FINAL REPORT

Effect of Climatic Variability on Freshwater Inflow, Benthic Communities, and Secondary Production in Texas Lagoonal Estuaries: FY 2007

To:

Texas Water Development Board

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1. Introduction

The ecology of estuaries is strongly influenced by the quantity, timing, frequency, and duration of freshwater pulses to coastal ecosystems (Montagna et al. 2002). In Texas, there is a strong climatic gradient with decreasing precipitation, and concomitant freshwater inflow, from northeast to southwest (Montagna et al. 2007). Along this gradient, rainfall decreases by a factor of two, but inflow balance decreases by almost two orders of magnitude. Inflow balance is the sum of freshwater inputs (gaged, modeled runoff, direct precipitation, plus return flows) minus the outputs (diversions and evaporation). The net effect is a gradient with estuaries with similar physical characteristics but a declining salinity gradient (Montagna et al. 2009).

Another characteristic of Texas estuaries is the extreme year-to-year variability of precipitation, inflow, and salinity. This climatic variability is caused by the El Niño Southern Oscillation (ENSO) (Tolan 2007). The ENSO climate signals are correlated to salinity structure within Texas estuaries within 4 to 6 months. During El Niño events, salinities in Texas estuaries decrease because of increased freshwater flows to the coasts. During La Niña periods, salinities increase because of the drier climatic conditions. These cycles occur with a periodicity of 3.55, 5.33, and 10.67 years. The ENSO is dominated by the 3.55- and 5.33-year periods and the 10.67-year period is defined by the Pacific Decadal Oscillation. The combination of latitudinal and long-term climate differences (which drive inflow variability) and the varying geography and tidal dynamics of the estuaries is responsible for the uniqueness, or estuarine signature, common in estuaries throughout the world. Thus, the differences in inflow regimes among the Texas estuaries are driven by the combination of spatial and temporal climatic regime shifts.

The climate regime differences in Texas estuaries drives biological processes (objective of this project is to link long term data bases on temperature, freshwater inflow (which is driven by precipitation and runoff), and macrofauna communities to predict how changes in climatic variability influence the structure and function of estuarine communities. The approach is to study benthos and inflow dynamics over a large regional scale for the long term to capture spatial and long term effects. Benthos are good indicators of environmental change, because they are fixed spatially, have relatively long life cycles, and are at the bottom of the food chain, thus integrating long term changes in the overlying water column (Montagna and Kalke 1995).

Benthic data has been collected in the Coastal Bend estuaries from 1984 to 2000. At each quarterly sampling period, nutrient concentrations, primary producer biomass, and benthos biomass has been measured. The ecological model is mechanistic and calculates bay-wide productivity for two different trophic groups of benthos: those relying on the water column for food e.g., filter and interface feeders, and those relying on sediments for food, e.g., the deposit feeders. The model was first developed in 1995 (Montagna and Li 1996) and calibrated using data from 1990 to 1995. Recently, the model was further developed for a test case in Matagorda Bay (Montagna 2007). In the current project, the model has been refined and will be recalibrated using a 10-year dataset from 1990 to 2000 and validated with data from 2000 – 2005. The project will completed over a 2-year period from FY 2007 to FY 2008. The current report is on activities accomplished during the first period: FY 2007, which includes data assimilation and analysis, and model development.

2. Methods

2.1 Databases

2.1.1. Benthic macrofauna

Benthic data is used to calibrate the model. Long-term macrobenthos data were used to perform a modeling experiment to determine the effects of alterations in FWI. 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). During each sampling event, three independent macrobenthos samples were

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collected using 6.7-cm diameter sediment cores (35.4 cm^2 area) to a depth of 10 cm and preserved with 5% buffered formalin (Montagna and Kalke, 1992). Samples were sorted, and biomass was measured by major taxa. Concurrent hydrographic measurements were also made during each sampling period, and these included: chlorophyll *a*, nutrient concentrations, dissolved oxygen, salinity, temperature, and water depth. Once each year, in October, sediment grain size, total nitrogen, and organic carbon content are measured in sediments.

Four estuaries were studied in the South Texas Coastal Bend. These estuaries are are the Lavaca-Colorado (LCE), Guadalupe (GE), Nueces (NE) and Laguna Madre Estuaries (LME) (Fig. 1). The estuaries lie in a climatic gradient from wetter (LCE and GE) to drier (NE and LME). Within each estuary, stations were located in either the primary bay near the connection with the Gulf of Mexico, or the secondary bay near the freshwater inflow source. This enabled modeling of both the primary and secondary bays in each estuary. A 18-year (1988 – 2006) data set was pooled from each estuary for the comparison of simulation results (Table 1).

