 m^{-2} d^{-1}in April and -2.8 mg O_2 m^{-2} d^{-1} in July (Table 3.6). The negative values indicate the ecosystem was net heterotrophic near the river sources. In contrast, station E in July had had a positive (i.e., autotrophic) value of 4.3 mg O_2 m^{-2} d^{-1}. Consistent with autotrophy, Station E was had the highest average DO and pH. Station E was also was the warmest, the deepest, and closest to the Gulf of Mexico. Excluding station E, NEM increased with

increasing DOC, which indicates there was a greater accumulation of DIC when respiration rates were lower and less heterotrophic conditions prevailed (Figure 3.10).

Hydrography

Analysis of the hydrographic measures and the organic measures created a principal component axis where the concentrations of CDOM and DOC were highly correlated to one another, but were inversely correlated with salinity, sampling depth, and pH (Figure 3.11A). This indicates that as freshwater enters the bay, it brings in organic matter and dilutes salinity. Chlorophyll *a* concentrations were placed along the second principal components axis and are thus not correlated to the first axis containing organic matter and salinity. The stations separated base on those in Lavaca Bay (A, B, and FD) and those in Matagorda Bay (C, D, E, and F) (Figure 3.11B).

DISCUSSION

To quantify the potential importance of CDOM photochemistry to bacterial respiration within the Matagorda Bay system, an important first step is to quantify the magnitude of CDOM photoreactivity that may occur throughout the whole water column. Photobleaching experiments used in this study only predict the loss of CDOM in the very top of the water column because solar radiation capable of driving CDOM photochemistry attenuates with depth in natural waters. To calculate an average CDOM photobleaching rate (k_{avg}) for the entire water column, the following equation must be used (Hu et al. 2002):

$$k_{avg} = k(1\text{-}e^{-Kd*z})/(K_d*z)$$

where K_d (m^{-1}) is the diffuse attenuation coefficient of solar radiation representing the loss of light with depth and z is the depth of the water column. Since most CDOM photobleaching is caused by absorption of UV (Miller et al. 2002), K_d should represent the loss of the entire UV spectrum through the water column. For turbid estuarine waters like those experienced in Matagorda Bay during this study, K_d can be approximated at 30 m^{-1} (Shank et al. 2006). Data provided in Tables 3.2 and 3.3 show that measured photobleaching rates (k) for station E water ranged from 0.014 to 0.020 hr^{-1} for our experiments, so 0.017 hr^{-1} was chosen as the mean photobleaching rate. If the average depth is assumed to be 2 m throughout the Lavaca-Matagorda Bay system, the average photobleaching rate (k_{avg}) for the entire water column will be 0.0003 hr^{-1}. Using this k_{avg}, the loss of CDOM due to photobleaching throughout the water column can then be calculated using the following equation:

$$a=a_o e^{-kavg*t}$$

where a_o is the initial CDOM level and t is the time of exposure. Using this equation, the magnitude of photobleaching can be estimated for various residence times within the estuary. For a residence time of 60 days assuming 8 hours per day of sun (480 total hours of sunlight), the loss of CDOM would be approximately 15% throughout the water column. If the residence time was increased to 120 days (960 total hours of sunlight), then the loss of CDOM would be approximately 25% throughout the water column. Therefore, when the waters are turbid as experienced in this study, the overall magnitude of CDOM photobleaching is likely to be small. However, if the suspended sediment load diminished and the K_d value then decreased to 10 m^{-1} within Matagorda Bay waters because of increased

light availability through the water column (10 m^{-1} may be an overestimate of K_d during low flow), k_{avg} would increase to 0.0009 hr^{-1}, and the magnitude of CDOM loss would increase to 35% for 2 months residence time and 58% over 4 months. As shown in the DOM biological availability experiments, significant production of biologically labile compounds can occur when CDOM is reduced by 50%. Moran et al. (2000) also reported that significant bacterial respiration of DOM can occur when waters are 50% photobleached and that the production of labile DOM increases with the magnitude of CDOM photobleaching.

Provided freshwater discharge is an important source of DOM and CDOM to Lavaca Bay and Matagorda Bay and the photoreactivity of this allochthonous CDOM remains consistent (as can be inferred by consistent photobleaching rates measured for samples collected in both April and July), then the impact of the production of labile organic compounds from CDOM photochemical reactions on bacterial respiration is likely to be very important during the summer months. Our experiments showed that the addition of irradiated water to unfiltered Station E water stimulated bacterial processing of organic matter causing a decrease in DOC levels of 10 and 16% over 4 and 8 day periods, respectively. By comparison, Moran et al. (2000) reported a net DOC respiration of approximately 25% over the course of a 51-day incubation for an organic rich Satilla River sample that had been 44% photobleached. It is important to note that the potential stimulation of bacterial respiration in Lavaca Bay and Matagorda Bay could be much greater than what was measured in our experiments because the concentrations of labile organic compounds and bacterial population were both reduced by half during sample mixing. If the production rate of bioavailable DOM equals the CDOM photobleaching rate for relatively clear waters (k_{avg} = 0.0009 hr^{-1}), this rate would be nearly equivalent to the DOC respiration rate measured during the biological availability studies (0.001 hr^{-1}) indicating that these processes could be closely coupled in Matagorda Bay waters. Provided CDOM photobleaching supplies a sufficient source of DOM for microbial respiration during the summer as suggested by results from this study, we estimate that 50% of the freshwater derived DOM could be exhausted by bacterial respiration solely because of its photoreactive nature.

