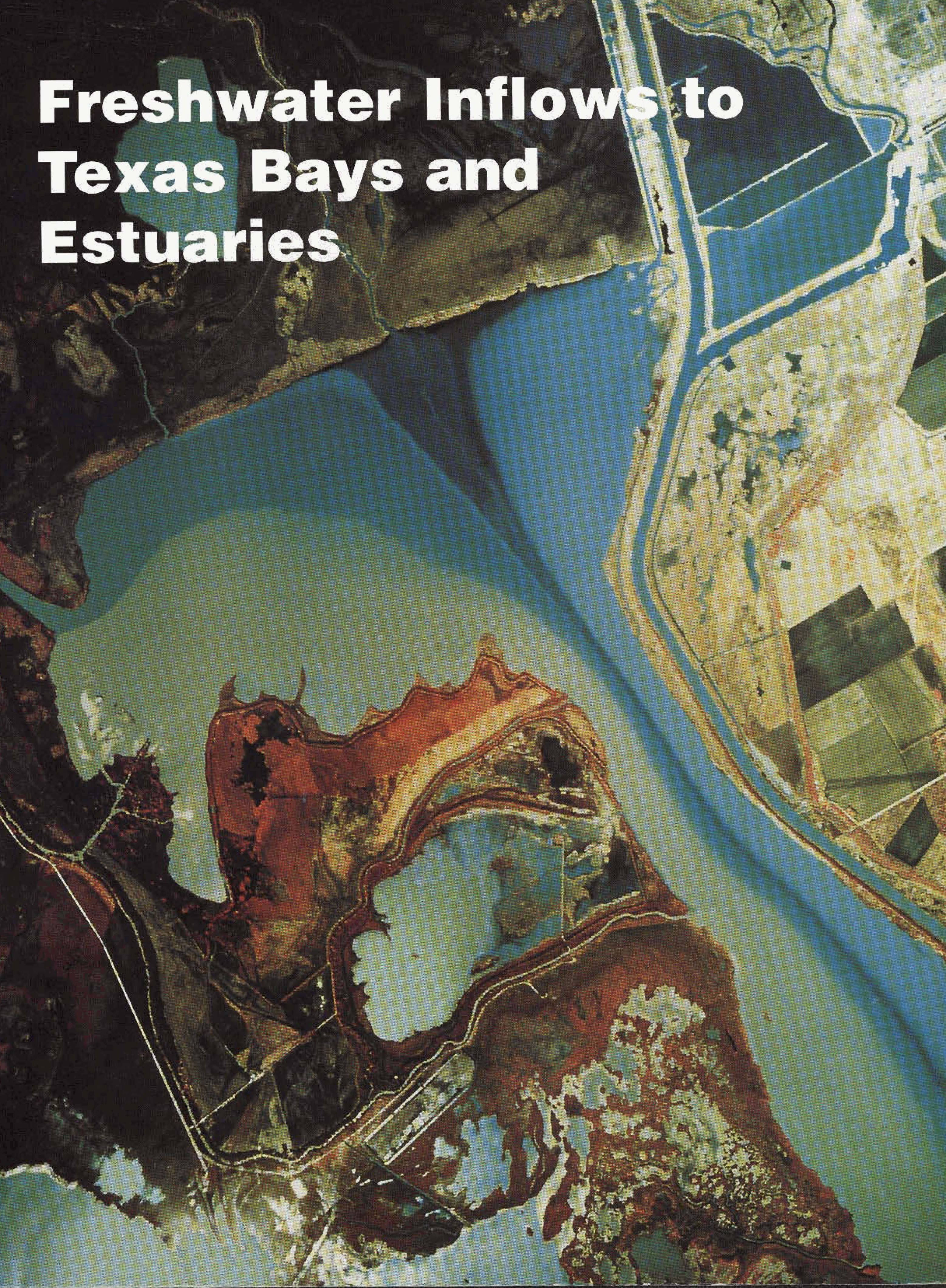
Freshwater Inflows to Texas Bays and Estuaries



FRESHWATER INFLOWS TO TEXAS BAYS AND ESTUARIES: ECOLOGICAL RELATIONSHIPS AND METHODS FOR DETERMINATION OF NEEDS

Edited by

William L. Longley
Texas Water Development Board
1700 N. Congress Ave.
Austin, TX 78711-3231

Bay and Estuary Study Project Directors

Gary L. Powell
Texas Water Development Board
1700 N. Congress Ave.
Austin, TX 78711-3231

and

Albert W. Green
Texas Parks and Wildlife Department
4200 Smith School Road
Austin, TX 78744

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FOREWORD

Natural resource managers have been concerned about freshwater inflow needs for Texas bays and estuaries ever since the planning and construction of large-scale water development projects commenced in the state. They realized that these projects might deprive the estuaries of needed fresh water that provides appropriate salinity regimes and concentrations of nutrients and sediments to sustain the health of the coastal environments. Initially, there was little scientific information on which regulatory authorities could base decisions concerning the quantity and timing of freshwater inflows to the estuaries. In the last 30 years, however, researchers from universities, state and federal agencies, and private research organizations have provided significant insight into how Texas estuaries function and the importance of freshwater inflows to their ecological health.

This report integrates the results of recent studies with earlier information to provide a comprehensive overview of the importance of freshwater inflows to Texas estuaries. The report emphasizes the relationship of inflows with the chemical composition and physical nature of estuarine ecosystems, bay habitat distribution, physiological processes, biological productivity, and abundance of fish and shellfish populations. In addition, the report presents a methodology for determining the amount and timing of beneficial inflows needed to maintain the productivity of economically important fishery species, and the estuarine life on which they depend. This procedure deals effectively with competing inflow requirements among organisms and includes provisions for achieving management goals for specific estuarine habitats and species. The report provides data and example analyses of inflow needs for San Antonio Bay and the Guadalupe Estuary using several state management objectives. The participating agencies are now preparing detailed analyses for each principal bay and estuary using the information and methods presented here.

Applying this approach to determining freshwater inflow needs of the state's major estuarine systems will improve management of our coastal resources. The analytical procedure is based on real data and relationships established for each bay system, rather than just theoretical formulations. In addition, the procedure allows use of management goals that are appropriate for each estuary and take into account the specific conditions of each bay system. This will provide a solid quantitative basis on which future decisions can be made concerning estuarine freshwater needs, improving the planning and management of important freshwater supplies. Regulatory decisions accounting for freshwater inflow needs, as determined by this procedure, along with a continuous monitoring effort to confirm benefits and suggest adjustments to inflow will ensure that good estuarine health is maintained.

The cooperative process used by the three participating state agencies in this joint effort illustrates the value of having experts from organizations with differing responsibilities work together to address problems of state concern. The participating agencies are committed to working together on problems involving water and other valuable natural resources of the state.

Craig D. Pedersen
Executive Administrator
Texas Water Development Board

Andrew Sansom
Andrew Sansom
Executive Director
Texas Parks and Wildlife Department

Anthony C. Grigsby
Anthony C. Grigsby
Executive Director
Texas Natural Resource Conservation Commission

PREFACE

In the 69th Legislative session and in subsequent sessions, the Texas Legislature directed the Texas Water Development Board and the Texas Parks and Wildlife Department to share responsibility for establishing and maintaining a continuous bay and estuary data collection and evaluation program, and to conduct studies and analyses for determining bay conditions necessary to support a sound ecological environment [TEXAS WATER CODE 16.058(1)]. This document reports the effects of freshwater inflows on the biological productivity of bays and estuaries, and effects on the distribution and abundance of economically important and ecologically characteristic fish and shellfish species, and the estuarine life on which they depend. The target audience for this document consists of natural resource managers and decision makers, estuarine scientists, and well-informed lay public.

Contract studies performed by personnel at the University of Texas Marine Science Institute (UTMSI), University of Texas Center for Research in Water Resources (CRWR), Texas A&M University, University of Houston (UH), University of Texas Bureau of Economic Geology (BEG), National Marine Fisheries Service (NMFS), and the Texas Parks and Wildlife Department (TPWD) were prepared for and funded by this study and exist as contract reports in the state library.

In addition to documenting the importance of freshwater inflows, this report presents an analytical methodology for determining freshwater inflow needs for Texas estuaries. The procedure was designed to assist the Texas Natural Resource Conservation Commission in quantifying beneficial inflows that are necessary for maintaining an ecologically sound environment. The analytical technique that is presented uses hydrodynamic modeling, optimization programming, and data that was compiled from special studies, monitoring programs, and historical records. It includes consideration of salinity, nutrient, and sediment loading regimes, all topics that the Texas Natural Resource Conservation Commission must consider when determining the beneficial inflows necessary to maintain an ecologically sound environment for the maintenance of productivity of fish, shellfish, and other estuarine life.

The methodology for determining inflow quantities and timing requires a variety of information about resource management objectives and limits (constraints) since it is a mathematical procedure that runs on a computer. Consequently, policy makers and regulators must provide unambiguous statements of these information types for the procedure to be used. The report provides examples of the kinds of objectives and limits that must be provided, and some of the results of applying different resource management policies to an example analysis for the Guadalupe Estuary. Actual inflow recommendations for the Guadalupe Estuary, as well as for other Texas estuaries, will be presented in separate reports. In addition, more complete information detailing the characteristics of the Laguna Madre Estuary will be presented in a future publication.

Any inquiries about this publication should be directed to:

Gary L. Powell
Environmental Section
Texas Water Development Board
P. O. Box 13231
Austin, Texas 78711-3231
(512) 463-8043

or

Albert W. Green
Resource Protection Division
Texas Parks and Wildlife Department
4200 Smith School Road
Austin, TX 78744
(512) 389-4800

SUMMARY

In response to the TEXAS WATER CODE 16.058(a), the Texas Water Development Board (TWDB) and the Texas Parks and Wildlife Department (TPWD) have prepared a report on the effects of freshwater inflows on Texas bays and estuaries. Two main themes are examined in the report: demonstrating the effects of freshwater inflows on living and non-living components of estuarine ecosystems; and presenting a methodology for assessing the freshwater inflow needs of Texas bays and estuaries that satisfies the requirement of maintaining an ecologically sound environment and the productivity of fish, shellfish, and other estuarine life.

Chapter 1. The first chapter of the report provides a historical perspective on freshwater inflows. Thirteen different functions of freshwater inflows have been identified in various studies. Some of the most important functions include: creation and maintenance of low-salinity nursery habitats; provision of a medium for transport of beneficial sediments and nutrients; transport of allochthonous (external) organic materials from upland or delta areas; and control of the timing of movement of some estuarine species. This chapter also provides a list of 15 effects of reduced freshwater inflows noted in the scientific literature. Among the most significant impacts are: increased salinities and vertical stratification in the water column; penetration of the salt-wedge farther upstream allowing intrusion of predators and parasites of estuarine species, and increased intrusion into groundwater and surface water resources; increased frequency of benthic anaerobic conditions and decreased inputs of nutrient and organic matter used by estuarine species; loss of characteristic species and economically important seafood harvests; and increases in erosion of delta areas resulting from the reduction of sediment influx.

Chapter 2. Chapter 2 of the report reviews the legislation authorizing this study, provides an interpretation of the meaning of the phrase "ecologically sound environment," and discusses the steps needed to determine whether beneficial inflows are adequate to maintain an ecologically sound environment. The chapter identifies three general goals for the study, and also lists a series of eight objectives that were initially defined by representatives of the TWDB, TPWD, and Texas Natural Resource Conservation Commission. The specific objectives are: compiling inflow, hydrographic, and biological data into computer-compatible files; developing circulation and salinity models; evaluating the effects of inflow and salinity change on estuarine plants and animals; assessing water quality trends during the past 20 years; determining the effects of inflow on bay sedimentation and river delta maintenance; evaluating the effects of fresh water on primary production, nutrient loading, and biogeochemical cycling; preparing statistical inflow-harvest equations for commercial catches; and developing a methodology for analysis of inflow needs by optimization modeling.

Chapter 3. The third chapter describes the analytical approach used in the study. First, the hydrologic database for Texas estuaries has been extended through 1987 to include 11 more years of information. Then, specific studies of the effects of freshwater inflows on components or natural processes of estuarine ecosystems are presented. Important ecosystem components examined include salinity, macronutrients (nitrogen, phosphorus, and carbon), sediment, phytoplankton, seagrasses, marsh plants, zooplankton, benthic organisms, and larval and adult fish and shellfish. Abundance and distribution information are among the most important characteristics investigated for these ecosystem elements. Dynamic processes given detailed analysis include: photosynthesis; zooplankton production; nutrient uptake, regeneration, nitrification, and denitrification; larval transport; spawning and larval development of fish; metabolism of juvenile and adult fish with respect to salinity; and production of commercial species as measured by commercial harvest of species.

The third step in the analytical approach is to develop and present an analytical procedure for determining freshwater inflow needs. There are several requirements for such an analytical procedure. It should give quantitative results on a monthly or bimonthly scale that is appropriate with managerial actions. The estimates should be realistic in terms of what kinds of flows can be supplied. The method should focus on the maintenance of productivity, but should specifically address the concerns of salinity, nutrient, and sediment loading. It should be easily modifiable, efficient, and flexible, and should provide a method to check the results. The method will then be applied on a test basis to an estuary on the Texas coast. In addition to being a realistic test case, the analysis should show the breadth and scope of the effects of varying some of the elements of the analytical method, especially differences from altering management objectives and limits. Finally, the types of information and decisions that state policy makers and regulators must provide to use the method are presented.

Chapter 4. Chapters 4, 5, and 6 provide an analysis of the effects of freshwater inflows on Texas estuarine ecosystems. In Chapter 4, the patterns of inflow and salinity are examined for the Nueces, Mission-Aransas, Guadalupe, Lavaca-Colorado, Trinity-San Jacinto, and Sabine-Neches estuaries. Differences in inflow between these estuaries are dramatic and follow the precipitation gradient of the state. On average, the Sabine-Neches Estuary annually received more than 50 times its volume in freshwater inflow during the 47-year period of record. The Trinity-San Jacinto and Guadalupe estuaries received four to seven times their volumes on an annual basis, while the Lavaca-Colorado received about twice its volume, on average. Both the Nueces and Mission-Aransas estuaries received only 60 to 70% of their volume on an average annual basis, and the relative variation in inflow was higher from year to year than for the estuaries along the upper coast. Salinities were just the inverse of the inflow situation with lowest salinities in the Sabine-Neches Estuary and highest in the Nueces system.

During the past 47 years, the Mission-Aransas Estuary had a statistically significant increasing trend in inflow. None of the other systems showed a significant trend during that long period. The Nueces Estuary had a decreasing trend of inflow (about 4% per year) from 1966 through 1987. A trend analysis for salinities from 1968 through 1987 showed decreasing salinity in the lower Sabine-Neches Estuary and increasing salinity for West Galveston Bay, lower San Antonio Bay, and Nueces Bay.

The proportion of the nutrient load carried by the gaged flow of rivers compared to the total nutrient load varied from one estuary to another, in part because of differences in the relative proportion of gaged and ungaged areas. The combined inflow from gaged and ungaged inflow sources accounted for more than 52% of the nutrient load for the Nueces Estuary and more than 75% for the other estuaries. Thus, freshwater inflows provide the majority of the nutrient influx to estuaries. None of the Texas estuaries appeared to be nutrient limited; all had a decreasing gradient of nutrient concentration from the head to the mouth. High turbidities in the upper estuary and efficient regeneration in the lower bays allow nutrients to move through the system and be reused without contributing to eutrophication problems.

While there are differences in the texture of suspended sediment loads of Texas river systems, sediment load is uniformly dependent on inflow. On examining cumulative sediment-cumulative discharge relationships, the Trinity River showed a definite change in the sediment load after the completion of construction of Lake Livingston, located upstream of the delta. Lake Corpus Christi retains 97% of the sediment entering it, and only about 70% of the sediment that formerly entered Lavaca Bay is now deposited there due to Lake Texana. The Nueces and Trinity river deltas are currently undergoing changes that can be attributed to the reduced sediment loads. In contrast, several other rivers showed changes in sediment load through time, but the changes could not be attributed to human activities; a number of sediment load changes occurred immediately after the drought period in the 1950's.

Chapter 5. While Chapter 4 is largely concerned with the abiotic portions of the estuarine ecosystems, Chapter 5 considers the primary producers (phytoplankton and plants), the primary consumers of production and detritus (zooplankton and benthos), the nutrient regeneration process, and the distribution and use of terrestrial and delta carbon throughout the estuary. Both phytoplankton and vascular plants (seagrasses and marsh vegetation) show responses to freshwater inflow. Under high inflows, flagellates often dominate the phytoplankton, while diatoms predominate during periods of low inflow and high salinities. Because of their longer life cycle, vascular plant response to freshwater inflow must integrate the effects of inflows over a longer period than phytoplankton. Inundation and salinity level are probably the factors resulting from inflow that most affect vascular plant composition; long periods of high or low inflow result in major population shifts. While productivity of all types of plants is probably influenced by freshwater inflows, it is difficult to measure directly because of the high variability in phytoplankton abundance and the ancillary effects of soil moisture, turbidity, epiphyte growth, and other factors on vascular plant production.

The zooplankton community changes as a result of inflows. High inflows displace estuarine zooplankton, which are then replaced by freshwater species. As salinities increase through mixing and evaporation, freshwater zooplankton are replaced by estuarine species. On an annual basis, zooplankton abundance appears to be influenced by inflows, although the pattern is complicated. Zooplankton abundance is higher in years with greater-than-average inflows in estuaries that normally receive less than one bay volume of inflow each year. In estuaries that, on average, receive five or more bay volumes of inflow per year, increased inflow results in decreased annual zooplankton abundance. The different patterns of change in zooplankton abundance from inflow increases may be caused by opposing forces stimulating zooplankton production under low-flow conditions and displacing them under high flows.

Benthic species also have a response to changes in inflow. Under high inflow, macrobenthic organisms adapted to low salinities flourish until higher salinity conditions return. The more marine species reappear and rebuild to higher levels than existed before the high inflow conditions commenced. Then, the populations decline to pre-inflow levels. A hypothesis for regulation of benthic species by inflows has been suggested and is still being tested by researchers. The rates of regeneration of nutrients by benthic organisms also seem to be related to inflow. They appear to increase both during and immediately after increased inflows, although prolonged periods of low salinity may kill many of the organisms that participate in regeneration.

The nutrient cycling process, of which benthic regeneration is only one element, appears to change depending on the level of freshwater inflow. Those ecosystem components that use and store nitrogen assimilate higher levels of it during periods of high inflow than during periods of low inflow. But during times of high inflow, the efficiency of transfer of nitrogen between ecosystem components per unit of nitrogen input to the ecosystem is much lower than during low-flow periods when the material is cycled more times before being lost from the system.

Inflows also transport and distribute terrestrial- and delta-produced carbon throughout the bays and estuaries. Studies on the Guadalupe and Lavaca systems showed a terrestrial carbon signal in the sediment and in the particulate organic carbon of the water column after large inflows. Analyses of benthic organisms, fish, and shrimp show that in some estuaries, the terrestrial materials transported by freshwater inflows directly or indirectly provide a portion of the carbon assimilated by organisms in their diet. This was particularly true for animals that lived in the upper bays close to the river mouth.