2.1.2. Predators

Predator data is used as a loss term in the model. Fisheries data from 1988 – present were obtained from Texas Parks and Wildlife Department (TPWD). The Coastal Fisheries Division samples monthly in the four estuaries (LCE, GE, NE, and LME) using an otter trawl and bag seine. A study of stable isotopes and mercury bioaccumulation in different food chains determined that black drum, red drum, and blue crab are the main predators on benthic infauna (Montagna, unpublished data). There was no indication that predators selected any trophic level over the other, so it is assumed that predation rates on both trophic levels are equal. Therefore, the average value for density of each of these three main predators during each sampling period was used.

2.1.3. Other environmental variables

Primary production was used as input to the model. However, there is no empirical primary production data in each bay to form a time series over the entire modeled period. Therefore, we parameterized primary production as a function of

limiting factors (day length, temperature and nutrient concentration) from the historical maximum value. Previous studies report a range from 0.5 to 5 g C m^{-2} d^{-1} in the Lavaca-Colorado Estuary (Armstrong, 1985; Stockwell, 1989; Brock, 1994). To derive a forcing function which can simulate day length, a dataset of monthly day length for the Texas coastal area was obtained from Tony Amos, University of Texas Marine Science Institute.

Table 1. Sampling estuaries, bay type and names, stations, and periods for the continuous long-term database in the Lavaca-Colorado Estuary. [*]Environmental/biological variables used in the model were selected from the same database.

Estuary	Bay Type	Bay Name	Stations	Sampling Period
Lavaca-Colorado	Secondary	Lavaca	A	1984 – 2007
	Secondary	Lavaca	B	1988 – 2007
	Primary	Matagorda	C, D	1988 – 2007
	Lagoon	East Matagorda	E, F	1993 – 2007
Guadalupe	Secondary	Upper San Antonio	A, B	1987 – 2007
	Primary	Lower San Antonio	C, D	1987 – 2007
Nueces	Secondary	Nueces	A, B	1988 – 2007
	Primary	Corpus Christi	C, D	1988 – 2007
	Primary	Corpus Christi	E	1988 – 2007
Laguna Madre	Secondary	Baffin Bay	6, 24	1988 – 1997
	Primary	Laguna Madre	189G, 189S	1988 – 1997

[*]Environmental/biological variables include temperature, salinity, nutrients (N, P, Si), predator density and benthos biomass.
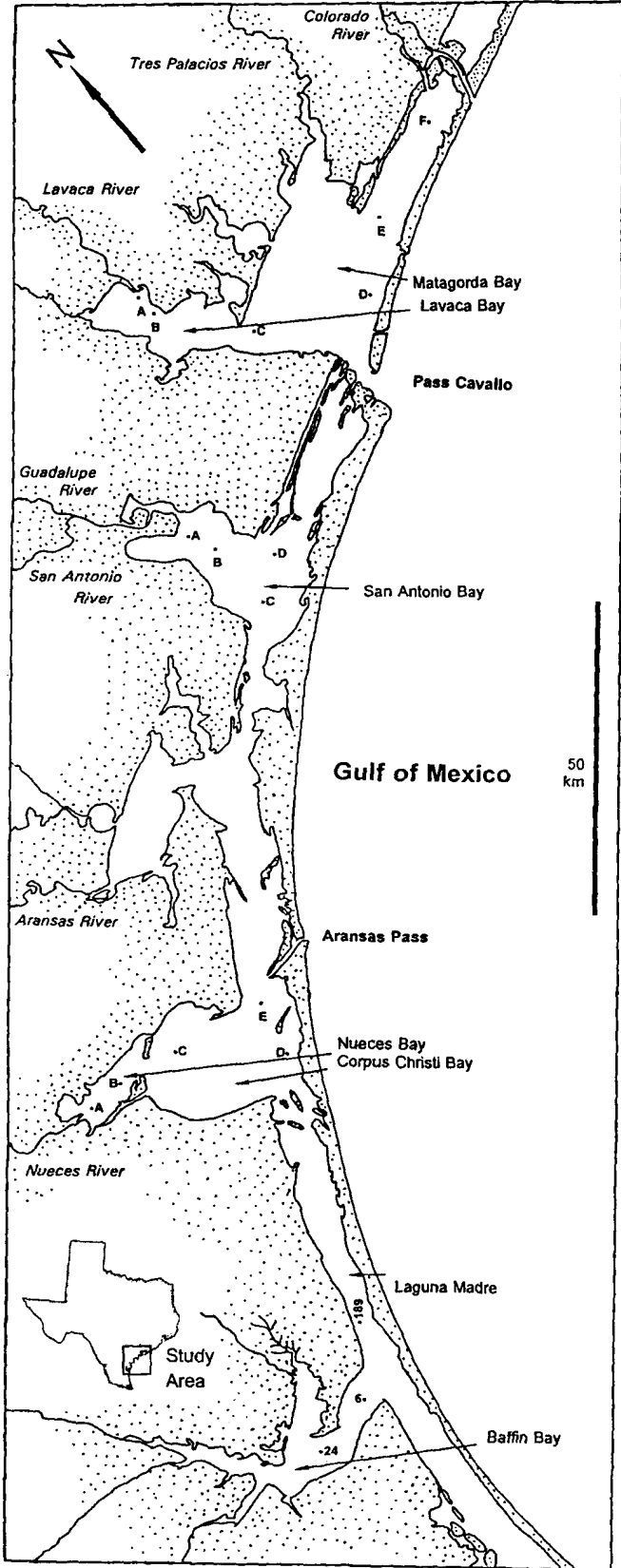
Figure 1. Locations of long-term sampling stations in estuaries of the Coastal Bend region of Texas.