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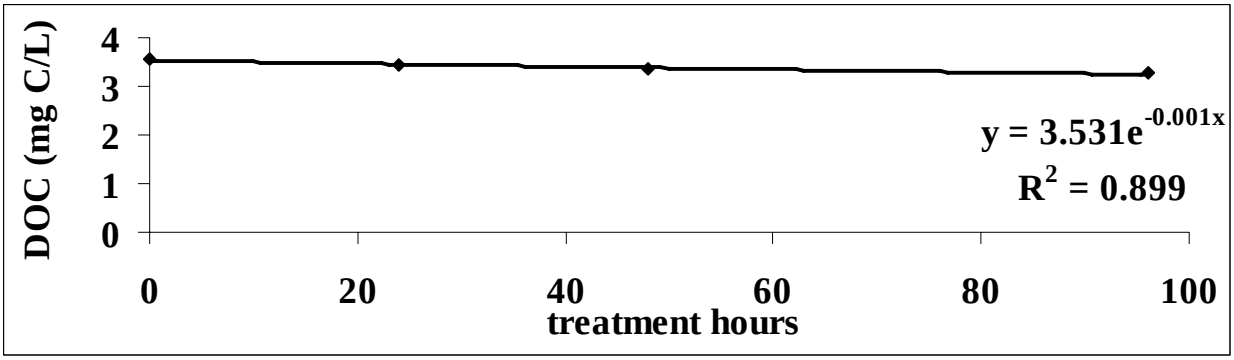

Figure 3.8. DOC loss in unfiltered Station E water mixed with filtered and irradiated Station E water.

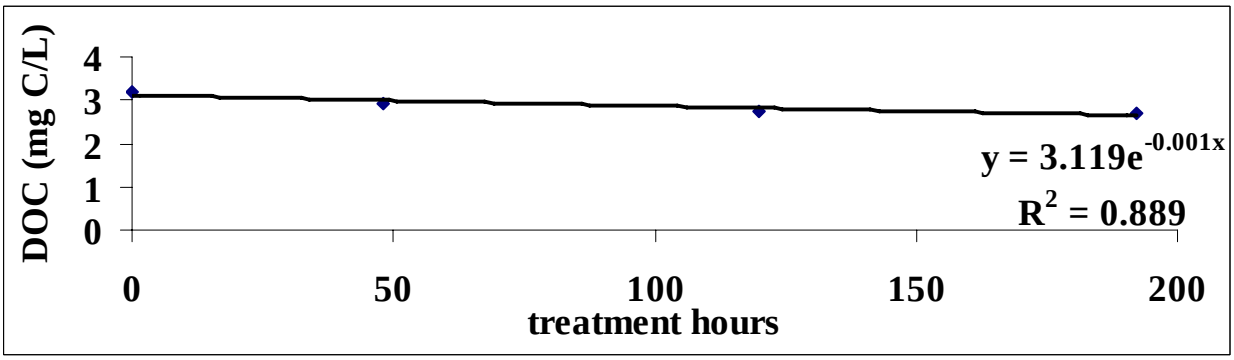

Figure 3.9. DOC loss in unfiltered Aransas Pass inlet water mixed with filtered and irradiated Station E water.