Chapter 6. Chapter 6 examines the relationships between freshwater inflow (as expressed by salinity) and the distribution, abundance, physiology, and production of fish and shellfish. Movements of larval species from the Gulf to the bays were studied to determine whether freshwater inflow provided a cue for their return after spawning and hatching. Transport of larvae was affected by tidal currents, on-shore wind, and tidal type (spring or neap), but no salinity- or inflow-related relationship was found.

A study of the general distribution and abundance of larval and juvenile species showed that higher densities of juvenile shellfish were found in areas with lower salinities than is considered optimal for growth and survival. It also showed that salinity preferences or tolerances change with size (maturity) of the animals, although some species were less affected by salinity than others. Periods of high water associated with high inflows allowed some species (juveniles and adults) to move into inundated wetland areas and feed directly on detritus, benthic species, and epiphytes even though salinity conditions might be low enough to be substantially less than optimal. Many estuarine organisms move as close to sources of fresh water as their physiological adaptations will permit, probably in response to the detritus loads, plankton densities, and high levels of production of benthic animals in the area.

The metabolic costs of ionic and osmotic regulation that result from living in an estuarine environment with a fluctuating salinity regime was presented for several estuarine adult and juvenile fish. At low and high salinity extremes, proportionally more of the metabolic energy of fish must be expended maintaining their internal ionic and osmotic conditions than at more moderate salinities. Therefore, at the extremes of salinity, the metabolic scope for activity such as migration or foraging is substantially reduced. This difference could be seen in both metabolic measurements and maximum sustained swimming speeds of various fish species. Juvenile estuarine fish tend to have a broader range of scope for activity than adults, but proportionally may have a larger reduction in scope for activity than adults at extremes, until they acclimate to the new conditions. Since some species are more numerous in suboptimal salinity regions of estuaries, the advantages of inhabiting low salinity areas must outweigh the considerable metabolic costs.

In addition to influencing the active metabolism of estuarine animals, freshwater inflows affect the reproduction of estuarine species. Studies of reproductive females of two species showed that growth of the ovaries, particularly during the first 30 days of development, was reduced at very low and very high salinities. Egg hatching of spotted seatrout, which remain in the estuary for spawning, showed very little effect from salinity variation; eggs of Atlantic croaker, which spawn offshore, had reduced rates of hatching at high and low salinities, with maximum hatching at marine salinity levels. The effect of salinity on larval development for different ages of larvae was tested over a wide range of salinities. Spotted seatrout had the widest tolerance range, followed by red drum and then Atlantic croaker. These studies showed that salinity level is an important influence in fish reproduction; the adults of these species are mobile, however, so they can usually move to areas where salinity levels are favorable.

While the effects of salinity on scope for metabolism and reproduction are clear, many adult finfish and shellfish are not as sensitive to freshwater inflow and salinity level as juveniles. Gill net data from several estuaries showed that there was no relationship between mean estuarine salinity and the density of (mainly adult) black drum, southern flounder, Gulf menhaden, striped mullet, red drum, and spotted seatrout. Trawls, capturing smaller individuals than gill nets, suggested that there was a relationship between mean estuarine salinity and density for a number of species; white shrimp and Gulf menhaden abundance was reduced when mean salinity was greater than 25‰; brown shrimp, small Atlantic croaker, and small southern flounder densities decrease at salinities of less than 15‰ or greater than 30‰. In addition, literature and other measurements show that mean salinities greater than 25‰ reduce the density of eastern oysters.

The relationship between commercial fishery harvest, a surrogate for productivity, and freshwater inflow was investigated using 29 years of shrimp data and approximately 26 years of data for other fishery products. Equations relating harvest to seasonal inflow, temperature, and harvest effort were prepared for all estuaries but the Laguna Madre where the hydrology was not complete. Results in the Sabine-Neches Estuary were poor due to unfavorable changes in estuarine conditions and discontinuous time series records resulting from shifts in the local fishing industry. Regression equations presented in the report accounted for an average of 69% of the harvest variation (range of 44 to 90%). While the statistically significant terms relating inflows to harvest seemed appropriate for many equations, for some, there was no clear biological explanation for the significant terms.

Chapter 6 shows that through salinity control, inflows do affect fish and shellfish populations. The clearest effects were seen as physiological responses to salinity. The relationships become less clear, however, when they are not based on controlled laboratory experiments. Animals are free to move and select salinities. They go through life history changes as they mature, and are exposed to a host of other conditions in addition to salinity that could alter their survival. Consequently, the importance of freshwater inflows to fish and shellfish density or harvest is clear since it can still be discerned regardless of other influences on population density and harvest.

Chapter 7. Chapter 7 begins the second phase of the report by providing information that will be used in an example analysis of freshwater inflow needs for the Guadalupe Estuary. The estuary received an annual average combined inflow of 2,344,140 acre-ft. The maximum inflow in any one month was 2,457,912 acre-ft in June 1987, and the minimum inflow was 5,123 acre-ft in June 1956. Highest inflows are usually in May and June, with the lowest flows in August. About 82% of the combined flow is from gaged watersheds. A statistical trend analysis of inflows showed no significant changes in inflow in the Guadalupe Estuary during the 1941 through 1987 period, except from droughts. However, there are indications from exceedance probability charts that inflow during the 1968 through 1987 period has increased compared to the 1941 through 1967 period. It is possible that runoff from urbanization, return flows (originating from surface or ground water), or precipitation may be increasing, but the increase has not been detected by the statistical test.

The chapter presents information on water quality and nutrient loading of the Guadalupe Estuary. In general, nutrient conditions seem to have been fairly stable during the period of record. Nutrient budgets for nitrogen, phosphorus, and carbon are included, based on extensive monitoring program measurements and modeling results from the TXBLEND hydrodynamic model. Budgets were prepared for a high- and low-flow year. Loading during the high-flow year was several times greater than during the low-flow year, as expected. Export of nutrients to the Gulf was also several times higher during the high-flow year. There were differences in import and export of nitrogen and phosphorus that may be related to differences in the biochemical processes that trap and release these materials. The proportion of phosphorus retained by the estuary was higher for a low-flow year than for a high-flow year, while the estuary retained proportionally more nitrogen in the high-flow year than in the low-flow year.

More than half of the nitrogen flowing into the estuary from the river basin was lost to geochemical sinks during the low-flow year, but only about 16% was lost to sinks during the high-flow year. While none of the macronutrients appeared to be in short supply for plankton and macrophyte production, there was a greater excess of carbon and phosphorus than nitrogen. Consequently, nitrogen is closer to being a limiting nutrient than the other materials. Using inflow volumes and the nitrogen budget, a minimum freshwater inflow requirement of 286,000 acre-ft/yr was calculated that would be needed to offset the losses of nitrogen to biogeochemical sinks.

The effects of inflow on sediment loading in the Guadalupe Estuary are assessed in Chapter 7. The lower delta that juts into San Antonio Bay below Mission Lake is undergoing an inevitable phase of decay and subsidence as the site of deposition has changed to Mission Lake. The upper portion of the delta above Mission Lake, which has existed in its present form for 2,000 years, shows no evidence of subsidence or deterioration. Sediment transport and deposition throughout Guadalupe, Hynes, and San Antonio bays is not understood well enough to allow a sediment requirement to be estimated for these areas. Consequently, a minimum sediment load estimate was determined for Mission Lake, to offset the effects of relative sea-level rise on the lake bottom and to preserve the Traylor Cut subdelta, which is currently under construction. A sediment load of 132 acre-ft/yr of sediment must be transported to the estuary, which requires an annual freshwater inflow of 355,235 acre-ft/yr.

The biological community of the Guadalupe Estuary is also described in Chapter 7. There are 29 dominant plant species, 12 to 31 dominant benthic species (which vary from year to year) and 30 dominant finfish and shellfish species. Based on salinity characteristics, the estuary could be divided into four zones: the upper estuary including Mission Lake, Hynes Bay, Guadalupe Bay, and the upper portion of San Antonio Bay; a mid-estuary region including San Antonio Bay down to the Intracoastal Waterway; a lower estuary area including the lower third of San Antonio Bay and Ayres Bay; and Espiritu Santo Bay. The species that characterize each of these regions are discussed and the general salinity requirements for each area are presented. Salinity requirements were based on a combination of physiological tolerances from experimental studies and occurrence information from estuarine sampling programs.

Chapter 8. This chapter presents the two analytical techniques to be used for determining freshwater inflow needs. The first is a multiobjective optimization technique called the Texas Estuarine Mathematical Programming Model (TXEMP). This method can use stochastic representations of some of the inflow-dependent relationships. This analysis method uses quantitative inflow-salinity and inflow-harvest relationships as well as salinity limits for various regions of the bay, historical inflow information, harvest targets, nutrient and sediment inflow requirements, and other information to calculate month-by-month inflow needs based on predefined management objectives. The effects on fishery harvest of using different objectives and varying constraints are presented in the chapter. The results from the example analyses show how important it is for resource managers and regulators to select appropriate objective functions and constraints in determining freshwater inflow requirements.

The TXBLEND hydrodynamic and conservative transport model is presented as a check of the inflow requirements. Compared to the hydrodynamic model, the inflow-salinity equations used in the analytical method described above are crude but quick predictors of salinity. The calculated inflows from the TXEMP Model are used as inputs to the hydrodynamic model. Circulation and salinity patterns are calculated for the estuary using these inputs to determine whether salinity levels remain within the bounds necessary for maintenance of productivity. The example analysis uses calculated inflows and weather and tidal conditions from a dry year (1984). The inflow quantities calculated through this method are compared to three previous inflow estimates for the estuary that use various estimation methods.

Chapter 9. The final chapter of the report reviews the objectives that are presented in Chapter 2 and discusses how each objective was fulfilled in the study. In addition, general conclusions that can be drawn from the special studies and analyses are presented with special emphasis on the effects of freshwater inflow on salinity, sediment, and nutrient loading, and maintenance of productivity of fish, shellfish, and other estuarine life.

The final chapter also describes the state of the analytical tools and the supporting information needed to apply the assessment method to the other Texas estuaries. Some enhancements and additions to the technique are presented for future improvement. The last portion of the chapter reviews the decisions that must be made by policy makers and regulators to allow the assessment method to be applied. Some of the decisions include overall management objectives in terms of species to be harvested and the relative value of one species compared to another; inflow bounds based on historical inflow records; salinity limits within different regions of the estuary; minimum nutrient and sediment loading requirements; fishery harvest targets; and chance constraints on statistical uncertainties.

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ABBREVIATIONS AND SYMBOLS

AASHTO	American Association of State Highway and Transportation Officials
acre-ft	acre-foot
Adj. R^2	adjusted R^2
ANOVA	analysis of variance
Board	Texas Water Development Board
BP	before present
C	carbon
cfs	cubic feet per second
Commission	Texas Water Commission
CPUE	catches per unit effort
CRWR	Center for Research in Water Resources, The University of Texas at Austin
Department	Texas Parks and Wildlife Department
DIN	dissolved inorganic carbon
E	Einsteins (0.004 erg s x frequency)
ELMR	Estuarine Living Marine Resources Program
Fisher's PLSD	Fisher's protected least significant difference
FL	fork length
ft	foot
GIWW	Gulf Intracoastal Waterway
gm	gram
GSI	gonadosomatic index
h	hour
ha	hectare
kg	kilogram
l	liter
m	meter
mg	milligram
mi	mile
mm	millimeter
mg/l	milligram per liter
mt	metric ton
μgm/l	microgram per liter
μm	micrometer
N	nitrogen
NMFS	National Marine Fisheries Service
P	probability or phosphorous
pp.	pages
pers. comm.	personal communication
POM	particulate organic matter
ppt	parts per thousand (salinity)
R	correlation coefficient
s	second
S.E. Coeff.	standard error of the coefficient
S.E. Est.	standard error of the estimate
SCS	Soil Conservation Service
St. D.	standard deviation
TDWR	Texas Department of Water Resources
TL	total length
TN	total nitrogen
TNRCC	Texas Natural Resource Conservation Commission

TOC	total organic carbon
TP	total phosphorus
TPWD	Texas Parks and Wildlife Department
TWC	Texas Water Commission
TWDB	Texas Water Development Board
USFWS	U.S. Fish and Wildlife Service
USGS	U.S. Geological Survey
yr	year
‰	parts per thousand (salinity)

CONVERSION TABLE

Multiply	By	To Obtain
Length:		
centimeters (cm)	0.3937	inches
inches	2.54	centimeters
feet (ft)	0.3048	meters
kilometers (km)	0.6214	miles
meters (m)	3.281	feet
miles (mi)	1.609	kilometers
Area:		
acres	43,560	square feet
acres	0.4047	hectares
hectares (ha)	10,000	square meters
square feet (ft^2)	0.0929	square meters
square kilometers (km^2)	0.3861	square miles
square miles (mi^2)	2.590	square kilometers
Volume:		
acre-feet (acre-ft)	43,560	cubic feet
acre-feet	1,233.4819	cubic meters
acre-feet	325,851.445	gallons
cubic feet (ft^3)	0.02832	cubic meters
cubic meters (m^3)	0.0008110	acre-feet
cubic meters	35.31	cubic feet
gallons (gal)	3.785	liters
liters (l)	0.2642	gallons
Flow:		
acre-feet per day (acre-ft/d)	0.5041667	cubic feet per second
acre-feet per day	0.325851445	million gallons per day
cubic feet per second (cfs)	1.983471	acre-feet per day
cubic meters per second (m^3/s)	70.0626	acre-feet per day
million gallons per day (mgd)	0.04381	cubic meters per second
million gallons per day	3.0688832	acre-feet per day
Weight:		
grams (gm)	0.03527	ounces
kilograms (kg)	2.205	pounds
metric tons (t)	2,205.0	pounds
metric tons	1.102	short tons
ounces (oz)	283.5	milligrams
pounds (lb)	0.4536	kilograms
short tons (ton)	0.9070	metric tons
Temperature:		
Celsius degrees (°C)	1.8 * (°C) + 32	Fahrenheit degrees
Fahrenheit degrees (°F)	5/9 * (°F - 32)	Celsius degrees

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This report was prepared under the authority of the TEXAS WATER CODE 16.058(a) which directs the Texas Water Development Board (TWDB) and Texas Parks and Wildlife Department (TPWD) to share joint responsibility for establishing and maintaining a continuous bay and estuary data collection and evaluation program, and to conduct studies and analyses for determining bay conditions necessary to support a sound ecological environment. Additional assistance was provided by the staff of the Texas Natural Resource Conservation Commission (TNRCC) and individuals from federal agencies, foundations, and universities. The following persons participated in the preparation of this report:

Project Supervisors:	Gary L. Powell (TWDB) and Albert W. Green (TPWD)
Report Editor:	William L. Longley (TWDB)
Editing Assistance:	Cindy Loeffler (TPWD) Kathy Mills (TWDB)
Word Processing:	Donna Tiemann (TWDB) Bonnie Smith (TPWD)
Computer Graphics:	Roger Wolff (TWDB) David A. Brock (TWDB) Greg Malstaff (TWDB) Ruben S. Solis (TWDB) Junji Matsumoto (TWDB) William L. Longley (TWDB) Juan Vega (TWDB) Anita M. Morgan (TPWD) Carl D. German (TPWD)
Data Assistance:	John Berry (TWDB) Tom Martin (TWDB) Terrell Robison (TWDB) Yixing Bao (TWDB)
Page Layout:	Zelphia Bloodworth (TWDB) Mike McCathern (TWDB)
Reviewers:	Norman Boyd (TPWD) Hal Osburn (TPWD) C.E. Bryan (TPWD) Larry McEachron (TPWD) Craig S. McColloch (TNRCC) Bruce Moulton (TNRCC)

Reviewers (cont'd.):

Martin Munroe (TNRCC)
Frank S. Shipley (TNRCC)
George M. Staff (TNRCC)
Ken Dunton (University of Texas at Austin)
Paul Montagna (University of Texas at Austin)
Terry E. Whitledge (University of Texas at Austin)
Armando A. de la Cruz (Mississippi State University)
John D. McEachran (Texas A&M University)
Ned P. Smith (Harbor Branch Foundation)
Roger Zimmerman (National Marine Fisheries Service)

Chapter and Section Authors:

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CHAPTER 1: HISTORICAL PERSPECTIVE ON FRESHWATER INFLOWS

1.0 INTRODUCTION

The inflow of freshwater is widely recognized as an essential factor influencing the biological productivity of estuarine (tidal) areas as diverse as the Black Sea (Rozengurt and Haydock 1981), the Caspian Sea (Rozengurt and Hedgpeth 1989), the Nile Delta (Ben-Tuvia 1973; Halim 1975; Mancy 1979); the Gulf of St. Lawrence (Sutcliffe 1972, 1973), San Francisco Bay (Turner and Chadwick 1972; Stevens 1979; Herrgesell 1983), Chesapeake Bay (Pearson 1948; Shea et al. 1980; Ulanowicz et al. 1982), and the bays and estuaries of the Gulf of Mexico (Chapman 1966; Copeland 1966; Copeland et al. 1972; Hackney 1978; Schroeder 1978; Stone et al. 1978; Powell 1979; Texas Department of Water Resources 1982c).

Functional role of freshwater inflow. Freshwater inflow affects estuaries at all basic levels of interaction; that is, with physical, chemical, and biological effects. The functional role of freshwater inflow in the ecology of estuarine environments has been scientifically reviewed (Snedaker et al. 1977; Hackney 1978; Texas Department of Water Resources 1982c; Skreslet 1986), and the effects on these living coastal systems were found to include but may not be limited to:

1. Dilution of seawater to brackish conditions;
2. Dilution and transport of harmful materials and contaminants;
3. Creation and maintenance of low salinity nursery habitats which provide food and cover to juvenile fish, shrimp, crabs, oysters, and other biota;
4. Moderation of bay water temperatures;
5. Reduction of metabolic stresses and the energy required for osmoregulation (regulation of internal body salts) in estuarine-dependent organisms;
6. Provision of a medium for the transport of beneficial sediments and nutrients, the biogeochemical cycling of essential primary nutrients (carbon, phosphorus, and nitrogen), and the removal of metabolic waste products from living organisms;
7. Modification of concentration-dependent chemical reactions, ion-exchange, and flocculation (coagulation and precipitation) of particles in the saltwater environment;
8. Creation of a resource partitioning mechanism among estuarine plants and animals as a result of the combined effects of inflow on salinity, temperature, and turbidity of bay waters;
9. Distribution (horizontal displacement) and vertical movement of organisms in the water column related to the stimulation (release) of a positive phototaxic or negative geotaxic behavioral response;
10. Creation of a cutting and filling mechanism that affects both erosion and deposition in the bays and estuaries;

11. Creation of a salt-wedge and mixing zone in concert with tidal action from the ocean;

12. Transportation of allochthonous (external) nutritive materials (organic detritus from decaying plant and animal tissues) into bays and estuaries as a function of land surface topography, amount of rainfall, and size of the drainage area;

13. Migration (timing of arrivals and departures) and orientation (direction of movement) of migratory organisms like the penaeid shrimps and many marine fishes; and

14. Stimulation of some plants and animals that may be considered less desirable or even a nuisance to man such as the plant-like "red tide" organism, the Eurasian water milfoil, the South American water hyacinth, and the Chinese grass carp.