2.2. Model description

The long-term, benthic macrofaunal data sets from the four estuaries were used to calibrate the model of biological processes using a monthly time step. The two principle environmental factors associated with FWI 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, salinity was used as a surrogate for inflow. Salinity values represent the integration of all the physical characteristics of the estuary (e.g., size, inflow, outflow, residence time, tidal exchange, and climatic variability). Other input data to the model included fish and crabs as predators, temperature, water depth, day length, and nutrient concentrations.

Using energy circuit language of (Odum, 1971, 1983), a schematic of stores and flows in the benthos was conceptualized to guide model development (Fig. 2). There are two main trophic guilds in benthic sediments: the grazing food-chain and the detrital food chain (Tenore et al., 2006). Grazers consume autotrophic production and detritivores utilize heterotrophic production. To simplify the model, all macrobenthic animals were separated into one of two groups: the suspension feeders to represent the grazing food chain and deposit feeders to represent detritivores. In Texas Lagoons, most of dominant taxa and ecologically important species fall into these two functional groups (see Table 3 in Tenore et al. (2006)), and these two trophic groups are state variables in this modeling exercise. Suspension feeders are those organisms obtaining their food by 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. 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.
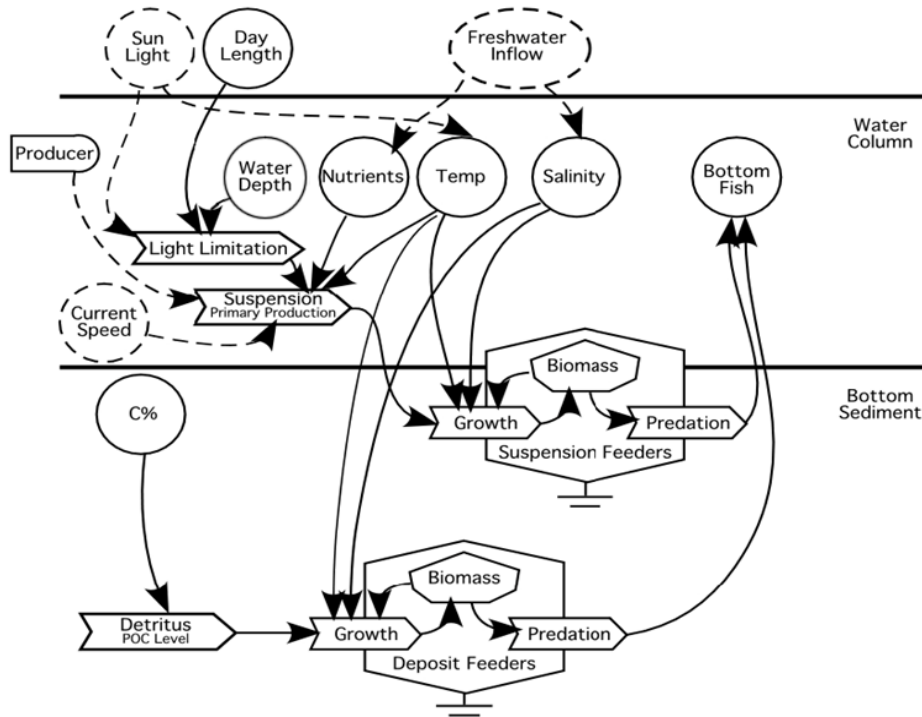

Figure 2. Energy circuit diagram for the structure of the benthic macrofauna biomass model. Dashed lines represent variable not included in the model.

The model schematic (Fig. 1) also shows two major environmental (salinity and temperature) and biological/ecological factors (food sources and predators), which regulate growth and loss of the two state variables. Effects of these environmental factors were parameterized in terms of scaling factors representing environmental limitations. Other environmental variables (nutrients and day length) were used for estimating food source availability (i.e., primary production). Primary producers, whose growth is based on irradiance, temperature, and nutrient concentrations, are the main food source for suspension feeders. 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 based on data (Montagna, unpublished). However, in the present study, the dynamics of POM, nutrients, and salinity caused by mixing and benthic-pelagic feedbacks were not simulated. Instead, the measured POM, concentration of nutrients, and salinity were directly used as input data, assuming that the observations of these properties are reflection of physical and biogeochemical mixing features in the estuarine system.