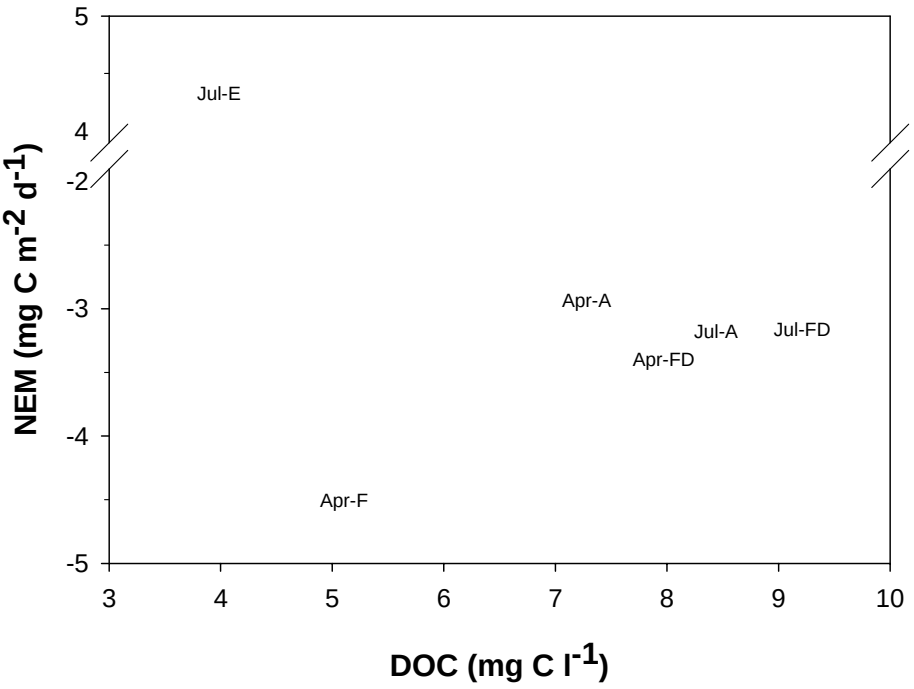

Figure 3.10. Relationship between net ecosystem metabolism (NEM, Table 3.6) and dissolved organic carbon (DOC, Table 3.1). Note the NEM axis is split.

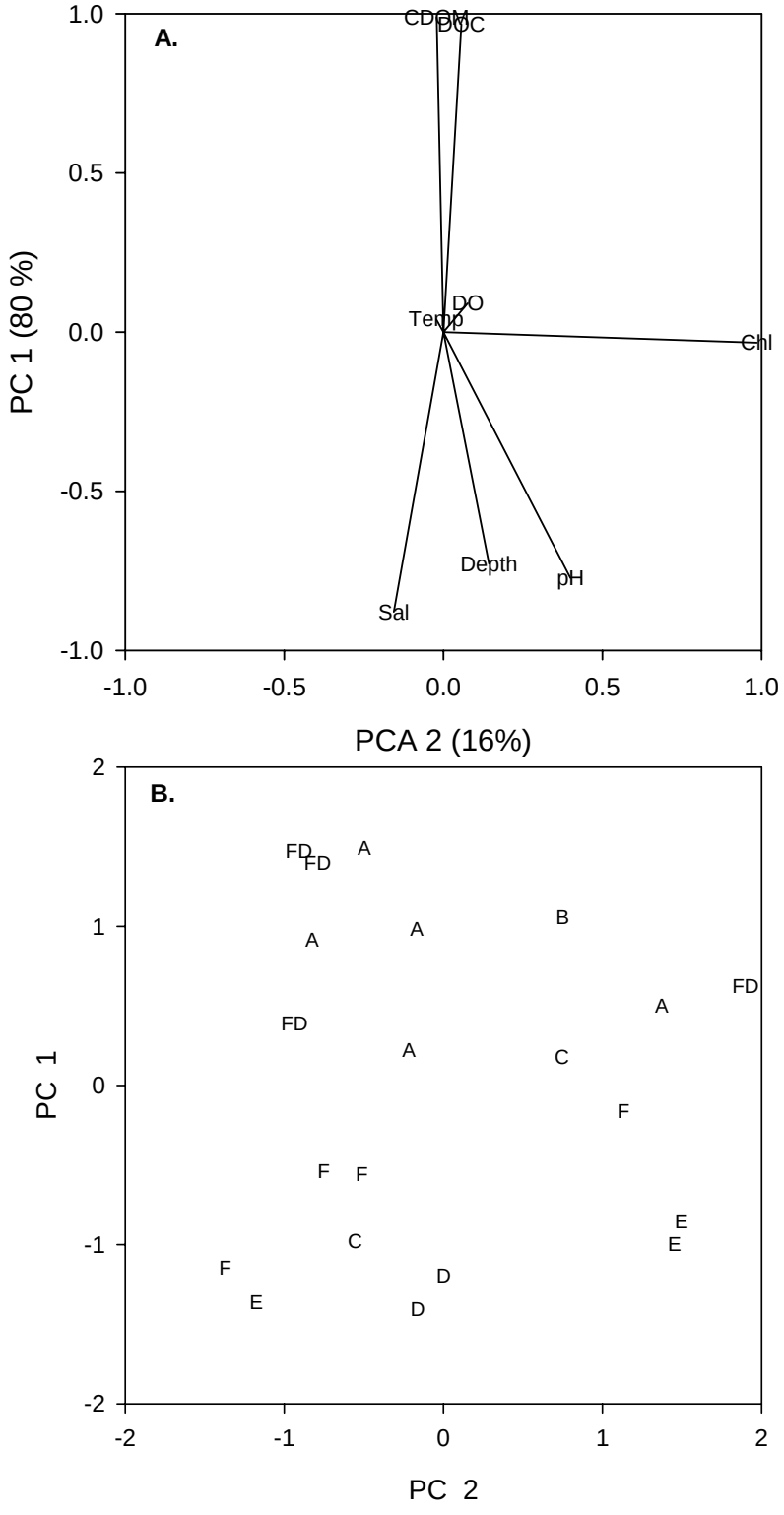

Figure 3.11. Principal components (PC) analysis of organic chemical and hydrographic measurements. A) Variable loads. B) Station scores.