The dynamic nature of estuaries. Nevertheless, this does not mean that estuarine needs for freshwater inflow are in some way constant or uniform. In fact, dynamic inflow fluctuations within the productive range, both seasonally and annually, are realistic and desirable for Texas bays and estuaries. Moreover, the seasonal timing of freshwater inflows is most important because adequate inflows during critical periods of reproduction and growth can produce greater benefits than constant inflow throughout the year. However, extended or semi-permanent inflow conditions which consistently fall below maintenance levels can lead to degraded estuarine environments, loss of important nursery areas for the young of economically valuable fish and shellfish (seafood) resources, and a reduction in the tremendous potential for natural assimilation of organic and nutritive wastes produced by man's activities (i.e., municipal, industrial, and agricultural wastes).

Effects of large-scale weather patterns on inflow. Perhaps the most dramatic reductions of freshwater inflow occur in connection with large-scale changes in weather patterns producing drought. Although the 22-year Hale double-sunspot cycle and the 18.6-year lunar nodal cycle seem related to periods of drought, high-resolution frequency analysis and statistical testing of tree-ring data from the past 300 years indicates that a recurrence interval of approximately 20 years for major droughts is too weak and irregular for use in forecasting (Meko 1985). Climatic cycles exhibit observed tendencies for either wet or dry years to occur in clusters, and for several individual years in a cluster to contain particularly extreme (flood or drought) conditions. This phenomenon has been described by Mandlebrot and Wallis (1968) as the "Noah and Joseph effect," named for events in the lives of the two famous biblical personalities. In this view, the occurrence of a number of hot, dry years during the decade of the 1980's after several wet years in the 1970's has been just a return to the "normal" cycle for a semi-arid state like Texas.

Freshwater inflow reductions due to human activities. Other causes of reduced freshwater inflow include the impoundment and diversion activities of man. The construction of dams on streams and rivers impounds state waters and creates multipurpose lakes and reservoirs. As a first step in the systems analysis of environmental impacts from dam construction, Darnell et al. (1976) proposed a conceptual sequence of impacts on coastal estuaries that includes the major physical and chemical effects that are possible (Figure 1.0.1). The actual impacts will vary from reservoir to reservoir, but many can be successfully reduced or mitigated, at least in part, through the use of optimization techniques to improve the operating rules for any necessary water resources projects. These optimization techniques can be used to create reservoir operating plans which will cause the least harm to the environment while allowing maximum beneficial use of state waters.

Diversion and beneficial use of impounded water, free-flowing surface water, and pumped groundwater in Texas have been an essential part of the development and growth of the state economy, enabling it to grow to a size matched by only a few nations during the last half of this century. Furthermore, water use can produce important beneficial flows by providing a substantial streamflow base during dry seasons when little natural flow may occur in Texas, whereas some water conservation and reuse activities can actually reduce freshwater inflows by reducing the amount of water used and discharged back to the state's waterways as wastewater return flows.

Effects of reduced freshwater inflows. The major effects associated with loss of inflow due to droughts, dams, or diversions of freshwater have been observed to include, but may not be limited to:

1. Increased salinity of bay, estuary, and neritic (nearshore) marine waters;

2. Reduced mixing due to salinity differences and stratification of the water column;

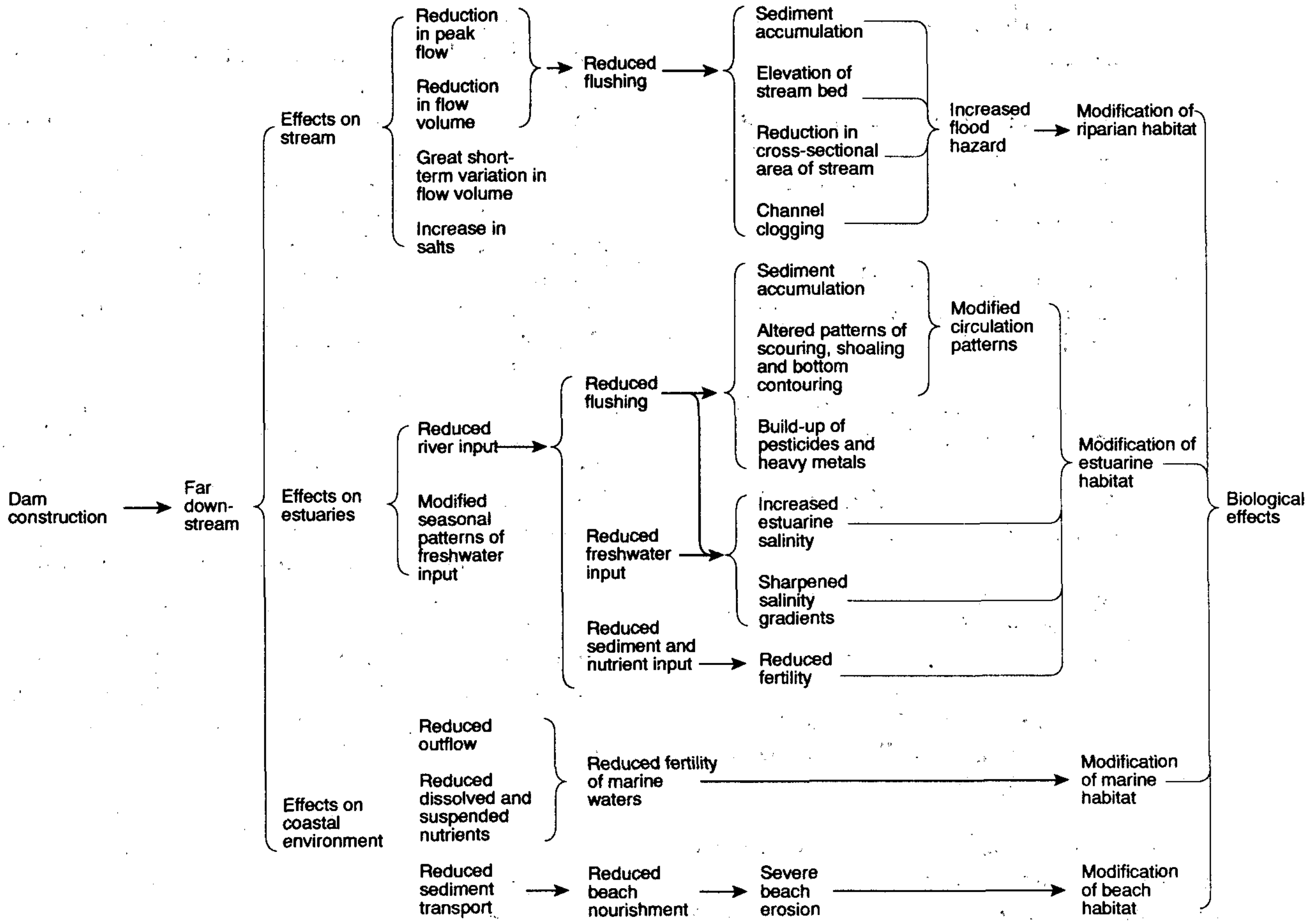


Figure 1.0.1. Factor train analysis of the far downstream (including estuarine and marine) effects of dam construction on wetlands. Only the major physical and chemical events are presented. After Darnell et al. (1976).

3. Penetration of the salt-wedge farther upstream allowing greater intrusion of marine predators, parasites, and diseases;

4. Saltwater intrusion into coastal ground- and surface water resources used by man;

5. Diminished supply of essential nutrients to the estuary from inland or local terrestrial origins;

6. Increased frequency of benthic (bottom) sediments becoming anaerobic (without oxygen), liberation of toxic heavy metals into the water column that had been sequestered in the benthic substrates, and sulphur cycle domination;

7. Reduced inputs of particulates and soluble organic matter with flocculation and deposition of the particles locally rather than being more widely dispersed throughout the estuarine ecosystem;

8. Loss of economically important seafood harvests from coastal fisheries' species for a variety of reasons related to high salinity conditions, reduced food supply, and loss of nursery habitats for the young;

9. Loss of characteristic dominance of euryhaline (widely salt-tolerant) species in the bays and estuaries to stenohaline (narrowly salt-tolerant) species as natural selection occurs for species more fully adapted to marine conditions in general;

10. Increased populations of salt-tolerant mosquitos and flies;

11. Increased incidence of human diseases such as cholera caused by the bacteria *vibrio cholerae* in improperly cooked seafood;

12. Deterioration of salt marshes, mangrove stands, and seagrass beds if under constantly elevated salinities;

13. Loss of sand/silt renourishment of banks and shoals resulting in erosion;

14. Alteration of littoral drift and nearshore circulation patterns; and

15. Aggravation of all negative effects during low-flow (drought) periods with increasing severity as the frequency of occurrence increases (Odum 1970; Snedaker et al. 1977; Hackney 1978; Texas Department of Water Resources 1982c; Skreslet 1985).

The crucial need for freshwater inflows to Texas bays, estuaries, and their economically important fishery resources was first recognized by Hildebrand and Gunter (1953). At that time, virtually all parts of the state were experiencing the effects of one of the most severe droughts in modern history. Beginning in 1948, the drought was finally broken by heavy rains and flooding in the spring of 1957. During 1956, the worst year of the decade-long drought, combined river discharges measured at the last streamflow gaging station on each major Texas river amounted to only 4.1 million acre-feet, or about 14% of the average annual freshwater inflows to the state's bays and estuaries. As a result of the drought, bay oyster (*Crassostrea virginica*) production in Texas practically ceased, white shrimp (*Penaeus setiferus*) harvests were drastically reduced, and estuarine-dependent fishes such as the black drum (*Pogonias cromis*) were blinded and exhibited body lesions from extreme high salinity stress (Simmons and Breuer 1962).

1.1 LEGISLATIVE DIRECTIVES

Creation of a state-wide water plan. Because the 1950's drought ranked among the most severe of the past 400 years (Texas Water Development Board 1968), there was a clear stimulus for legislative action. In response, the 55th State Legislature enacted the Texas Water Planning Act of 1957 in special session. This act, followed by several related acts and amendments over the next ten years, ultimately resulted in the creation of a Texas Water Plan by the Texas Water Development Board (TWDB). The plan was published in November 1968 and formally adopted in 1969 as a flexible guide to the conservation and development of the state's water resources. Because of specific legislative directives (Section 16.051, Texas Water Code), the plan was to give consideration to the effect of upstream development on the bays, estuaries, and arms of the Gulf of Mexico. Consequently, the first Texas Water Plan called for an estimated 2.5 million acre-feet of supplemental freshwater inflows annually to Texas bays and estuaries, and a coastal canal system to facilitate distribution of surplus water among the 11 major river basins that flow to the coast, ten of which have headwaters that originate entirely within the state. Although voters declined to approve funding for parts of this first state effort, the TWDB regularly updates the plan to ensure use of the best available information in meeting today's water needs and planning for the future. The current version of the plan emphasizes water supply, treatment, distribution, conservation, and the collection and treatment of wastewaters; however, the coastal canal system proposed in the past is not included in the present plan.

Early cooperative interagency studies. In preparation of the first plan, the TWDB also initiated a cooperative Bays and Estuaries Program in 1967 for the purpose of collecting the physical, chemical, and biological data necessary for state water planning. Activities under the Program expanded in 1975 when the 64th State Legislature enacted Senate Bill 137 which required comprehensive studies of the effects of freshwater inflows on the bays and estuaries to be performed in cooperation with the Texas Water Rights Commission, Texas Water Quality Board, General Land Office, Texas Parks and Wildlife Department (TPWD), and the Texas Coastal and Marine Council. This early round of cooperative interagency studies was completed by December 31, 1979.

Total expenditures by the TWDB and the Texas Department of Water Resources (formed by the consolidation of the TWDB, Water Rights Commission, and Water Quality Board in 1977) on the Bays and Estuaries Program over the 12-year period between 1967 and 1979 were estimated at $6.8 million and resulted in a series of published reports covering each of the state's seven major estuarine systems (Texas Department of Water Resources 1980a, 1980b, 1981a, 1981b, 1981c, 1981d, 1981e, 1982a, 1982b, 1982c, and 1983). Although the reports did include preliminary estimates of the freshwater inflows needed from major Texas rivers to meet several management alternatives (Table 1.1.1), many scientists, engineers, and legislators believed that the essential databases were still too limited and

Table 1.1.1.Summary of previous estimates of gaged river flows [a] in millions of acre-ft needed for Texas bays and estuaries based on 1975-1979 studies.

Estuary	Sabine-Neches	Trinity-San Jacinto	Lavaca-Colorado	Guadalupe	Mission-Aransas[b]	Nueces[c]	Laguna Madre
Average annual gaged river flows (1941-1976)	11.2	7.09	2.54	1.81	0.104	0.575	0.335
Alternative I (sustenance)[d]	5.69	4.61	1.46	1.24	0.0155	0.356	0.182
Alternative II (maintenance)[e]	no estimate possible	4.89	2.41	1.62	0.0194	0.397	0.285
Alternative III (enhancement)[f]	no estimate possible	4.75	2.44	1.83	0.0427	0.550	0.292
Alternative IV (viability limit)[g]	2.02	0.822	0.808	0.755	0.0028	0.118	0.138

[a] Freshwater inflow to each estuary as measured at the last nontidally affected gage located on each contributing river.
[b] Gaged flow only on Mission River at Refugio.
[c] Gaged Nueces River flow adjusted for diversions at Calallen just above Nueces Delta.
[d] Estimate based on salinity and delta marsh inundation needs of each estuary.
[e] Estimate based on salinity, inundation, and fisheries needs to maintain commercial harvests at average levels.
[f] Estimate based on salinity, inundation, and fisheries needs to enhance harvests of selected major commercial species.
[g] Estimate based on monthly limits of bay salinity within which economically important and ecologically characteristic fish and shellfish survive, grow, and maintain viable populations.

the estimated needs too unreliable to be used for water management and regulatory purposes. This point of view was reinforced later at the National Symposium on Freshwater Inflow to Estuaries (San Antonio, Texas, September 11-13, 1980) where several technical papers emphasized the need for much longer-term databases before any real understanding of the complex estuarine ecosystems could be developed (Cross and Williams 1981).

1984 Joint Interim Legislative Committee study. Subsequently, a Joint Interim Legislative Committee on Water Resources was formed and a public hearing was held to consider important bay and estuary issues (Houston, Texas, January 26, 1984). At that hearing, representatives from the Texas Department of Water Resources (TDWR) reported on the early freshwater inflow studies, their limitations, and the need for additional information to support decision-making and water management. The TDWR also proposed legislative language for protection of living estuarine resources in the consideration and issuance of water rights permits to impound, divert, and use state waters. The proposed bill was generally similar to controversial legislation previously debated in the 68th Legislature (1983) which had caused a major package of water plan legislation to fail; however, in one significant difference, the TDWR's proposed bill called for an appropriation of $2.8 million to finance at least three additional years of data collection that would be used to develop management plans for each of the state's major bay and estuary systems.

Authorization for the present study. Eventually, much of this language was incorporated into House Bill 2, which was successfully enacted by the 69th Texas Legislature in 1985. Further clarifying amendments (Senate Bill 683) were added during the next regular legislative session in 1987. Under this new legislation, the TWDB (formed by the split of the TDWR into the Texas Water Commission and the TWDB) and the TPWD were directed to jointly establish and maintain a continuous data collection and analytical study program to determine the bay conditions (i.e., sediments, nutrients, and salinity gradients) necessary to support a sound ecological environment. The studies were to be completed by December 31, 1989, and following publication of study results, were to be submitted to both cooperating agencies for review and comment (Section 16.058, Texas Water Code). In addition, the 1987 legislation also directs the TPWD and another regulatory agency, the Texas Water Commission (now renamed the Texas Natural Resource Conservation Commission, or TNRCC), to jointly review the studies for the purpose of determining the specific quantities and qualities of freshwater inflow which are necessary for maintaining the bays and estuaries, and to provide information necessary for water resources management (sections 11.1491 and 11.147, Texas Water Code). Again, any publication of reports on specific freshwater inflow needs completed under this section of the law by the regulatory agencies were to be submitted to both agencies for review and comment.

CHAPTER 2: STATEMENT OF PURPOSE

2.0 INTRODUCTION

An estuary is a semi-enclosed coastal body of water which has a free connection with the open sea and within which sea water is measurably diluted with fresh water derived from land drainage (Pritchard 1967b). Texas coastal plains estuaries typically include wetlands and open bay waters in which nutrients from river inflows, adjacent land runoff, and the sea support a productive community of plants and animals.

There is concern that modifications of our estuaries, particularly creation of navigational channels and alteration of inflows, have or could cause reductions in estuarine populations of fish and wildlife. Construction of the Intracoastal Waterway may have caused the Laguna Madre to become less saline and the Chenier Plains Marshes near Sabine Lake to become more saline. Alteration of seasonal inflows in the Sabine River due to the Toledo Bend Reservoir may have sharply reduced shrimp production in the Sabine Lake area. Striped bass (*Morone saxatilis*) no longer exist in Texas estuaries in harvestable quantities (Collins and Smith 1893; Gulf States Marine Fisheries Commission 1986; Quast et al. 1989). Snook (*Centropomis undecimalis*) have also dramatically declined in abundance (Matlock and Osburn 1987). Oyster (*Crassostrea virginica*) landings from Nueces Bay have essentially disappeared (Collins and Smith 1893; Quast et al. 1989), and extensive dredging of oyster beds for shell during the period 1941 through 1967 reduced available substrate for oyster populations (Quast et al. 1988) in many Texas estuaries. In addition, numbers of brown pelicans (*Pelecanus occidentalis*), Eskimo curlews (*Numenius borealis*), piping plovers (*Charadrius melodus*), and American alligators (*Alligator mississippiensis*) also have historically declined. Waterfowl populations have been dramatically reduced. Recently, however, there have been significant increases in the numbers of brown pelicans, whooping cranes (*Grus americana*), and American alligators.

The reasons for the reductions in these populations are not always clear. Overfishing, habitat destruction or alteration, and pollution have occurred and are undoubtedly contributing factors. Alteration of seasonal freshwater inflows has also had some influence, and there is concern about the quantity of inflow reaching the estuaries. Even though there have been changes in the abundance of some species, the available data suggests that Texas estuaries still retain much of their historical composition and productivity. Care needs to be exercised to ensure that future water development does not impair or contribute to the reduction of the unique natural heritage and productivity of Texas bays and estuaries.