2.3. Equations

2.3.1. Governing equation for state variable

The governing equation uses a template of Lotka-Volterra growth model (Lotka, 1925) that models a density-dependent logistic growth of a population (Brown and Rothery, 1993):

$$\frac{dB}{dt} = r \cdot B \cdot \left(1 - \frac{B}{c}\right) - g \cdot F \tag{1}$$

where r is the maximum net growth rate of benthos (B) without predation pressure. c is the biomass carrying capacity for a population that is limited by space. The predation loss is calculated by the feeding rate of predators, g, and the density of predatory crab and fish, F. Therefore, Eq. (1) has a unit in biomass (mg or g) over time (day or month). In general, the net growth rate implicitly represents the filtering rate, ingestion rate, assimilation efficiency, respiration rate, aging mortality, and excretion rate; and the only loss of benthos biomass is by the predation rate.

However, growth rates in a population are also influenced by many other environmental effects. In this study Eq. (1) was modified to include environmental limitations as dimensionless scaling factors ($E's$), which range

between 0 and 1. When $E = 1$, there is no environmental limitation, and the benthic population can obtain a maximal growth rate or the predators can reach a maximal feeding rate. When $E = 0$, environmental factors reach maximum limitation, benthic populations do not grow, or predators do not consume benthos. The governing equation of the model is as follows:

$$\frac{dB_{(i,j)}}{dt} = r_{(i)} \cdot E_{ben(i,j)} \cdot B_{(i,j)} \cdot \left(1 - \frac{B_{(i,j)}}{c_{(i)}}\right) - g_{(i,j)} \cdot \sum_k F_{(k,j)} \qquad (2)$$

where $i = 1$ - 2 for deposit feeders or suspension feeders, $j = 1$ - 2 for the two bay systems (primary and secondary bay), $k = 1$ - 3 for three different predators (red drum, black drum and blue crab), $r_{(i)}$ is the monthly net growth rate (month $^{-1}$), $E_{ben(i,j)}$ is the environmental limitation for benthic biomass growth (dimensionless), $c_{(i)}$ is the biomass carrying capacity levels for the two feeding groups (mg dw m^{-2}), and $g_{(i,j)}$ is the temperature- and density-dependent predation rate (g dw m^{-2} month^{-1} individual^{-1}) by fish k in bay j to prey benthos i. $F_{(k,j)}$ is average density for predator fish k (number of individuals).

The next two sections describe the parameterizations of rate processes (e.g., growth and predation) and effects of limitations (e.g., temperature, salinity and density) in the governing equation (Eq. 2), and these parameters will be derived for deposit and suspension feeders in all four bays. For convenience, species and location index, i and j, will not be included in the parameterizations described in the following sections.

2.3.2. Dimensionless scaling factor of benthic growth (E_{ben})

Benthic growth can be controlled by three environmental variables: temperature, salinity and food availability. Thus, the term E_{ben} in Eq. (2) can be

parameterized with respect to temperature (E_{ten}), salinity (E_{sa}), and food concentration limitation (E_{food}):

$$E_{ben} = E_{tem} \cdot E_{sal} \cdot E_{food} \tag{3}$$

An Arrhenius-type exponential equation was used to represent the limiting effect of temperature (Carrada, 1983). This formula can provide an accurate temperature-dependent metabolism function and is used in estimating temperature-dependent growth rate:

$$E_{tem} = \frac{1}{e^{\frac{|T - T1_{opt}|}{T_{w\_ben}}}} \tag{4}$$

where E_{ten} is the temperature limitation, T is the temperature (ºC), and $T1_{opt}$ is the most suitable temperature (ºC). When $|T - T1_{opt}|$ is close to T_{w_ben}, $E_{ten}=1$, and there is no temperature limitation. Therefore, T_{w_ben} is a parameter (ºC) that describes the weighting due to temperature limitation, and higher T_{w_ben} leads to higher sensitivity of E_{ten} to temperature (Fig. 2).