2.1 REGULATING FRESHWATER INFLOWS TO BAYS AND ESTUARIES

Legislative Direction

The 69th Texas Legislature assigned the responsibility for water rights permitting to the Texas Natural Resource Conservation Commission (TNRCC) and gave the Texas Parks and Wildlife Department (TPWD) authority to be a party in hearings on applications for permits to store, take, or divert water—actions that can change the pattern or quantity of freshwater inflow. The Legislature directed the TNRCC to consider effects on bays and estuaries for all water rights permits, with a specific directive to include protective provisions in certain permits by applying the following performance standard when making decisions concerning water rights on rivers and streams leading to bays and estuaries:

> For permits issued within an area that is 200 river miles of the coast, to commence from the mouth of the river thence inland, the commission shall include in the

> permit, to the extent practicable when considering all public interests, those conditions considered necessary to maintain beneficial inflows to any affected bay or estuary system. TEXAS WATER CODE 11.147(b)

The performance standard was indicated by the phrase, "conditions considered necessary to maintain beneficial inflows to any affected bay or estuary system." The crucial term "beneficial inflows" was defined in greater detail in the legislation:

> In this section, "beneficial inflows" means a salinity, nutrient, and sediment loading regime adequate to maintain an ecologically sound environment in the receiving bay and estuary system that is necessary for the maintenance of productivity of economically important and ecologically characteristic sport or commercial fish and shellfish species and estuarine life upon which such fish and shellfish are dependent. TEXAS WATER CODE 11.147(a)

The Legislature also directed the Texas Water Development Board (TWDB) and the TPWD to establish and maintain a continuous data collection and evaluation program and conduct studies and analyses aimed at determining bay conditions that provide a sound ecological environment:

> The Parks and Wildlife Department and the Board shall have joint responsibility, in cooperation with other appropriate governmental agencies, to establish and maintain on a continuous basis a bay and estuary data collection and evaluation program and conduct studies and analyses to determine bay conditions necessary to support a sound ecological environment. TEXAS WATER CODE 16.058(a)

Interpreting the Legislation

An ecologically sound environment. The legislative goal is to maintain an ecologically sound environment so that it is possible to maintain the productivity of commercial and sports species, and other organisms. While the legislation did not specifically define the phrase "ecologically sound environment," an interpretation is possible. Conceptually, an estuary can be considered to be ecologically sound when the typical physical, chemical, and biological parameters that are measured—including the characteristic biological communities—fall within the range of values that historically occurred before humans interfered with natural processes (e.g., by constructing waterways, introducing pollutants, and altering freshwater inflows). Operationally, an ecologically sound estuarine environment can be defined as one having densities of animals and plants not significantly different from the historical patterns of abundance or composition. In practice, this may be difficult unless the physical and chemical properties are sufficiently close to historical conditions so that the biotic communities are not degraded. Adoption of this definition by resource managers should result in the retention of the estuary's biological diversity and ensure that the estuarine system remains resilient to natural and man-made disturbances. Disturbance is a factor of life on the Texas coast—intense hurricances occasionally hit the coast, and hard freezes, droughts, floods, and chemical spills also can dramatically affect estuarine habitats.

The legislation specifically mentions that production of fishery species is to be maintained. This implies that more than just a remnant population is to be present. The population should support an economically harvestable surplus.

Other characteristics of ecological soundness. An ecologically sound estuarine environment is characterized by having several trophic levels through which nutrients are routinely cycled (Odum 1959; Perkins 1974). Texas estuaries have several trophic levels that include harvestable quantities of predatory fish at the top level; shrimp, crabs, and oysters, which function at the middle trophic level and are also valuable as food; and an abundance of lower-trophic-level consumers on which all the animals in the higher trophic levels depend. It is the presence of these groups, in combination with the large numbers of algae and other plant species in the estuary which creates a food-web capable of sustaining those relatively few species important to man for food and recreation. These estuarine animals vary in abundance among the Texas estuaries depending on differences in the physical, chemical, and biological parameters in each system. In addition to these estuarine species, a variety of amphibians, reptiles, migratory waterfowl, shorebirds, and mammals are associated with adjacent wetlands and intertidal areas.

Necessary and sufficient conditions to maintain productivity. The legislation assumed that maintaining an ecologically sound environment is a necessary condition to maintain the productivity of various fish and shellfish species and the estuarine organisms on which they depend. However, maintaining an ecologically sound environment is not sufficient by itself to guarantee maintenance of productivity. Overfishing or catastrophic chemical spills are examples of human activities that could cause productivity declines even if an ecologically sound environment had been maintained. Regulation of fishing, navigation, and hazardous material handling are some of the kinds of managerial measures that must be used to build upon an ecologically sound environment to maintain productivity of the desired species.

Managing for an ecologically sound environment. A primary management method to achieve the legislative goal with respect to freshwater inflows is to incorporate special conditions in state permits to store, take, or divert water. In general, these conditions will regulate the quantity and timing of the permitted water use. The legislation recognized that the dilution of marine water by fresh water and the supply of nutrients and sediments were the three major influences that rivers and streams have on estuaries. The quantity and pattern of freshwater inflows over time is the normal mechanism that regulates the salinity of estuarine waters and the inflow of nutrients and sediments. Therefore, special conditions in water rights permits have to be designed so that the salinity and nutrient levels and sediment supplies are adequate over time to provide an environment in which the production of estuarine organisms may be maintained. In addition to managing the flows in rivers and streams, regulating the quality and quantity of wastewater discharges for the benefit of the state's estuaries must become a recognized strategy.

Summarizing the legislation. Paraphrasing the logical sequence of statements from the legislation: (1) the TNRCC has the authority through the issuance of permits to control uses of fresh water that normally flows into the bays and estuaries; (2) where practicable, the Commission shall include special inflow conditions in the permits; (3) the conditions will be based on consideration of the effect fresh water has on the salinity, nutrient, and sediment loading regime of the receiving estuary; and (4) the salinity, nutrient, and sediment loading regime shall be adequate to maintain an ecologically sound environment in which the productivity of various target species can be maintained.

2.2 PLANNING THE ENGINEERING AND ECOLOGICAL STUDIES

Implications of the Legislation

The sequence of logic specified in the legislation has several consequences for the engineering and ecological studies. First, the studies must investigate the relationships between freshwater inflow and salinity, nutrient, and sediment loading regimes. Otherwise, it will be impossible to relate the quantitative inflow conditions specified in the permits with the appropriate salinity, nutrient, and sediment loading regimes.

It will also be necessary to rationalize and describe the bounds of various environmental variables that define an ecologically sound environment. These determinations will have to be made on a bay-by-bay basis and include information about historical patterns of inflow and other variables, as well as the physiological and habitat requirements of various species or biological communities.

It will be necessary to provide analytical methods to identify or calculate quantities and patterns of inflow that supply the salinity, nutrient, and sediment loading regimes that are adequate to maintain an ecologically sound environment. The methods will have to deal with the direct relationships between inflow and salinity, nutrient, and sediment loading. They also will have to include the effects of other variables (e.g., tides and local weather conditions) not controlled by inflow but which influence salinity, nutrient, and sediment loading, while still producing results that fall within the range of values that satisfy the ecological goal of the legislation.

Goals for the Study

With these requirements in mind, the following three goals were identified and agreed on by staff representatives of the TWDB, TPWD, and TNRCC:

Goal 1. To provide qualitative and specific quantitative relationships among freshwater inflows and selected physical, chemical, and biological processes involved in the productivity of coastal bays and estuaries.

Goal 2. To provide information about impacts of normal inflow variations (drought to flood conditions) on bay environments and their living resources.

Goal 3. To provide state-of-the-art tools for addressing decision-makers' questions about the impacts of water development, as well as other human activities, on the bays and estuaries.

Specific Study Objectives

A set of explicit program objectives were defined to allow specific studies and analysis to be planned and performed. Studies were undertaken by the TWDB and the TPWD. In addition, a variety of studies and services were performed by other state and federal agencies and universities through interagency contracts with the TWDB and TPWD which were funded by the TWDB's Water Research and Planning Fund. Participating institutions included: the TNRCC, Texas Department of Health, National Marine Fisheries Service, University of Texas at Austin, Texas A&M University, University of Houston, and Texas A&M University at Corpus Christi. The specific objectives for these studies were:

Objective 1. To compile freshwater inflow, bay hydrographic, and biological data into computer-compatible format files.

Objective 2. To develop circulation and salinity models for Texas bays, including finite-element mathematical models of estuarine hydrodynamics and conservative mass transport, as well as statistical salinity-inflow regression equations.

Objective 3. To evaluate effects of salinity and salinity change on estuarine plants and animals. This would include marine bacteria; phytoplankton; benthic algae; vascular macrophytes; zooplankton; benthic infauna; and fish and shellfish larvae, juveniles, subadults, and reproductive adults. Also, analyses of fishery-independent data are a part of this evaluation.

Objective 4. To assess water quality trends over the last two decades, including correlation of antecedent inflow conditions with the concentrations of selected chemical and water quality parameters.

Objective 5. To determine effects of freshwater inflows on river deltas and bay sedimentation, including sediment loadings, whether effects are continuous or episodic, and how this relates to estuarine maintenance.

Objective 6. To evaluate effects of freshwater inflows on estuarine primary (plant) production. This would include effects other than direct salinity effects, such as light limitation (turbidity), nutrient loading, and the biogeochemical cycling of essential nutrients in estuaries.

Objective 7. To develop statistical harvest-inflow regression equations for commercial catch of estuarine-dependent fisheries.

Objective 8. To develop a methodology to define objective functions and constraints for use with optimization procedures, such as mathematical or dynamic programming models, and to perform example analyses.

Each of these objectives became topics for individual studies. The final reports from these studies constitute the technical appendices to this report. These studies provide a wide base of information about the organization and productivity of four major Texas estuaries. This base of information has been drawn on extensively to provide the data, functional relationships, and quantitative relationships used in this report.

Other Objectives

Five other objectives were identified by the TWDB, TPWD, and TNRCC as suggested by the legislative requirements (TEXAS WATER CODE 11.1491). These additional objectives are beyond the scope of the present study required under Section 16.058. In general, they involve the consideration of management objectives for each bay system, the application and verification of the new methodologies to each major bay and estuary in Texas, and the establishment of estuarine management councils.

Objective 9. To develop state management objectives and constraints for use with the new optimization procedures.

Objective 10. To perform optimization analyses and develop estimates of freshwater inflow needs for each major bay and estuary system over a range of conditions (short-term or instantaneous requirements versus long-term ecosystem needs).

Objective 11. To review and validate inflow relationships using existing data, as well as any new data.

Objective 12. To continue a data collection program and update or revise inflow estimates as necessary during 1990-1995. This would include changes made necessary by large-scale modifications such as the direct diversion of the Colorado River into West Matagorda Bay, the opening or closing of Gulf inlet passes like Cedar Bayou and Yarborough Pass, and major navigation and development projects in estuarine areas like Galveston Bay and Corpus Christi Bay.

Objective 13. To establish Estuary Management Councils for each principal bay and estuary system, and provide technical assistance as requested in their efforts to develop alternative water management methods to meet the estimated needs for maintaining an ecologically sound coastal environment.

2.3 SUMMARY

The goal of the 1985 legislation concerning freshwater inflow to estuaries is to ensure inflow conditions that provide an ecologically sound environment in which the productivity of commercial and sports species, and other organisms on which they depend, can be maintained. The means to achieve this goal is to evaluate the effect each permit application would have on downstream water quality, instream uses, fish and wildlife habitats, and bays and estuaries. Permits within 200 river miles of the coast would include special conditions that will provide a salinity, nutrient, and sediment loading regime in the bay that maintains an ecologically sound environment for these plants and animals. In 1985, the Legislature appropriated 5% of the annual firm yield to the Department for releases specifically for bays and estuaries from any reservoir built with state financial participation within the 200-river-mile region. The firm yield of a reservoir is that amount of water that can be diverted from the reservoir during the critical drought of record without incurring a shortage. The firm yield is calculated using a computer model to simulate reservoir operations assuming historical inflows.

To achieve the legislative goal, the state agency staffs cooperatively determined that it was necessary to: investigate the relationships between inflow and salinity, nutrient, and sediment loading; determine the conditions required for an ecologically sound environment; and prepare analytical methods that could be used in quantifying freshwater inflow needs. Eight specific study objectives were defined and five additional objectives were identified that must be fulfilled before the analytical methods can become operational.

CHAPTER 3: ANALYTICAL APPROACH

3.0 BACKGROUND

Prerequisites for the Analytical Procedure

An essential task of the bay and estuary studies was to prepare an analytical procedure that estimates the quantity and timing of freshwater inflows to maintain an ecologically sound environment so productivity of target species is maintained. This procedure had to satisfy several requirements:

Provide quantitative results. The procedure must provide quantitative estimates of inflow volumes. The TNRCC must make decisions about quantities of water that applicants propose to divert from rivers and streams. Ultimately, the TNRCC must compare the amount of water that will be diverted with the average stream flow, and decide whether the amount of water remaining after diversion and the temporal pattern of inflow is sufficient to maintain an ecologically sound environment and bay productivity. Only quantitative results will provide the information needed for inflow management decisions.

Give inflow estimates appropriate for management actions. Some of the inflow needs of fish, shellfish, and other target species are related to osmotic requirements or preferences during particular stages of the organisms' life cycles; other inflow needs may be related to environmental cues, nutrient demands, or other requirements that are not well understood. Many target species have life cycles of a year or less; some fish and shellfish live for several years but have reproductive cycles timed to particular periods of the year. Since the inflow requirements of these organisms differ from season to season or month to month, the inflow estimates from the analytical procedure must be more detailed than simply annual inflow volumes. Monthly or bimonthly inflow volumes are an appropriate time scale for specifying inflows to satisfy the physiological and ecological needs of the organisms.

Historical data on stream flow, precipitation, and ungaged runoff are available on a daily basis, and are easily summed to monthly totals. Permits for diversions already use monthly and daily volumes or flow rates in specifying permissible diversion quantities; self-reporting data from permittees on return flow and diversions are monthly values. Since the regulatory mechanisms and management information for controlling inflows already operate on a monthly time frame, and monthly inflow volumes are appropriate for satisfying the physiological and ecological needs of the target species, the analytical procedure should provide monthly estimates for inflow management.

Supply attainable inflow estimates. Inflow is largely dependent on rainfall throughout a river basin; rainfall is not under human control. The estimated inflow volumes from the analytical procedure must be appropriate for the river basin and estuary system being studied. It would make no sense to require inflows greater than normally occur as shown by naturalized streamflow records (the natural flows if diversions and return flows did not exist); rainfall could not deliver these quantities even if all human uses were suspended.

Approximately 6.5 million acre-ft of fresh water was diverted from streams flowing to Texas estuaries in 1980 (TDWR 1984)—about 15% of the average runoff carried by streams that flow to Texas estuaries. This was the year of highest surface-water use on record; it has declined since 1980 due to a significant reduction in irrigation. Under average conditions, existing impoundments and diversions upstream of estuaries capture only a small proportion of the streamflow to the coast; during drought years, the amount of the streamflow reaching the estuaries will be reduced due to decreased runoff and increases in diversion, impoundment, and evaporation. Consequently, the monthly inflow patterns computed by the analytical procedure must, to a great extent, reflect the basin's historical streamflow pattern since the volume and pattern of inflow are largely the result of the amount and timing of basin rainfall. It would be inappropriate for the month-to-month inflow pattern computed by the analytical procedure to be radically different than naturalized streamflow records due to rainfall unless upstream impoundments had adequate capacity and releases were managed specifically to satisfy the freshwater inflow needs of the estuaries.

Address estuarine life and relationships identified by statute. Legislation setting state policy in Texas for freshwater inflow management declares that the goal for management is to maintain an ecologically sound environment so that it is possible to maintain productivity of commercial and sports species, and other organisms. The analytical procedure should focus on these target species (assuming this effort will maintain characteristic estuarine communities) and, where possible, apply quantitative estimates of their productivity to ensure that an objective measure is used to relate inflow and productivity.

The legislation defines beneficial inflows to be a salinity, nutrient, and sediment loading regime adequate to maintain an ecologically sound environment in the receiving bay and estuary. Since the legislation specifically addresses salinity, and nutrient and sediment loading as the mechanisms for ensuring an ecologically sound environment, the analytical procedure should specifically encompass these factors in the methodology.

Offer easy addition of quantitative relationships and information. The analytical procedure should be built around the target species and beneficial inflow relationships described above. It should be adaptable so that new quantitative relationships involving inflow or new information concerning hydrology, productivity, nutrients, sediment, or other pertinent factors can be readily incorporated without requiring complete redesign.

Allow efficiency and flexibility in operation. The procedure should be a useful tool to decision makers. It should be efficient in its operation so that results are available quickly. It should also be flexible so inflow estimates can be determined over a range of conditions and management objectives. In this manner, decision makers can attain an understanding of the bounds within which they have latitude to make inflow decisions.

Provide for checks of method. While the procedure should give results in terms of inflow volumes that satisfy the requirements, there should be an independent check on the results to confirm that the bay and estuary environment remains ecologically sound as specified in the legislation. If the results of the analytical procedure do not satisfy the independent check, the procedure should be flexible so changes can be made in the analysis and an acceptable result confirmed.

3.1 STEPS IN THE ANALYTICAL APPROACH

This set of specifications suggested an ordered analytical approach to this problem. The elements described below were accomplished through a combination of studies contracted with state and federal agencies and universities, and completed by the staffs of the TPWD and TWDB.

Coastal Hydrology

Extend base of information from 1977 through 1987. Basic to all aspects of this problem was coastal hydrology. The first step was to prepare updated hydrology information, extending the original data set—covering the period 1941 through 1976—through 1987. This includes data on gaged and ungaged inflow, as well as evaporation and direct precipitation on the estuary. Diversions and return flow information for ungaged areas was more extensive for the 1977 through 1987 period than in earlier studies.

Improve ungaged hydrology methods. Earlier freshwater inflow studies used a monthly water yield model for ungaged areas based on daily precipitation, Soil Conservation Service curve numbers (Williams and LaSeur 1976), and a soil depletion index to compute runoff from small watersheds. The model was calibrated by using rainfall-runoff data from gaged watersheds. While this model gave fairly good results, a refined model was prepared that improved overall runoff estimates and provided daily estimates.

Provide data for use in studies. The hydrology data provided the basis for many of the analyses by contractors and agency staff. In particular, the data were used for inflow-salinity regressions, inflow-harvest equations, sediment and nutrient loading estimates, and the analysis of inflow-productivity data for some of the contracted studies. Section 4.1 presents an overview of the coastal hydrology information for the Nueces, Mission-Aransas, Guadalupe, Lavaca-Colorado, Trinity-San Jacinto, and Sabine-Neches estuaries. The section shows the general patterns of inflow and its components, and trends of inflow from 1941 through 1987.

Identification of the Effects of Freshwater Inflow on Ecosystem Components

Since earlier freshwater inflow studies on Texas estuaries were completed (TDWR 1980a, 1980b, 1981c, 1981d, 1981e, and 1983), significant advances have been made in measurement techniques and concepts of bay and estuary operation. At the same time, estuarine ecosystem components have been unevenly studied, with some processes better understood than others. To broaden and balance our knowledge of ecosystem operation, a series of study topics were prepared to provide a better picture of the system components and natural processes occurring in bays and estuaries. Some of these studies were designed to provide specific information about the relationship of nutrients, sediment, and salinity to inflow, as required in the legislation; others were undertaken to expand information about the target species. A common need was to definitively determine whether a direct relationship existed between freshwater inflow and an ecosystem component or natural process.

Selecting the number and complexity of the components for study. Brandes (1976) presented a computer simulation model of Texas bays and estuaries that was originally intended to be used in estimating freshwater inflow requirements. The model included virtually every component and process that was known to occur in estuarine ecosystems at the time. It contained mathematical representations of populations of phytoplankton, zooplankton, benthic organisms, several groups of consumer fish and shellfish, and microbial decomposers.

Some of the processes modeled included nutrient uptake and regeneration, photosynthesis, growth and migration of species, organism grazing and mortality, gas exchange between the air and water column, and detrital production and decomposition. The ecological model was integrated into a simulation model of bay circulation; consequently, biological, chemical, and hydrodynamic processes were combined into a single mathematical representation of bay and estuary dynamics. Although the model was well-conceived and carried out, the lack of detailed information about specific components and natural processes in Texas bays and the complexity of the model limited its use.

For this study, simple and direct relationships between freshwater inflow and ecosystem components or processes were emphasized. To the extent possible, the relationships that were quantified were based on direct measurements from Texas bay systems rather than literature for other estuaries. Considerable importance was given to demonstrating the presence or absence of a relationship between inflow and the various ecosystem components. With the resources and time available for this study, it was not possible to quantify the components and processes to the extent needed to prepare a refined simulation model such as Brandes' (1976) original attempt.

The components and processes included in the studies largely encompassed those described in Brandes' (1976) simulation model, although there were a few differences. For example, the pathways of nutrient use and regeneration are now better defined; nitrification, denitrification, and water column regeneration that were not specifically included in the Brandes model were included for study by this effort. A general description of the components and processes examined is given in the paragraphs below.

Inflow, salinity, and nutrient and sediment loading. Inflow was determined on a daily basis in the hydrology studies. Describing the relationship between inflow and salinity, nutrient loading, or sediment loading required quantification with field data that was collected over a long time frame. Equations describing the inflow-salinity relationship for the Guadalupe Estuary are given in Section 8.2. Inflow-salinity equations for other bay systems will be given in later reports for each estuary. Section 4.1 provides an overview of salinity characteristics and trends for the estuaries over the 1941 through 1987 period. Further detail for the Guadalupe Estuary, presented as an example analysis, is contained in Section 7.1.

The relationship between inflow and nutrient loading for most of the coast, is discussed in Section 4.2 and is discussed in more detail for the Guadalupe Estuary in sections 7.2 and 7.3. Inflow-nutrient relationships for other bay systems will be discussed in later reports for each estuary. Nutrient distribution and abundance is a topic of importance since a spatial view of nutrient concentration change provides evidence of where nutrient uptake and regeneration are taking place, and of differences that occur throughout the estuary. Temporal differences in daily, seasonal, and annual nutrient abundance at selected sites provides information about the uptake and release of nutrients and seasonal or annual differences that may occur. Nutrient distribution and abundance for nitrogen, phosphorus, silicate, and carbon is presented in Section 4.2 for the whole coast and in Section 7.2 for the Guadalupe Estuary, with emphasis on nitrogen.

The inflow-suspended sediment loading relationship is discussed in Section 4.4 for most of the major river systems flowing to the coast; specific information about the Guadalupe estuary is given in Section 7.5. A major role of sediment loading to the estuaries is to maintain habitat structure in deltas and shallow bay bottoms. The relationship of sediment loading to the creation and maintenance of these habitat features has not been widely studied along the Texas coast. A literature survey was conducted to gather all the site-specific information available concerning this topic. Field measurements were taken at the deltas on the Colorado and Trinity rivers to assess the rate of wetland aggradation (vertical growth) and erosion from freshwater inflow and bay water movement. Results of these studies are also reported in Section 4.4.

Primary production. Primary production forms the base of the food chain in estuaries and provides energy for all estuarine animals. Texas bays have several types of primary producers: phytoplankton, benthic algae, microphytobenthos, marsh grasses, and seagrasses. Phytoplankton provide the largest share of the primary production in most estuaries. Studies were made of the distribution, abundance, and production of phytoplankton. Important aspects of the studies were the relationship of abundance, production, and nutrient uptake to freshwater inflows. Section 5.1 reviews the results of these studies and other information about phytoplankton from several Texas bays.

Marsh plants in bayhead deltas, fringing marshes along the bay edges, and on the backside of barrier islands produce a significant quantity of organic material and provide important habitat to species targeted for inflow management. A study of the factors affecting the distribution of marsh plants was undertaken for the Guadalupe Estuary; emphasis was placed on those factors that are affected by freshwater inflow. Observations from this study are applicable to other estuaries and are discussed in Section 5.2.

Seagrasses, vascular plants rooted in the bay bottom that live their entire life cycle submerged, are important primary producers. Moving from north to south on the Texas coast, the area of marsh plants declines while the area of seagrasses increases. Seagrasses provide organic matter to the estuarine community, a hard substrate for epiphytic algal growth, and habitat for species of importance. Seagrass communities were compared in the Guadalupe and Nueces estuaries; and some differences were noted between years of high and low inflows. Observations from these studies about distribution and production, and the relationship to freshwater inflow, are presented in Section 5.2.

Freshwater inflow transports organic materials produced on land to the bays via runoff. Materials carried to the bays include leaves, twigs, and decaying organic matter. Freshwater inflow may also transport organic material from delta marshes during periods of flooding. The importance of these materials compared to the organic matter produced by phytoplankton has been a major question in determining how estuaries function. Studies were conducted to trace the sources of organic material that is used by organisms and is present in the sediment to understand the role freshwater inflow may have in supplying organic matter to the estuary. Results of these studies are presented in Section 5.6.

Primary consumers. Zooplankton and benthic organisms are the primary users of phytoplankton produced in the estuary and organic material transported by river flow. Both groups are prey for organisms higher in the food chain and qualify as species on which fish and shellfish are dependent. Zooplankton have relatively short generation times and may respond through rapid growth to changes in phytoplankton populations associated with inflow. They may also be swept away with large inflow volumes. Section 5.3 examines studies done in the Lavaca-Colorado, Guadalupe, and Nueces estuaries and discusses the effect of freshwater inflows on the distribution, abundance, and productivity of zooplankton.

Most benthic organisms are relatively immobile and are not physically displaced by large freshwater inflows. They may, however, respond to increased levels of phytoplankton production or organic input resulting from freshwater inflow by increased productivity. They may also respond to drastic changes in their habitat, such as long periods of very high or low salinity, by decreased productivity or death. Data on macrobenthos have been collected in most bays over ten to 20 years. Section 5.4 discusses patterns of distribution from the long-term benthic data collections and from studies carried out in the Lavaca-Colorado, Guadalupe, Mission-Aransas, and Nueces estuaries. The section also reports on differences in abundance of small benthic organisms (meiobenthos) related to inflow, and differences in their consumption of bacteria and algae from the bottom sediments.

Nutrient processes. Sediment and salinity are conservative constituents of estuarine ecosystems; they are neither used up nor transformed, and the movement of sediment or saline waters can be traced in a relatively straightforward manner. Tracing the fate of nutrients (carbon, nitrogen, and phosphorus) is much more complicated since they are actively taken up by organisms, processed many times before they are lost from the system, and may be imported or exported through many pathways that are not well documented. To be able to use nutrients in the analytical procedure, several contract studies were planned to investigate their uptake, regeneration, and loss in bay systems.

The processes of nutrient uptake, benthic and water column regeneration, nitrification, and denitrification, were studied in the Guadalupe and Nueces estuaries. Some discussion of regeneration is given in Section 5.4. A comparative view of nutrient cycling in the Guadalupe and Nueces estuaries is given in Section 5.5; this section demonstrates some of the differences that may occur in nutrient cycling during wet and dry years.

One way to get an overview of nutrient processes in estuaries is to prepare complete nutrient budgets that account for all the inputs to and losses from the system. Studies in the Guadalupe and Nueces estuaries provided the detail about individual processes such as regeneration and denitrification, but a complete nutrient budget requires the addition of inflow information and bay hydrodynamics to account for all the gains and losses. The models of bay circulation can be used to provide this detail. Section 7.3 is a complete nutrient budget for the Guadalupe estuary. This budget was prepared under conditions of high and low inflows so differences in the overall pattern of nutrient processing and movement with respect to inflow could be understood. Nutrient budgets constructed in a similar manner for other bay systems will be discussed in later reports for each estuary.

Studies on larvae, and juvenile fish and shellfish. Freshwater inflow may have an effect on larval and juvenile fish and shellfish: it may influence the transport of animals into the estuary from the Gulf or spawning areas in passes; and, it may control the use of certain habitats favored by species for their growth and development. Eggs and larvae are carried into the estuary from the Gulf of Mexico or from spawning areas near passes. A few studies on other coasts have suggested that freshwater inflow influences egg and larval transport. A study was pursued to evaluate the importance of these factors and determine whether any were related to freshwater inflow. Results of that study are discussed in Section 6.1.

As the organisms approach the juvenile stage, their salinity requirements for osmoregulation become less important and their habitat requirements for cover and food begin to dominate their distribution. Section 6.2 presents a review of salinity and habitat preferences for juveniles of six common species and recent distribution information for the Guadalupe Estuary from a large-scale sampling program used to provide comparative abundance measurements in all bays.

Marsh, seagrass, open-bay bottom, and reef habitats are not distributed evenly throughout estuaries. In most bays, large brackish-to-freshwater marsh areas are associated with river deltas. Salt marshes occur on the backside of barrier islands, by the edges of bays in areas draining the uplands, and along the bay front of the mainland that parallels the barrier islands. Seagrasses are restricted by depth, salinity, and wave energy, and are found in narrow fringes around the bay edges and in locations that are sheltered from strong wave action. Reef location depends on hard substrates and orientation to currents. These habitats are known to be used by juvenile organisms, but the relative importance of one area compared to another as nursery habitat has not been rigorously tested, especially for different wetland areas. Studies were conducted in the Trinity-San Jacinto, Lavaca-Colorado, and Guadalupe estuaries comparing abundance of juveniles among habitats, at different locations within a habitat, and before and after major freshwater inflow events. These studies allow some general patterns to be identified about the use of habitats by juveniles; the patterns and their relationship to freshwater inflow are discussed in Section 6.3.

Studies on metabolism, reproduction, and larval survival in fish. Adult organisms have well-developed osmoregulatory abilities. The larger fish and shellfish species that are of interest because of their sports or commercial value can move about at will, taking advantage of opportunities for food or cover. Their physiology allows them to inhabit environments that are suboptimal by paying a metabolic price: they commit energy to osmoregulation so they may temporarily take advantage of other habitat benefits. The effects of varying salinities on adult fish metabolism is discussed in Section 6.4.

Another effect of salinity on adult fish and shellfish involves reproduction. The metabolic cost of inhabiting suboptimal environments can be measured by differences in egg production and spawning frequency. Acclimation conditions experienced by females may influence the size and condition of the eggs, fertilization success, and other aspects of reproduction. Fresh water also alters the osmotic environment experienced by eggs and larvae. Their ability to adapt to different salinities strongly affects their survival. While adaptability is known to vary according to different salinity regimes, few studies have systematically looked at salinity tolerances and quantified day-to-day differences in survival after hatching. Studies were conducted to evaluate the effects of salinity on adult reproduction, egg hatching, and larval development of three fish species with different life histories. A review of the results of these studies is presented in Section 6.5.

Studies on adult fish and shellfish abundance, distribution, and harvest. Metabolism, habitat preferences, feeding opportunities, and reproductive needs generally govern the distribution and abundance of adult fish and shellfish. Section 6.6 presents a review of environmental requirements for nine major bay species and an evaluation of the effects that freshwater inflows have on their distribution and life history. This section also presents monthly relative abundance information for the species in two salinity zones in San Antonio Bay.

Freshwater inflow legislation provides a management objective of maintaining productivity of sport or commercial fish and shellfish and the estuarine life on which they depend. Productivity, as mentioned in the legislation, means the abundance of organisms or the yield from harvest, rather than its stricter interpretation, the rate of storage of energy or mass, which is difficult to measure. While abundance or yield are simple in concept, they also present measurement problems. The best methods of measuring abundance require attention to the location, timing, and selectivity of the collecting gear as well as adequate sample sizes to ensure that variation can be treated statistically and that samples are unbiased. This type of sampling program has been conducted in Texas bays for a few years. Unfortunately, there have not yet been enough years of data collected within any one bay system to allow results of this fishery-independent sampling program to be the basis of the quantitative relationships between inflow and abundance needed by this study. However, these data can be used to demonstrate correlations between densities in areas of the bay and salinities. They can also be used to evaluate likely conditions for growth and reproduction. Section 6.7 uses this abundance information to assess differences among the bays from the lower Laguna Madre through the Trinity-San Jacinto Estuary.

Annual harvest information has been collected since the 1940's. Data earlier than 1962 suffer from a variety of inconsistencies and cannot be used. Consequently, about 25 years of harvest data are available for analysis. The use of harvest data has a number of drawbacks: there are problems of accuracy and unreported harvest; fishing effort must sometimes be considered in the analyses; new information is not available for species that are no longer harvested; and, data are available only for commercially harvested species. Even with these caveats, harvest information provides a significant data set over a wide range of inflow conditions for determining a quantitative relationship. Section 6.8 presents an analysis of the effects of inflow on commercial harvest, based on data for seven species, using multiple regression to relate the variables.

While harvest statistics provide direct information about major commercial fishery species, there is no comparable quantitative data available for those species characterized as estuarine life on which sport and commercial fish and shellfish are dependent. In practice, the inflow-harvest relationship embodies more than just the direct effects of fresh water on commercial fish or shellfish; it includes indirect effects of inflow on the habitats and organisms on which they feed. Thus, it is an integrative measure of the effect of freshwater on the fishery rather than just on the individual organism's physiology. The fishery-independent sampling program may provide more direct information for other estuarine life since small fish and shellfish that may be prey to the carnivorous species are collected.

Development of the Analytical Method

Optimization methods. The basic problem in establishing freshwater inflow needs is to determine the minimum quantity of freshwater and the timing of inflows necessary to provide the proper conditions in an estuary to maintain productivity. Minimum inflows are of interest because the use of freshwater by agriculture, municipalities, and industries can potentially reduce the quantity that reaches the estuary. Management science, a branch of applied mathematics, has developed analytical methods to solve problems such as this. The analytical methods usually involve calculating the best solution to a complicated equation—the objective function—involving a number of independent variables and a dependent variable such as inflow volume. The objective function may not provide a cause-and-effect explanation of the relationship between the dependent and independent variables, but it does represent the measure of effectiveness between the dependent variable and the independent variables.

There may be other information available about the variables that has a bearing on the problem. For example, values of some of the independent variables may be allowed to fluctuate only within certain ranges. These restrictions or limits are called constraints, and they confine the range of values of the objective function's dependent variable.

A collection of computational techniques called mathematical programming was developed to solve these types of problems. While there are a variety of techniques available, each designed to solve specific kinds of problems, they all share the same basic purpose: to provide an optimum solution of the objective function that satisfies all the constraints and gives the best possible value (minimum or maximum) for the dependent variable.

Specific requirements for the freshwater inflow problem. Determining the freshwater inflow needs of estuaries is an optimization problem that can use the techniques of mathematical programming to calculate solutions. Objective functions can use inflow or harvest (or another measure of productivity) as dependent variables. The constraints can include equations relating salinity to inflow, salinity viability limits of organisms, historical inflow patterns and limits, and minimum loadings of nutrients and sediment. The problem is complicated because the physiological limits of organisms for salinity change as they develop through their life stages, and life-stage development is related to the season of the year. The equations relating harvest or productivity of particular species have terms in them that depend on inflows in particular seasons of the year. The species differ with regard to which seasons are important, the magnitude of the importance, and whether the effect of seasonal inflow is positive or negative. Consequently, species within a single bay system may have conflicting inflow requirements.

Several advances in mathematical programming techniques have been made in the past decade, the most important of which are: the use of nonlinear relationships; the addition of stochastic elements to the analysis; and the ability to consider multiple objective functions. Linear programming, a form of mathematical programming, requires that the relationship between the dependent and independent variables must be linear. Linear programming was used in some of the previous studies on freshwater inflow needs for Texas bays (TDWR 1980a, 1980b, 1981c, 1981d, 1981e). The last study of that series, involving the inflow needs for the Laguna Madre (TDWR 1983), used a nonlinear optimization procedure that had not been available for the earlier studies. Since many of the relationships needed for this analytical method are nonlinear and cannot be transformed into linear relations, the general ability to use nonlinear relationships is a significant improvement.

In the previous inflow studies, the optimization methods used regression equations involving inflow and harvest or inflow and salinity as deterministic constraints with no consideration of the underlying variance or uncertainty of the regression expressions. Methods have been developed to incorporate stochastic aspects of regression expressions into mathematical programming models. Thus, the variance of the inflow-salinity and inflow-abundance regression relationships can be included in the analysis. This allows natural resource managers to specify that the analysis must provide solutions that satisfy salinity bounds or abundance targets at a given probability level. The addition of statistically based constraints will allow managers to estimate the effects that increasing the certainty of satisfying the salinity bounds will have on abundance of the target species.

Recent improvements in mathematical programming techniques allow multiple objective models to be used. As an example, multiple objectives could be used to minimize the annual inflow while at the same time maximizing the annual harvest. This can become complicated because in some portions of the range of solutions to the problem, the objectives can conflict.

A mathematical programming model incorporating all these advances did not exist as an off-the-shelf computer program. Consequently, one project in this series involved the development of the analytical software, combining all of these concepts and refinements. In addition to adding these mathematical programming model advances, the software was designed to allow the user to specify that a specific salinity gradient should exist throughout the estuary. This allowed the model to account for salinity conditions that were different in distinct regions of the estuary, adding an additional realistic aspect to the analytical procedure. A description of the method of formulating objective functions, constraints, and stochastic considerations is given in sections 8.1 and 8.2.

Example Analysis

Site selection. With information from the special studies and the mathematical programming model, it was possible to formulate and conduct an example analysis of inflow needs of an estuary. The Guadalupe Estuary was selected for the test case. Because research studies were recently conducted in the estuary, a reasonably complete set of data was available for use in the analysis including harvest equations, nutrient and sediment loading information, and hydrodynamic models and salinity equations. The Guadalupe Estuary is not urbanized and water quality does not appear to be a complicating factor. The basin is not strongly affected by upstream diversions or impoundments, but may be the site of future reservoir development as several potential sites have been identified in the lower half of the basin. The economy of the area around the estuary is affected by sport and commercial fisheries, and in 1987 there was a period of extreme high inflows that nearly eliminated the oyster harvest for the next two years (Quast et al. 1989). As a test case, the Guadalupe Estuary occasionally experiences inflow that is sometimes too great or sometimes too small to maintain productivity of some species. Aransas Bay to the south, which receives Guadalupe River water via the Guadalupe Estuary, had a significant increase in oyster landings in 1987 and 1988.