Salinity is one of the most influential environmental variables affecting benthic communities and is directly correlated with FWI. All invertebrates have optimal salinity ranges at which population growth is maximal (Wohlschlag et al., 1977). We used an Arrhenius exponential function in the model in order to represent salinity limitation:

$$E_{sal} = \frac{1}{e^{\frac{|S - S_{opt}|}{S_{w\_ben}}}} \tag{5}$$

where E_{sa} is the salinity limitation, S is salinity (psu), S_{opt} is the optimal salinity (psu) for a population, and S_{w_ben} is a parameter (psu) that describes the weight of the salinity limitation. There is no salinity effect when $S_{w\_ben}=\infty$. Salinity limitation has a centralized optimum, with greater effects at high and low salinities. The greater the salinity tolerance range is, the higher is the S_{w_ben}

value (Fig. 3). The equation has the same form as that used for temperature limitation, but the parameters (T_{opt}, S_{opt}, T_{w_ben} , S_{w_ben}) were calibrated independently.
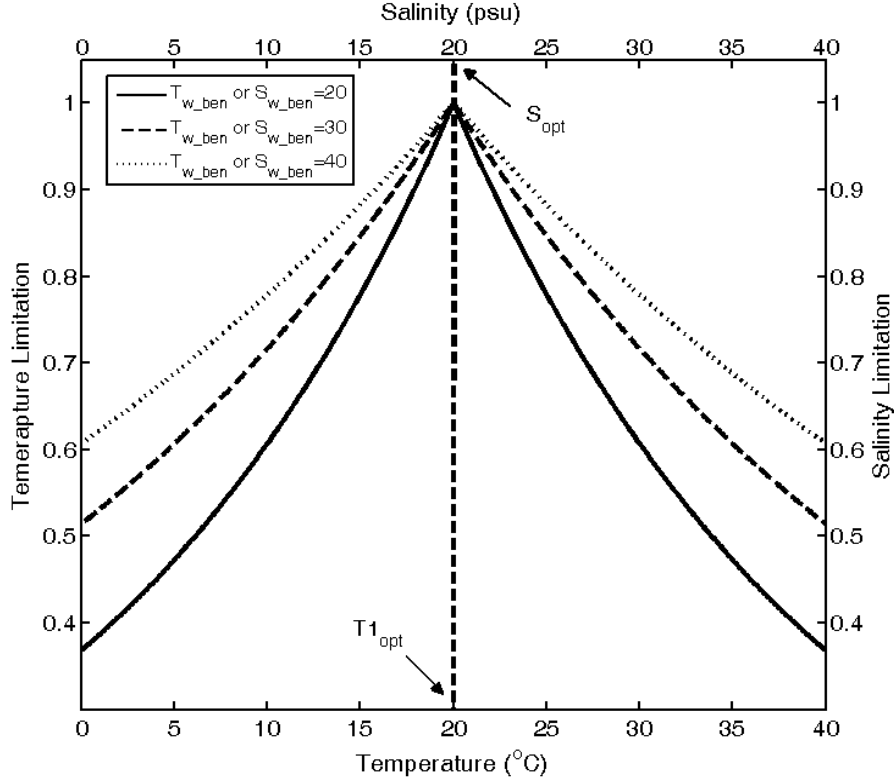

Fig. 3. Limitation by temperature and salinity. Optimal temperature (T_{opt}), salinity (S_{opt}) and sensitivity parameters (i.e., T_{w_ben}, S_{w_ben}) are based on Eqs. (4) and (5), respectively, and these parameters were calibrated independently.

The limitation of food source was described with a Michaelis-Menten type of uptake kinetics (Keen and Spain, 1992):

$$E_{food} = \frac{M_{(i)}}{M_{(i)} + K_M}$$ (6)

where E_{food} is the food limitation, $M_{(i)}$ is the concentration of the food source for benthic organisms (i = 1, 2), and K_M is a parameter at which the food concentration is at half the maximum level of the population growth rate.

As two feeding groups are simulated in the model (deposit and suspension feeders), there are 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.

The POM levels (p_{pom}) were calculated from the measured percent carbon content in the sediment samples (C%) for two bays:

$$C(\%) = \frac{p_{poc}}{p_{sed}} \cdot 100 \tag{7}$$

where p_{poc} is the sedimentary POC level (g C m^{-2}) for each bay, and p_{sed} is an average dry weight of the whole sediment (g dw) per core sample (where area of the core is 35.4 cm^{-2}), which was set at 16.4 (Montagna, unpublished). These sedimentary POM levels represent the food sources available for deposit-feeders ($M_{(1)}$) in each bay:

$$M_{(1)} = p_{pom} \tag{8}$$

Primary production is expected to be the most important food source for suspension feeders ($M_{(2)}$). Primary production is simulated as a function of day length, temperature, and nutrient concentration:

$$M_{(2)} = F \cdot p_{pp} \cdot \frac{1}{e^{\frac{|T-T1_{opt}|}{T_{w\_pp}}}} \cdot \frac{L}{13.9} \cdot E_{nut} \tag{9}$$

where $M_{(2)}$ is the available food for suspension feeders, F is the unit conversion factor described in Eq. (10), and p_{pp} is the maximum daily primary production previously reported (5 g C m^{-2} d^{-1}; Stockwell, 1989), respectively. The following term, $\frac{1}{e^{\frac{|T-T1_{opt}|}{T_{w\_pp}}}}$, is the temperature limitation for primary production (dimensionless). The same type of response curve as in benthos (Eq. 4) was used, but a different weighting for temperature limitation (T_{w_pp} in ºC) was calibrated

for primary production. $\dfrac{L}{13.9}$ is the day length (L; hours) normalized by

maximum day length (13.9 hours) in the area between July and August. This is a

scaling factor to represent light limitation resulting from the length of daylight.

E_{nu} is the nutrient limitation (dimensionless) for photosynthesis that includes

concentrations of nitrogen (N), silica (Si), and phosphorus (P). The following

adjustment is to convert the unit:

$$F = \frac{30}{0.42} \times \frac{10}{100 \cdot d_{(j)}} \tag{10}$$

where $d_{(j)}$ is the water depth (m) and the constants are used to convert from a day

to a month (30 days per month), from carbon to dry weight (42% of carbon

content per dry weight), and meter to centimeter (100 cm per m). Because

suspension feeders are assumed to use the available food source 10 cm above the

sediment surface, 10 (cm) is the fraction of the entire water column where

suspension can reach to feed on primary production carbon. Therefore, the final

unit for suspension feeder food availability is measured in $\left(\dfrac{g \cdot dw}{m^2 \cdot month}\right)$ where dw

stands for dry weight.

Nutrient limitation (E_{nut}) for photosynthesis was based on Liebig's law of

the minimum, which in this case states that nutrient-dependent photosynthesis

occurs at the rate permitted by the most limiting nutrient. In other words, the

minimum of the three Michaelis-Menten equations for uptake kinetics of N, Si

and P will determine the photosynthetic rate. The conversion of units followed

the Redfield ratio (C:N:Si:P = 106:16:15:1), which assumes that producers use

carbon, N, Si, and P proportionally (Redfield, 1934):

$$E_{nut} = MIN\left(\frac{[N]}{[N]+K_N}, \frac{[P]}{[P]+K_P}, \frac{[Si]}{[Si]+K_{Si}}\right) \tag{11}$$

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where [N], [P], and [Si] are concentrations (µM) of inorganic nitrogen, phosphorus and silica, respectively. K_N, K_P, and K_{Si} are half-saturation concentration (µM) for nitrogen, phosphorus and silica, respectively. It is assumed the Redfield ratio (C:N:Si:P = 106:16:15:1) can be used to convert among half-saturation constants (K_N:K_{Si}:K_P). For example, when phosphate was used as model currency, $K_N = K_P \times 16$; $K_{Si} = K_P \times 15$ in Eq. (11). Therefore, the rate constant determining photosynthesis (the minimum value of Michaelis-Menten kinetics in Eq.11) was determined by ambient nutrient concentrations.

A forcing function was derived to simulate day length (number of hours) in order to represent seasonal effect of photosynthesis:

$$L_{(t)} = 12.15849 - 1.755811 \cdot \cos\left(\frac{2\pi \cdot (t)}{12} - 0.2244535\right) \tag{12}$$

where $L_{(t)}$ is day length (h) at time t (month).

2.3.3. Predation

Predation is a function of environmental factors (e.g., temperature), benthic biomass, and predator density. In this study, the predation rate (g dw m^{-2} month^{-1} individual^{-1}) is modeled as a function of temperature and prey benthic biomass, because temperature (Houde, 1987; Houde, 1989; Pepin, 1991; Poulet et al., 1995; Calbet and Agusti, 1999) and prey aggregation (Montagna et al., 1993) are related to the feeding rate of predators.

Predation was simplified as a function of temperature and prey density. Therefore, the term g in Eq. (2) is the temperature- and density-dependent predation rate (g dw m^{-2} month^{-1} individual^{-1}) and was parameterized as follows:

$$g = 10^{T_{w\_pred} \cdot (T - T2_{opt})} \cdot I_v \tag{13}$$

where, T is temperature (°C) in both bays, $T2_{opt}$ is an optimal temperature (°C) that determines the range of responses, and T_{w_pred} (°C^{-1}) is the sensitivity parameter for temperature-dependence. I_v is a modified Ivlev's equation (Ivlev, 1961) by Mayzaud and Poulet (1978) to express the non-linear effect of prey density on predation rate (Eq. 14):

$$I_v = \lambda \cdot B \cdot (1 - e^{-\lambda \cdot B})$$
(14)

where, B is the biomass of the benthic prey in g dw m^{-2}, and λ ((g dw m^{-2})$^{-1}$) is the parameter for the aggregation effect of prey. Therefore, as B increases, the term I_v increases to its maximal grazing rate. When benthic biomass (B) is at a very low level the value of term I_v is close to 0, and aggregation effect is nil.