Other relationships or constraints. Use of the mathematical programming model required specification of conditions for analysis, selection of relationships to use in the model, and determination of constraints for salinity, inflow, harvest, nutrients, and sediment. Salinity, inflow, and harvest constraints were jointly selected by the staff of the TPWD and TWDB to test the operation of the model. The final constraints for these variables are discussed in sections 8.1 and 8.2.

The nutrient and sediment constraints required significantly more analysis. It was first necessary to develop a concept of how to use nutrient information to set a constraint for the programming model. Sections 7.3 and 7.4 provide the logic and computations for the establishment of the nutrient constraint. The form of the equation used by the mathematical programming model is given in Section 8.2. In the same manner, a concept was required for the use of sediment as a constraint. The rationale and calculations for the sediment constraint are presented in Section 7.5, and the equation and limit are shown in Section 8.2.

Definition of ecologically sound environment that satisfies the purpose of the law. Criteria that could be used to compare the output from the model were needed to judge whether the salinity, nutrient, and sediment loading regimes were adequate. Section 7.6 presents the salinity criteria and the nutrient and sediment loading requirements for providing an ecologically sound environment.

Specifying objectives for model computations and presenting results. Because mathematical programming models are optimization techniques, objectives for management must be specified by the decision maker to be able to select an answer for use. In Section 8.2, the results of this analysis are presented for different probability levels. The results presented span the inflow scale from minimum to maximum inflows which still satisfy the constraints. Maximum harvest values and maximum probabilities for achieving salinity bounds are also shown. Other policy issues including the selection of species used for the analysis and the relative weighting of species by value or importance are discussed in this section. The results of the analysis show the range of answers that are feasible. Existing policies or new policy decisions must be applied to reduce the range of solutions so a decision can be made.

Hydrodynamic and Salinity Model to Check the Results

Once the policy decisions are decided, a final set of inflows can be computed that satisfy the objectives and constraints. The inflow volumes are then used to check the salinity pattern in the estuary to confirm that the salinity distribution requirements are being met.

Run model over a year cycle to display salinity distributions. Data for a typical year must be assembled for the circulation and salinity model runs. This includes wind direction and velocity, tide elevation, and hydrology information based on the computed inflow values from the mathematical programming model. The hydrodynamic model is run to simulate an entire year; this requires several days of continuous computer simulation. Salinity information is selected from the model run at monthly intervals to produce a salinity contour map of the estuary. Section 8.4 discusses the hydrodynamic model and shows the series of salinity contours that result from the computed monthly inflows.

Compare patterns with ecologically sound environment requirements. The last step in the process is to confirm that the simulated salinity patterns are consistent with the requirements for an ecologically sound environment defined earlier in the analysis. This is a judgement step requiring experienced persons to observe the isohalines from monthly plots of salinity and compare those results with the salinity requirements defined in the ecologically sound environment criteria. If the salinity contours are judged to satisfy the requirements, a feasible solution has been achieved and can be used by the decision maker. If the salinity distribution pattern does not satisfy the ecologically sound environment criterion, the process must return to the mathematical programming model for reassessment.

There are many ways that this model can be modified to achieve the desired goal; salinity bounds in the different regions can be changed for particular months to force the model to provide more or less inflow. Changing probability levels, species, weighting, and inflow or harvest bounds are also possible changes that can be made. The decision of what to change will depend on the situation and the objectives for management. Changes such as these will require interaction between the staff making the analysis and the decision makers to be sure that the changes are compatible with the policy directions.

3.2 SUMMARY

The analytical approach described here starts with a set of requirements aimed at providing quantitative estimates of inflow. A series of studies were defined and carried out under contract by state and federal agencies or universities, and by the staffs of the TPWD and the TWDB The studies were designed to broaden the understanding of estuarine ecosystem functions and provide specific data and methods for use in an analytical procedure for determining inflow needs. Topics of the studies included: hydrology; hydrodynamics; coastal geology; estuarine water chemistry; intensive water quality monitoring; nutrient dynamics and processing; and distribution, abundance, and productivity of estuarine organisms, from primary producers through secondary consumers. In addition to the information used by the analytical procedure, the studies provided for specific tests of the existence of relationships between inflow and ecosystem components and natural processes. Results of these studies are summarized in later chapters.

With the study information, the mathematical programming model, and the hydrodynamic models, the analytical procedure was applied to the estuary of interest. First, the mathematical programming model was used to calculate the required inflow volumes. This required specification of a variety of management objectives and constraints that must be determined by decision makers with authority for estuarine management; some analyses are required to justify and set constraints relating to nutrients and sediment. The mathematical programming model takes into account the desired salinity gradient, historical inflow patterns, the relationship between harvest or abundance and inflow, statistical uncertainties of the harvest equations and the inflow-salinity equations, harvest or abundance targets, and the estuary's requirements for potentially limiting nutrients and for sediment.

When an inflow pattern was computed that satisfied the constraints and achieved the management objectives, the inflow was tested with a detailed hydrodynamic model to be sure that the month-to-month salinity distribution was consistent with the requirements needed to define an ecologically sound environment. If the salinity distribution was inconsistent with the ecologically sound environment criteria, modifications were made to the mathematical programming model constraints or objectives to refine the analysis. Once the salinity distribution was consistent with the ecologically sound environment criteria, the results could be used in making water allocation decisions.

CHAPTER 4: COASTAL HYDROLOGY AND THE RELATIONSHIPS AMONG INFLOW, SALINITY, NUTRIENTS, AND SEDIMENTS

4.0 INTRODUCTION

Freshwater inflow strongly affects bay salinity, nutrient loading, and sediment loading in Texas estuaries, although the climatic gradient down the length of the coast is the most important natural factor mediating the supply of fresh water for bays and estuaries. Human activities can determine both the quality and quantity of the inflows, so there is concern that this influence could be deleterious. However, sound resource management practices can balance human influences and the needs of the estuaries. Selecting levels of salinity, nutrient, and sediment loading that are adequate to maintain an ecologically sound environment requires perspective about these materials for each estuary. The purpose of this chapter is to provide background information about freshwater inflows, bay salinity levels, nutrient loading and distribution, and sediment loading to Texas estuaries. In general, historical patterns of distribution and variation will provide the appropriate information for resource management consideration.

Hydrology. Section 4.1 provides a detailed analysis of coastal hydrology. The section includes a comparison of the components of freshwater inflow among six estuaries over a 47-year period. Salinity characteristics of the bays are also compared graphically and statistically. This section provides an analysis of trends of freshwater inflow over 47 years, and trends of salinity covering a period of about 20 years. The period of record includes several droughts and years of high inflow.

Distribution of nutrients. Section 4.2 shows historical information about the spatial distribution of nutrients in five Texas estuaries. The data for this analysis come from two coastal monitoring programs and from special studies of several estuaries. This section presents maps of nutrient distributions for high and low inflow regimes. The distribution patterns suggest an interpretation of the relative importance of turbidity, heterotrophic production, regeneration, uptake and recycling, and transport of materials from the head to the mouth of the estuary.

Nutrient loading. Section 4.3 compares the loading of nutrients into five Texas estuaries from several different viewpoints, relative to the estuaries' capacities and volume turnover. The section examines the relative importance of the major sources of nutrients to the estuaries and shows the degree of control that manipulation of gaged flows has on nutrient loading.

Sediment loading. Section 4.4 examines the third basic element for maintaining an ecologically sound environment, sediment. The section describes the characteristics of sediment transported by the major river systems that flow into Texas estuaries. In addition, it contains an analysis of sediment loading, showing changes through time. This section includes a description of how deltas develop at river mouths and an assessment of the status of sediment input to deltas and bays, based on recent sampling and analysis.

4.1 PATTERNS OF INFLOW AND SALINITY

Introduction

Freshwater inflows and salinity levels found in six Texas estuaries are described in this section. The six estuaries include, from north to south along the Texas coast (Figure 4.1.1), the Sabine-Neches, Trinity-San Jacinto, Lavaca-Colorado, Guadalupe, Mission-Aransas, and Nueces estuaries. Only the southernmost Texas estuary, the Laguna Madre, is not discussed here.

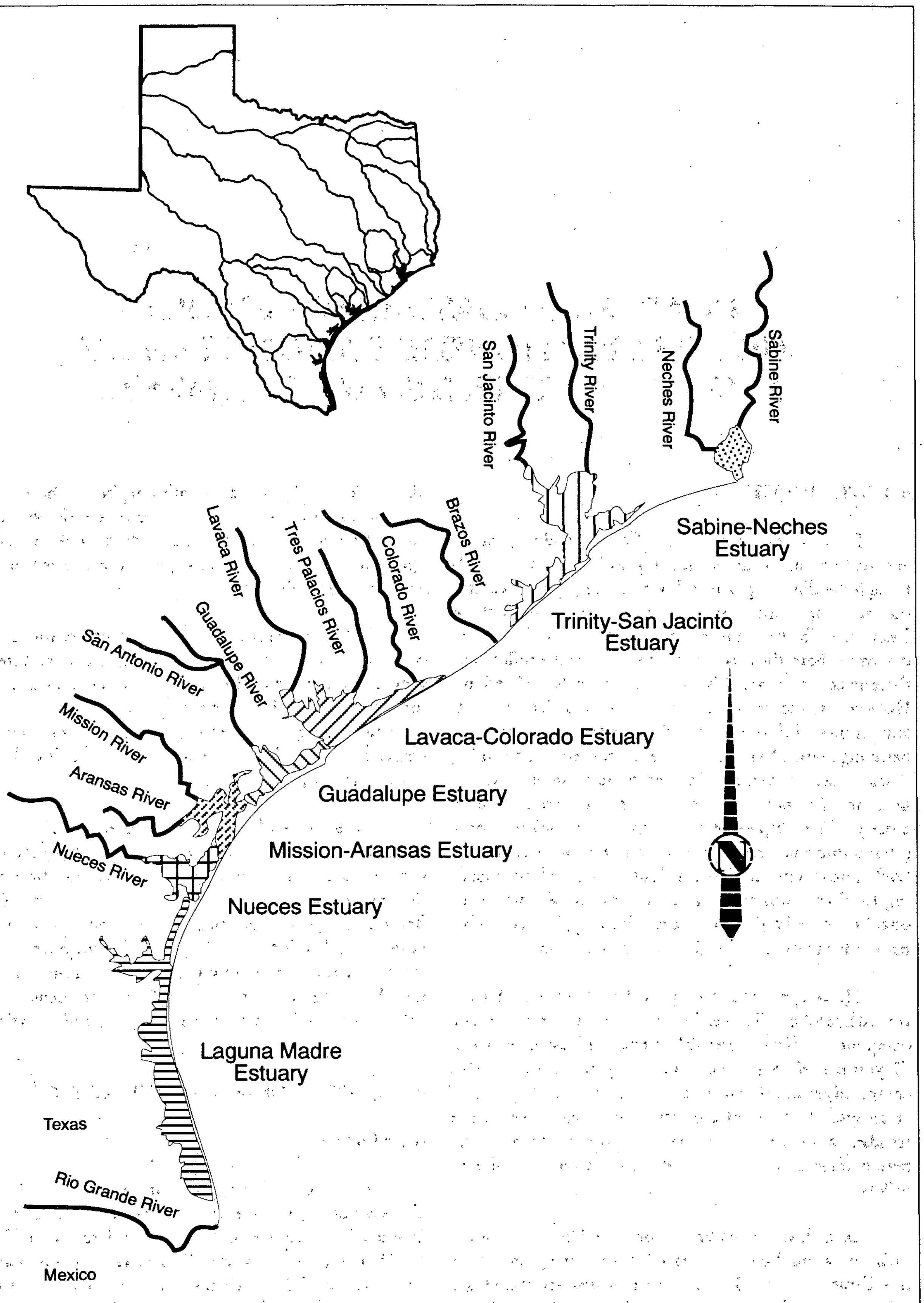


Figure 4.1.1. Location of Texas estuaries.

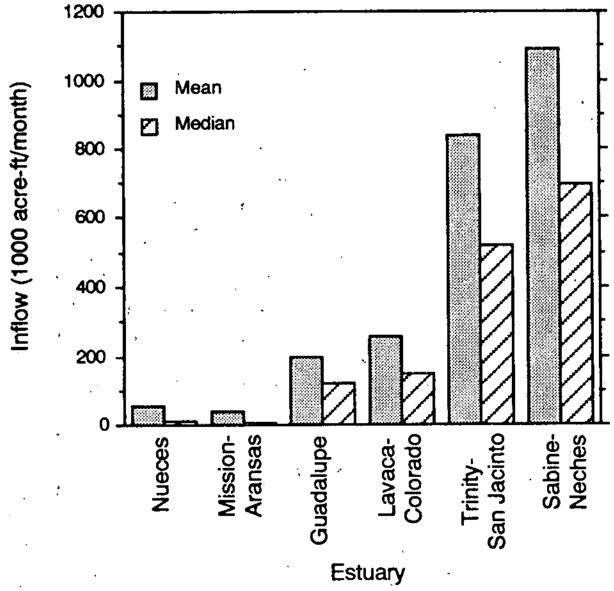


Figure 4.1.2. Mean and median monthly freshwater inflows to the Nueces, Mission-Aransas, Guadalupe, Lavaca-Colorado, Trinity-San Jacinto, and Sabine-Neches estuaries.

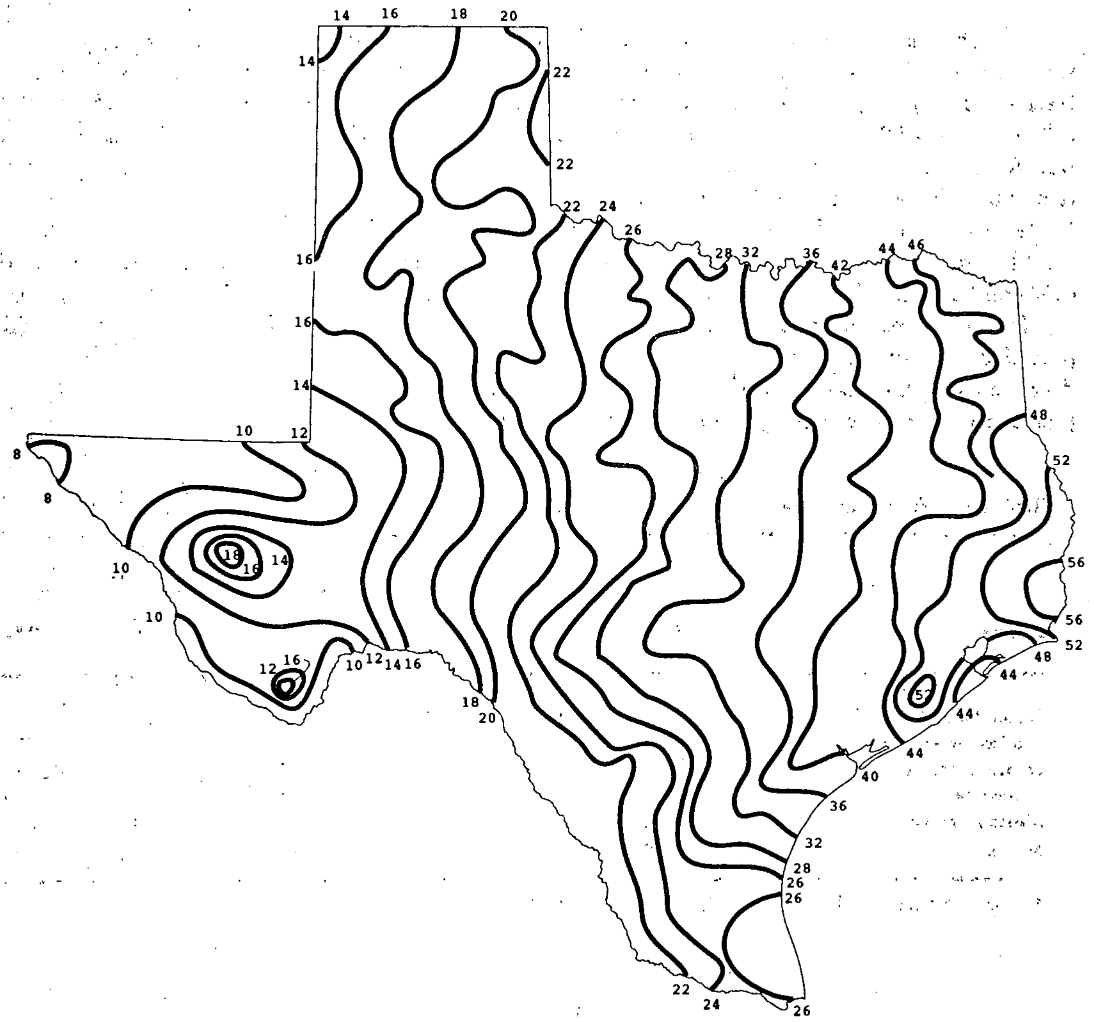


Figure 4.1.3. Mean annual precipitation in Texas—1951-1980.

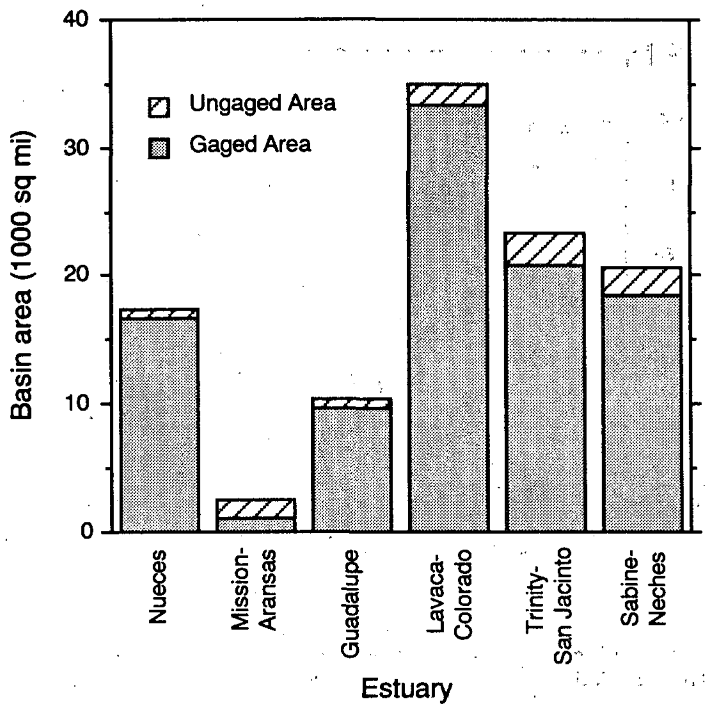


Figure 4.1.4. Gaged and ungaged areas in contributing basins for the Nueces, Mission-Aransas, Guadalupe, Lavaca-Colorado, Trinity-San Jacinto, and Sabine-Neches estuaries.

Freshwater inflow rates to Texas bays and estuaries vary widely. Of the six estuaries studied, the Sabine-Neches Estuary has, on average, the largest freshwater inflow rates at 1.09 million acre-ft/month, while the Mission-Aransas Estuary has the smallest at 0.04 million acre-ft/month (Figure 4.1.2, based on 1941-1987 averages). Median inflow rates for the same period were 0.69 million acre-ft/month for the Sabine-Neches Estuary and only 0.004 million acre-ft/month for the Mission-Aransas Estuary. In general, freshwater inflows to Texas estuaries follow precipitation patterns across the state. Both tend to decrease along the coast from north to south. Annual precipitation decreases from more than 50 inches along the northeast Texas coast to less than 30 inches in West Texas and in the Lower Rio Grand Valley (Figure 4.1.3). Only the Mission-Aransas Estuary differs from the trend. Inflows to the Mission-Aransas Estuary are the smallest of all Texas estuaries primarily because it has the smallest contributing basin (2,480 mi^2, Figure 4.1.4). Monthly inflow statistics including the mean monthly inflow, standard deviation, and monthly inflows for exceedance probabilities (probability of the given monthly inflow being exceeded) of 10%, 25%, 50%, 75%, and 90% are provided in Table 4.1.1 for each estuary.

Table 4.1.1. Freshwater inflow statistics. All flows in thousand acre-ft/month.

			Exceedance probability [a]				
Estuary	Mean	St. D.	90%	75%	50%	25%	10%
Sabine-Neches	1,090	1,070	151	334	694	1,540	2,520
Trinity-San Jacinto	879	887	121	234	530	1,273	2,119
Lavaca-Colorado	257	302	38.5	67.3	146	319	689
Guadalupe	195	225	37.4	70.1	119	241	424
Mission-Aransas	35.8	96.4	0.354	1.38	4.49	23.9	93.6
Nueces	52.8	133	2.36	3.89	8.22	40	143

[a] Probability that stated flow will be equaled or exceeded.

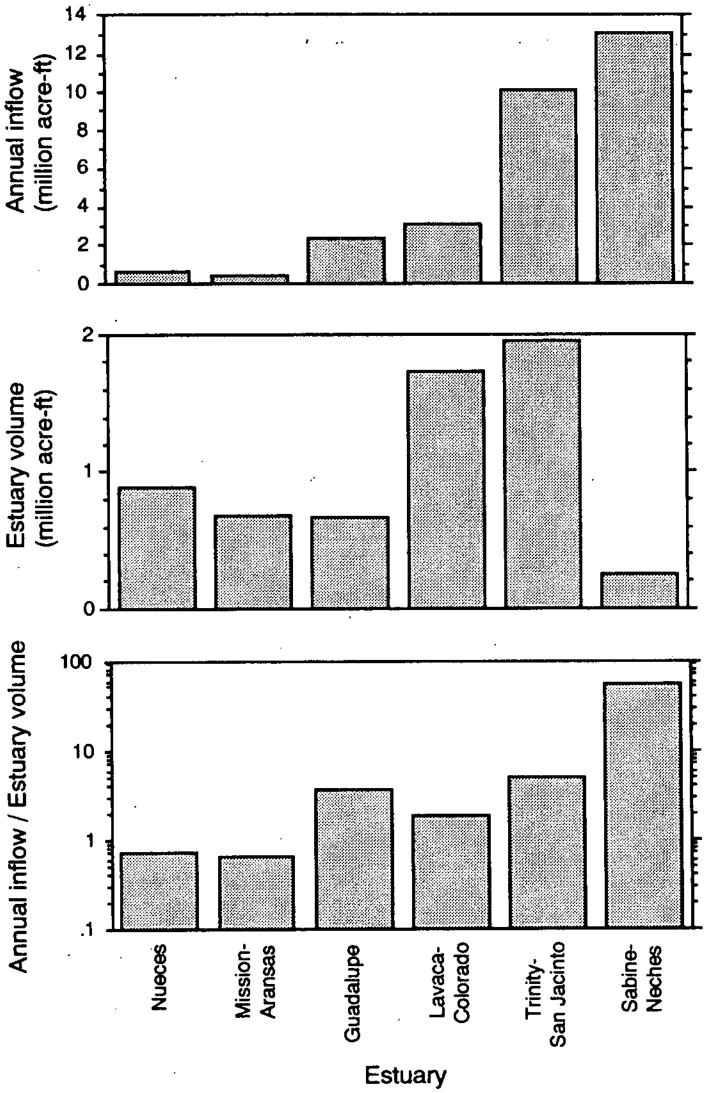


Figure 4.1.5. Average annual inflow, estuary volume, and annual inflow-estuary volume ratio for the Nueces, Mission-Aransas, Guadalupe, Lavaca-Colorado, Trinity-San Jacinto, and Sabine-Neches estuaries.

Mean salinities found in Texas estuaries are also quite diverse, but can be related roughly to the ratio of freshwater inflow to estuary volume. Estuaries with the largest inflow-volume ratios tend to have lower mean salinities than those with smaller inflow-volume ratios. The Sabine-Neches Estuary, with an annual inflow-estuary volume ratio of roughly 53 (Figure 4.1.5), generally has the lowest mean salinity of all Texas estuaries, varying from 5 to 10‰ (Figure 4.1.6). By contrast, the Nueces Estuary has a lower annual inflow-estuary volume ratio of only 0.7, and higher salinity in the range of 21 to 30‰. Mean and median salinities found in Texas estuaries are provided in Table 4.1.2. The actual salinity found in an estuary on any given day depends on the amount of freshwater flowing into the estuary, the rates of evaporation from and precipitation onto the estuaries, quantity of saline Gulf of Mexico water flowing into the estuary due to tidal influence, meteorological effects such as wind speed and direction, and estuary volume.

Freshwater Inflow Patterns to Texas Bays and Estuaries

Sources of hydrology data. Freshwater inflow data presented in this section were compiled by TWDB staff as part of a major study on the effects of freshwater inflows on Texas bays and estuaries. The period covered in this study extends from 1941 through 1987. Hydrological data for the period 1941 to 1976 were prepared in previous bay and estuary studies by TWDB staff (TDWR 1980a, 1980b, 1981a, 1981b, 1981c). The hydrological database, consisting of gaged river flows, modeled flows, diversions, and return flows, was extended in the current study to include the period from January 1977 through December 1987.

Freshwater inflow as used in this study is defined as combined inflows which drain into the estuary. These inflows consist of gaged river inflows, computed runoff from

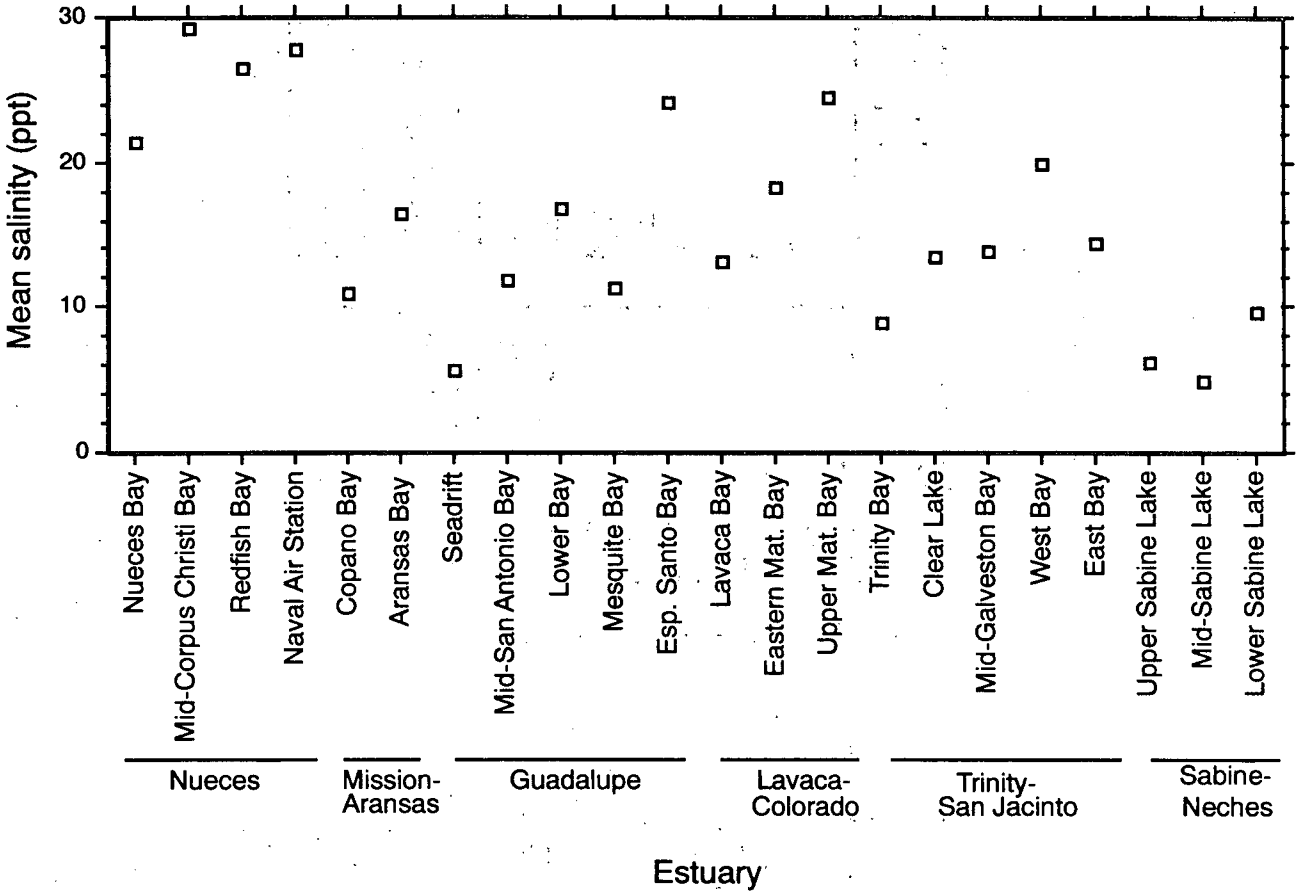


Figure 4.1.6. Mean salinity at several sites in the Nueces, Mission-Aransas, Guadalupe, Lavaca-Colorado, Trinity-San Jacinto, and Sabine-Neches estuaries.

drainage areas lacking river gages at their outlets, flows diverted from ungaged areas (e.g., for municipal, industrial, or agricultural use), and flows returned into ungaged areas as surplus wastewaters from upstream users. Daily gaged streamflow discharge data were obtained from the United States Geological Survey (USGS) surface water data-collection network in Texas. Daily runoff for ungaged areas was computed with a model developed by TWDB staff which is based on Soil Conservation Service (1972) techniques. The model was calibrated with USGS streamflow data and National Weather Service (NWS) precipitation data. Monthly diversion and return flow data were obtained from the TNRCC. Daily gaged streamflows and modeled flows were aggregated into monthly flows and combined with the monthly diversion and return flow data to provide a single monthly freshwater inflow:

inflow = gaged + modeled - diverted + returned.

The final data set consisted of monthly data from January 1941 through December 1987 for each of the six major estuaries studied in this report. The following analyses are based on the compiled data set.

Monthly inflows. Historically in Texas, the largest monthly inflows have occurred in the Sabine-Neches Estuary, where the largest single monthly inflow was 8.09 million acre-ft, while the smallest monthly inflows occur in the Mission-Aransas Estuary, where the largest single monthly inflow was 1.34 million acre-ft (Figure 4.1.7). Inspection of the inflow hydrographs reveals many peaks, some of which are attributable to hurricanes and tropical storms. For example, 2 of the largest 3 and 6 of the largest 20 inflows to each the Mission-Aransas and the Nueces estuaries, respectively, are associated with hurricanes. Eight of the largest 20 monthly inflows to the Lavaca-Colorado Estuary are associated with hurricanes and tropical storms. By contrast, only 1 of the 20 largest monthly inflows to the Sabine-Neches Estuary and the Guadalupe Estuary and only 4 of 20 in the Trinity-San Jacinto Estuary are associated with hurricanes or tropical storms. Many of the other peaks in the inflow hydrographs are associated with frontal passages from the north which trigger showers in the spring and autumn. In an anomalous event, the largest single monthly inflows to both the Guadalupe and Lavaca-Colorado estuaries occurred in June 1987 due to heavy precipitation following an extended wet period during which the soil was saturated.

Table 4.1.2. Salinity summaries for Texas estuaries. Mean, standard deviation, skewness, and median salinities; and number of data points for 22 estuary sub-areas during the period 1968 to 1987.

Estuary/ Location	Mean ‰	St. dev. ‰	Skewness	Median ‰	n
Sabine-Neches					
Upper Sabine Lake	6.30	5.07	0.89	5.60	254
Mid-Sabine Lake	4.95	4.66	0.84	3.65	92
Lower Sabine Lake	9.69	8.18	1.14	8.00	153
Trinity-San Jacinto					
Trinity Bay	8.95	6.31	0.36	8.61	433
Clear Lake	13.50	5.72	1.16	13.25	211
Mid-Galveston Bay	13.82	7.55	1.09	12.72	526
West Bay	19.92	7.79	1.22	19.26	248
East Bay	14.40	7.31	1.40	13.22	220
Lavaca-Colorado					
Lavaca Bay	13.17	7.27	0.11	13.59	563
Eastern Matagorda Bay	18.22	7.38	-0.11	18.76	89
Upper Matagorda Bay	24.51	5.89	0.21	24.40	119
Guadalupe					
Seadrift	5.64	6.48	0.58	3.25	716
Mid-San Antonio Bay	11.94	7.97	0.63	10.40	260
Lower Bay	16.85	8.26	-0.26	18.46	243
Mesquite Bay	11.36	6.41	0.52	9.93	493
Espiritu Santo Bay	24.14	7.26	-0.66	25.50	322
Mission-Aransas					
Copano Bay	10.94	5.19	0.98	10.97	495
Aransas Bay	16.47	8.11	-0.38	17.35	452
Nueces					
Nueces Bay	21.49	9.48	-0.87	23.53	501
Mid-Corpus Christi Bay	29.24	4.91	-1.38	29.99	444
Redfish Bay	26.52	6.36	-0.21	26.44	189
Naval Air Station	27.80	5.57	-0.85	28.60	77

The six estuaries of this study can be grouped into three pairs according to similarities in inflow volumes and in seasonal inflow characteristics. The first pair consists of the two northernmost estuaries, the Sabine-Neches and the Trinity-San Jacinto estuaries. These two estuaries have the greatest mean inflows (greater than 800,000 acre-ft/month, Figure 4.1.2) and have maximum inflows from December through June (Figure 4.1.8). The second pair, consisting of the Lavaca-Colorado and Guadalupe estuaries, have mean inflows of roughly 200,000 acre-ft/month and mean monthly inflow peaks in late spring and early autumn. Finally, the southernmost pair, the Mission-Aransas and Nueces estuaries, have mean inflows of less than 60,000 acre-ft/month, and each have a small inflow peak in the late spring followed by a larger peak in early autumn.

Monthly inflow peaks during May in the Sabine-Neches and Trinity-San Jacinto estuaries (Figure 4.1.8) are due to heavy springtime precipitation which occurs in northeast regions of Texas. By contrast, the two inflow peaks in the Lavaca-Colorado and Guadalupe estuaries indicate the influences of early autumn hurricanes, tropical storms, and coastal storms, and of springtime precipitation. The early autumn peak in the Mission-Aransas and Nueces estuaries is larger than the springtime peak, reflecting the diminished influence of springtime precipitation in the lower coastal bend region, and the significance of late summer-early fall storms in the region.

Inflow frequency distribution. Relative frequency distributions for freshwater inflows (Figure 4.1.9), which indicate the relative probability of finding inflows in a given range, and exceedance probability curves, which indicate the probability that flows of a given magnitude will be exceeded (Figure 4.1.10, Table 4.1.1), can also be grouped into pairs as described above. The modes (frequency distribution peaks) for the Sabine-Neches and Trinity-San Jacinto estuaries, the first pair, lie between $10^{5.5}$ (316,000) and 10^6 (1 million) acre-ft/month. The modes for the Lavaca-Colorado and Guadalupe estuaries, the second pair, lie between $10^{4.5}$ (32,000) and $10^{4.75}$ (56,000) acre-ft/month. Finally, the modes for the Mission-Aransas and Nueces estuaries lie between 10^3 (1000) and $10^{3.75}$ (5,623) acre-ft/month (Figure 4.1.9). Based on the smooth shapes and unimodality of the relative frequency distributions (Figure 4.1.9), inflows to the Sabine-Neches, Trinity-San Jacinto, Lavaca-Colorado, and Guadalupe, estuaries are nearly log-normal distributed. In contrast, the relative frequency distribution for the Nueces Estuary is highly skewed, and that for the Mission-Aransas Estuary is multi-modal, indicating that simple log-normal distributions may not be adequate for approximating inflows to these estuaries.

Cumulative inflows. Cumulative inflow hydrographs, or mass curves, are presented for each estuary in Figure 4.1.11. These curves, constructed by integrating over time the simple inflow hydrographs shown in Figure 4.1.7, represent the cumulative amount of water which has flowed into the estuary, and can reveal changes in inflow rates which might be overlooked in the simple inflow hydrographs. Changes in slope of the cumulative inflow hydrographs indicate either increasing inflow rates (increases in slope), or decreasing inflow rates (decreases in slope). Decreases in slope appear during known drought periods in Texas, i.e., from roughly 1950 to 1958 and from 1962 to 1966. Although several major reservoirs were built during the 1941 to 1987 period (Table 4.1.3), these hydrographs provide no clear evidence that the projects have significantly altered freshwater inflow rates to the estuaries. That is, there are no evident step changes in the slope of the cumulative inflow hydrographs immediately following completion of the reservoirs.

Over the last 20 years of record, 1968 to 1987, freshwater inflow rates for 5 estuaries—the Sabine-Neches, Trinity-San Jacinto, Lavaca-Colorado, Guadalupe, and Mission-Aransas estuaries—appear essentially constant. However, the freshwater inflow rate for the Nueces Estuary has decreased, possibly due to disproportionate increases in water demand by Corpus Christi or irrigation without corresponding increases in runoff due to urbanization. The above assessments are qualitative in nature, based on visual inspection of the slope of the cumulative inflow hydrographs. Quantitative support is provided in the next section with nonparametric statistical analysis techniques.