2.4 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).

$$\%RMSD = \sqrt{\frac{\sum \frac{(X_{mod} - X_{obs})^2}{n}}{\sum \frac{(X_{obs})^2}{n}}} \times 100$$
(1)

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

2.5 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)

3. Preliminary results for Lavaca-Colorado Estuary

3.1 Data Results

Long-term differences in salinity and temperature can be seen among the estuaries (Table 2). Salinity and temperature is lower in the secondary bay than the primary bay in all estuaries. The difference is less between Baffin Bay and Laguna Madre.

Table 2. Mean (and standard deviation) of physical characteristics of the estuaries.

Estuary	Bay	n	Salinity(psu)		Temperature(C)		DO(mg/l)	
LC	Secondary	365	14.9	(9.2)	21.4	(6.6)	8.0	(1.6)
LC	Primary	568	23.4	(7.9)	22.6	(6.6)	7.5	(2.0)
GE	Secondary	296	9.4	(8.0)	22.4	(6.5)	8.6	(2.3)
GE	Primary	281	16.8	(9.5)	22.3	(6.4)	8.2	(1.9)
NC	Secondary	328	25.2	(9.5)	22.6	(6.3)	7.5	(1.6)
NC	Primary	474	31.1	(4.8)	22.4	(6.2)	6.9	(1.7)
LM	Secondary	233	37.8	(10.5)	23.0	(6.0)	6.7	(1.9)
LM	Primary	166	37.9	(8.1)	23.7	(5.8)	8.0	(1.8)

3.2 Model Results

The simulations of benthic biomass are based on the best fit parameters from the calibration of the period (1988 – 1999). All simulations were run from April 1988 until April 2005, and were compared to observed benthic macrofauna biomass data (Table 3, Fig. 4). Simulations of deposit feeders in Lavaca Bay followed a pattern of increases followed by a dramatic drop in biomass with the lowest biomass concentration occurring between 1994 and 1996 (Fig. 4a). After the year 2000, the biomass concentrations showed signs of slowly increasing. Simulations of suspension feeder biomass in the same bay followed the same pattern as the deposit feeder biomass, and had a trend of a slower increase in biomass after the year 2000 (Fig. 4b). In Matagorda Bay, simulations of deposit feeders showed a trend having low biomass during the period 1994 – 2000 and increased biomass post-2000 (Fig. 4c). Simulations of suspension feeders for

Matagorda Bay showed a similar trend to that of deposit feeders; having low biomass during the period 1994 – 2000 and increased biomass post-2000.
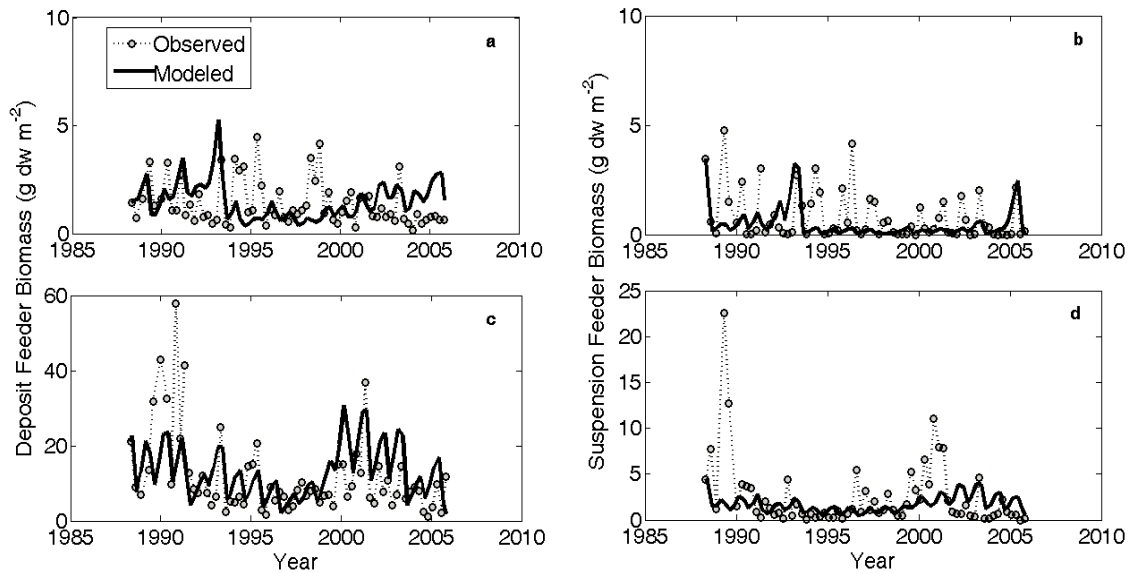

Fig. 4. Comparisons between observed (dashed line with open circles) and modeled (solid lines) results in Lavaca and Matagorda Bay for the period 1988 – 2005. Each panel represents comparison results for (a) deposit and (b) suspension feeder biomass in Lavaca Bay, and (c) deposit and (d) suspension feeder biomass in Matagorda Bay.

Table 3. The percent root mean square (RMS) difference between observed and simulated benthic biomass in the Lavaca-Colorado Estuary.

Periods for comparisons	Lavaca Bay		Matagorda Bay	
	Deposit Feeders	Suspension Feeders	Deposit Feeders	Suspension Feeders
January 2000 – April 2005 (for validation)	66.6	74.2	78.0	78.5
April 1988 – April 2005 (for long-term simulation)	76.1	86.6	64.1	82.5

The simulations for both bays and each feeding group fit the observed data relatively well during the entire period, 1988 – 2005 (Fig. 4, Table 3). It is

also noteworthy that the trends of the prediction over time fit the trends in the observed biomass for both bays. However, the model still needs improved performance (i.e., minor tuning processes) because a few segments of the time period showed large deviations. For example, the worst fit happened to the suspension feeders in Lavaca Bay (86.6%), which shows large discrepancies during 1994 – 2000 when the model predicts rather low and constant biomass compared to the higher values of the actual data (Fig. 4b). Similarly, simulations of deposit feeders on the same bay were underestimated during the same period (Fig. 4a).

The deviation in Lavaca Bay is attributed to the misfit for the period from 1994 to 2000 when predators' biomass was high (Fig. 5a). This can be explained by two possibilities: 1) an increase in predator populations during this period, particularly blue crabs (see Fig. 5a), did not have much predation impact on prey, and this caused prey populations to remain relatively high during this period; 2) whereas, the model simulations were still strongly influenced by the increased number of predators during this period when numbers of blue crabs caught in Lavaca Bay have increased (Fig. 5a).
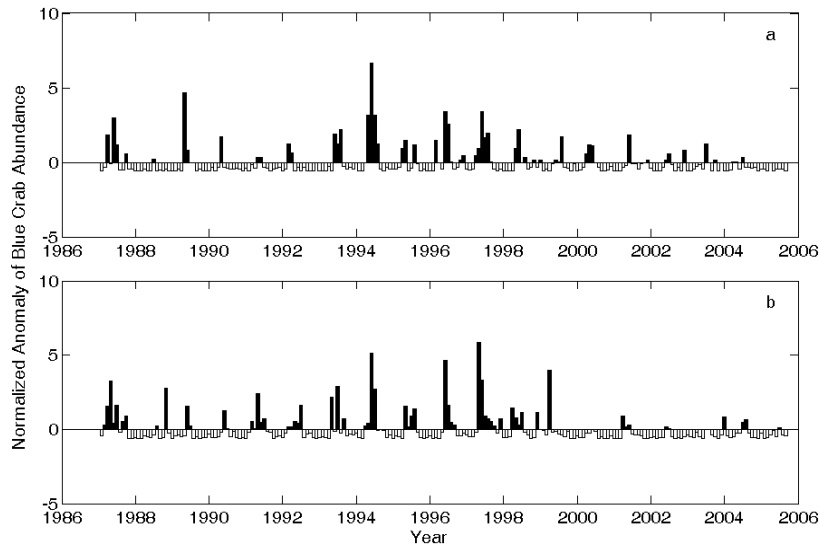

Fig. 5. Normalized anomaly of blue crab abundance. Positive and negative values

represent above and below long-term average, respectively. Anomaly was normalized by standard deviation. Blue crab population data (1987 – 2006) were collected by TPWD for (a) Lavaca Bay and (b) Matagorda Bay.

In Matagorda Bay, the simulated results (Figs. 4c, d) had a coupled trend of both deposit and suspension feeders with a decreasing trend during the high predation period (1994 – 2000), and an increasing trend during times of low blue crab biomass (Fig. 5b). Blue crab abundance in Matagorda Bay also had a similar trend of that in Lavaca Bay, having higher abundance during the period 1994 and 2000 and lower abundance in the period after 2000 (Fig. 5b). The simulations for the deposit feeders fit the data best (64.1%, Table 3), having low biomass during the high predation period (1994 – 2000) and high biomass when the number of predators starts decreasing in the post-2000 period (Fig. 5b). The higher variance for suspension feeders in Matagorda Bay can be attributed to underestimated biomass in the post-2000 period (Fig. 4b).

4. Discussion

Overall, the Matagorda Bay simulation fit the observed data relatively well for the periods 1994 – 2000 (Figs. 4c, d), but simulations for both deposit and suspension feeders fail to capture peaky biomass around 1990. Again, this can be explained by model behavior. For example, low biomass in predator populations during two periods: 1988 – 1994 and 2000 – 2005 (Fig. 5b), had a stronger effect in maintaining benthic biomass at a high level than the model could simulate. In other words, the sensitivity of the model to predation is not large enough to respond to a decreased level of the predators' population.

During the second year of the study (FY2008), the model will be run for the other three bays (GE, NC, and LM).

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