Trend analysis. Monthly freshwater inflow data used in this study was examined for long-term trends (i.e., long-term monotonic increases or decreases in freshwater inflow rates) with a nonparametric statistical analysis technique

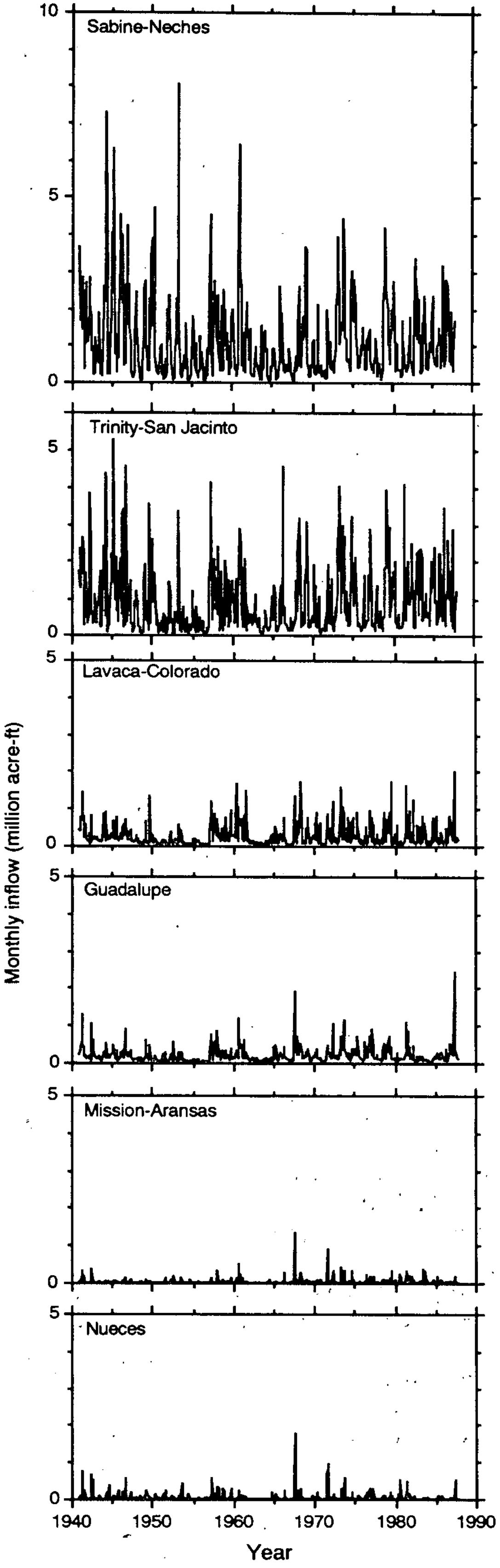


Figure 4.1.7. Monthly freshwater inflow hydrographs from 1941-1987 for the Nueces, Mission-Aransas, Guadalupe, Lavaca-Colorado, Trinity-San Jacinto, and Sabine-Neches estuaries.

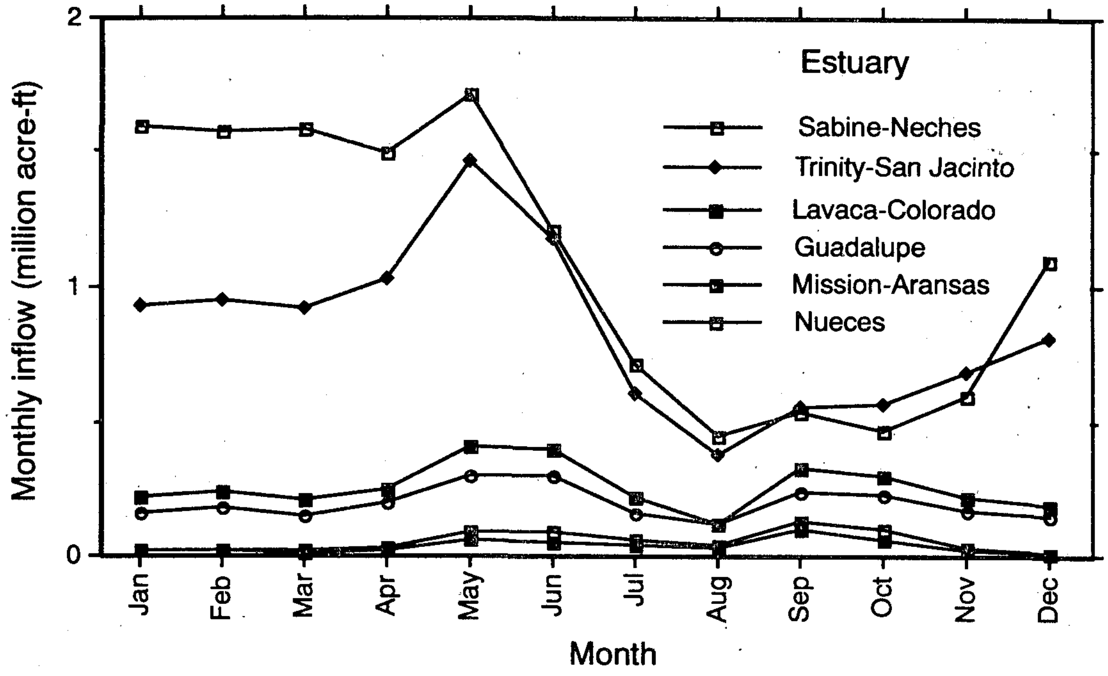


Figure 4.1.8. Monthly average inflows from 1941-1987 for the Nueces, Mission-Aransas, Guadalupe, Lavaca-Colorado, Trinity-San Jacinto, and Sabine-Neches estuaries.

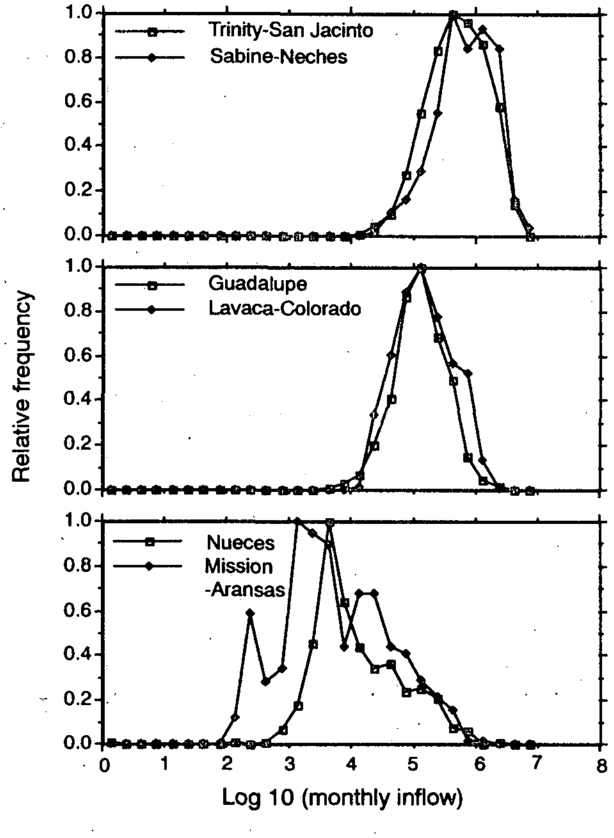


Figure 4.1.9. Relative frequency of monthly freshwater inflows to the Nueces, Mission-Aransas, Guadalupe, Lavaca-Colorado, Trinity-San Jacinto, and Sabine-Neches estuaries, where relative frequency is the ratio of numb er of occurrences of flow within the given range normalized by the maximum number of occurrences found in all ranges.

referred to as the Sen test. The Sen test, proposed by Farrell (1980) and Sen (1968), and described by Van Belle and Hughes (1984), is most appropriate for data sets which have no missing data points, such as the hydrological data set used in this study.

In the Sen test, monthly inflow data is deseasonalized by subtracting from each value the overall mean value for its month. For example, January inflows for each year are deseasonalized by subtracting the mean of all January inflows. Each deseasonalized inflow is next given a rank based on its magnitude. Next, the slope of the best-fit line passing through the set of data points whose coordinates are given by their rank and temporal order is computed. The slope is appropriately normalized so that it has the properties of a Gaussian distribution curve. The normalized slope is the Sen statistic, t. Values of the statistic t greater than 1.96 for an increasing trend or less than -1.96 for a decreasing trend are statistically significant at the 95% confidence level. Although a strong trend may be indicated by the statistic, the magnitude of the trend may not be significant. For this reason, the slope of the deseasonalized data is also computed in those cases where strong trends are indicated by the statistic t. Generally speaking, even if the presence of a trend is indicated, a slope of less than 1 or 2% per year in the deseasonalized data is considered insignificant. The cutoff slope for significance was arbitrarily taken as 1.5% per year.

Have freshwater inflow rates to Texas estuaries increased or decreased significantly from 1941 to 1987? To answer this question, the Sen test was applied to inflow records for three periods: 1941 to 1957, 1958 to 1966, and 1968 to 1987. Droughts are known to have occurred at the end of the periods 1941 to 1957 and 1958 to 1966. Thus, applying the Sen test for these periods serves as a test of the methodology. Decreasing trends would be expected for both of these first two periods. Application of the Sen test to the last period, 1968 to 1987, tests for significant changes in inflow during a time when urbanization spread in many areas throughout Texas.

The results of the trend analysis, Table 4.1.4, confirm that freshwater inflow rates to Texas estuaries decreased ($t < -1.96$) from the 1940's to the 1950's, a decrease attributable to the 1950's drought. Similarly, from the late 1950's to the mid-1960's, inflow rates decreased in association with the dry period of the 1960's. However, during the last 20 years of record, no significant trends are evident in any of the six estuaries studied.

Despite the appearance of decreased freshwater inflow rates to the Nueces Estuary during the period 1968 to 1987 in the cumulative inflow hydrograph (Figure 4.1.12), no trend is indicated by the Sen test at the 95% confidence level. However, the deseasonalized slope for the Nueces Estuary (Table 4.1.4) indicates a large change of -4.33% per year in inflow for 1968 to 1987. Examination of the Nueces cumulative inflow hydrograph indicates large inflows during the 1968 to 1987 period which may have contributed to the "noisiness" of the inflow record, possibly affecting the analysis results. Although the t-statistic is not significant at the 95% confidence level, it is significant at the 80% level.

Table 4.1.3. Some major reservoirs built in Texas from 1941-1987.

Reservoir	Year Completed	Estuary
Toledo Bend	1966	Sabine-Neches
Lake Anahuac	1954	Trinity-San Jacinto
Lake Houston	1954	Trinity-San Jacinto
Lake Livingston	1968	Trinity-San Jacinto
Lake Conroe	1973	Trinity-San Jacinto
Lake Travis	1942	Lavaca-Colorado
Lake Texana	1980	Lavaca-Colorado
Canyon Lake	1964	Guadalupe
Lake Calaveras	1969	Guadalupe
Coleto Creek	1980	Guadalupe
Lake Corpus Christi	1958	Nueces
Choke Canyon	1982	Nueces

It is important to note that the trend analysis was applied only to records of ten years or longer. Shorter periods of record occasionally contain anomalies which visually appear to indicate that trends are present but which cannot be statistically supported.

If the test is applied to the entire 47-year period, the Mission-Aransas Estuary shows a significant trend, a 2.1% per year increase in inflow. None of the other estuaries has a significant trend over the 47-year period.

Historical Salinity Levels for Selected Sites in Texas Bays and Estuaries

Data acquisition. Salinity data presented in this section were gathered from five separate sources. These were the Board's Coastal Data System (TWDB-CDS), the Commission's Statewide Monitoring System (TNRCC-SMS), the Texas Department of Health's Shellfish Sanitation Program (TDH-HSS), the Department's Coastal Fisheries Branch Catch Monitoring Program (TPWD-CMP), and the recently established Board Datasonde Network (TWDB-DN). Data from these sources available for the period 1968 to 1987 were used. In general, measurements made under each of these programs were taken at different

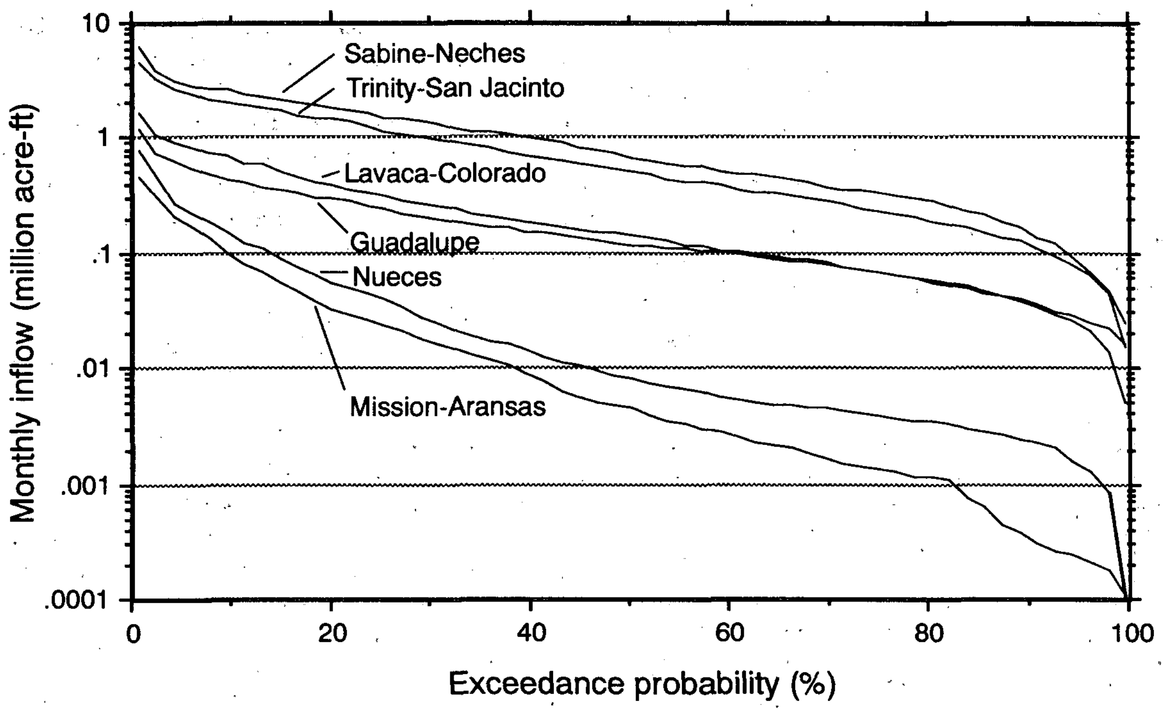


Figure 4.1.10. Monthly freshwater inflow exceedance probability for the Nueces, Mission-Aransas, Guadalupe, Lavaca-Colorado, Trinity-San Jacinto, and Sabine-Neches estuaries.

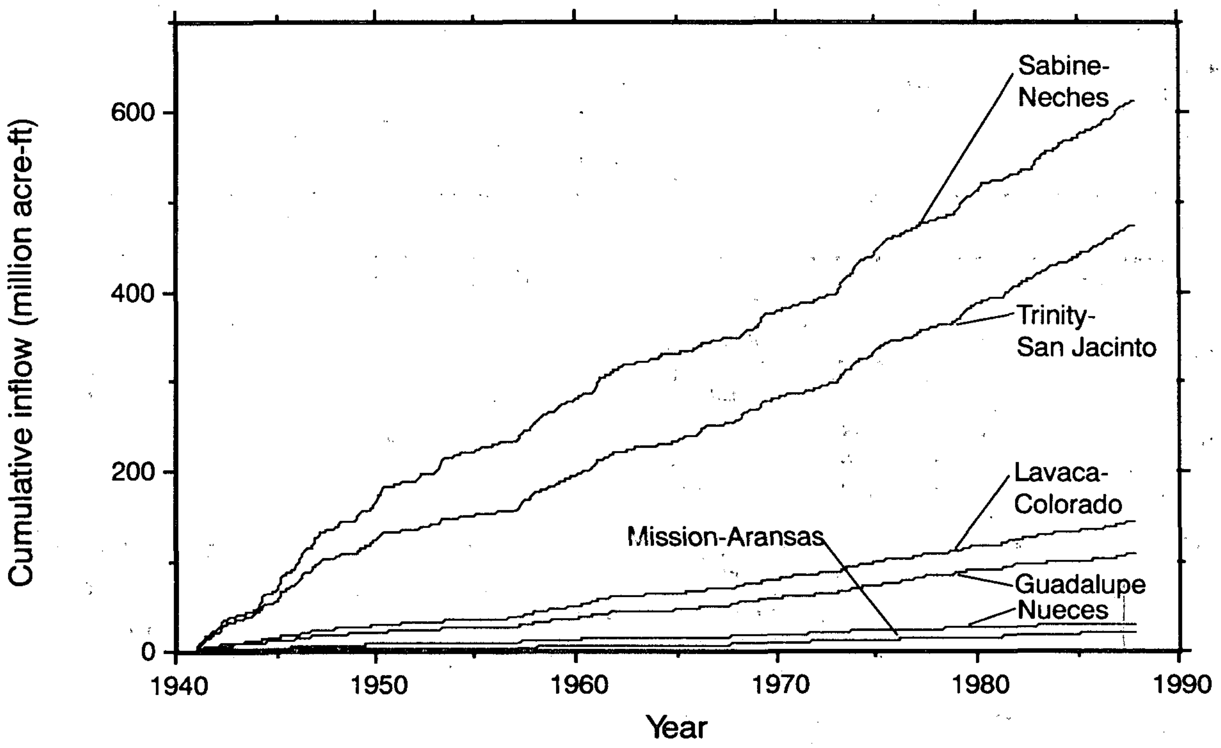


Figure 4.1.11. Cumulative inflow hydrographs (mass curves) for freshwater inflows to the Nueces, Mission-Aransas, Guadalupe, Lavaca-Colorado, Trinity-San Jacinto, and Sabine-Neches estuaries.

Table 4.1.4. Trend analysis of freshwater inflows to Texas estuaries.

Period and estuary	t	% change per year	Significant?
1941-1957			
Sabine-Neches	-4.67	-4.75	Yes
Trinity-San Jacinto	-5.46	-6.71	Yes
Lavaca-Colorado	-5.59	-5.94	Yes
Guadalupe	-5.85	-6.65	Yes
Mission-Aransas	-2.65	-9.77	Yes
Nueces	-2.91	-5.44	Yes
1958-1966			
Sabine-Neches	-3.99	-10.65	Yes
Trinity-San Jacinto	-2.13	-5.70	Yes
Lavaca-Colorado	-3.85	-13.34	Yes
Guadalupe	-3.24	-10.97	Yes
Mission-Aransas	-2.17	-16.64	Yes
Nueces	-2.46	-22.36	Yes
1968-1987			
Sabine-Neches	1.48	0.69	No
Trinity-San Jacinto	1.41	0.52	No
Lavaca-Colorado	-0.98	-0.63	No
Guadalupe	-2.14	-0.74	No
Mission-Aransas	-0.06	-1.24	No
Nueces	-1.29	-4.33	No

times and locations. Data collected under the TWDB-CDS, TNRCC-SMS, and TDH-HSS consisted of instantaneous (grab sample) measurements taken either once per month or once per quarter-year. TPWD-CMP data was collected approximately 10 times per month. TWDB-DN data consisted of hourly to bi-hourly measurements taken from November 1986 through the end of December 1987. Each data set suffered from missing data due to program discontinuity, equipment failure, etc. All data were put into a consistent daily format consisting of one value of salinity for each day in which a measurement was made.

Each estuary was divided into several sub-areas, providing a total of 22 representative sites throughout the six estuaries (Figure 4.1.13). The estuaries were generally divided into upper, mid-, and lower regions in order to establish mean salinity gradients. All measurements taken from any of the four data sources within a given sub-area were combined to form a single larger data set. Measurements taken on the same day within a sub-area were averaged. For each sub-area, the final data set consisted of a single salinity value for each day that any measurements were made in the sub-area. Mean salinities measured at each site are presented in Figure 4.1.6.

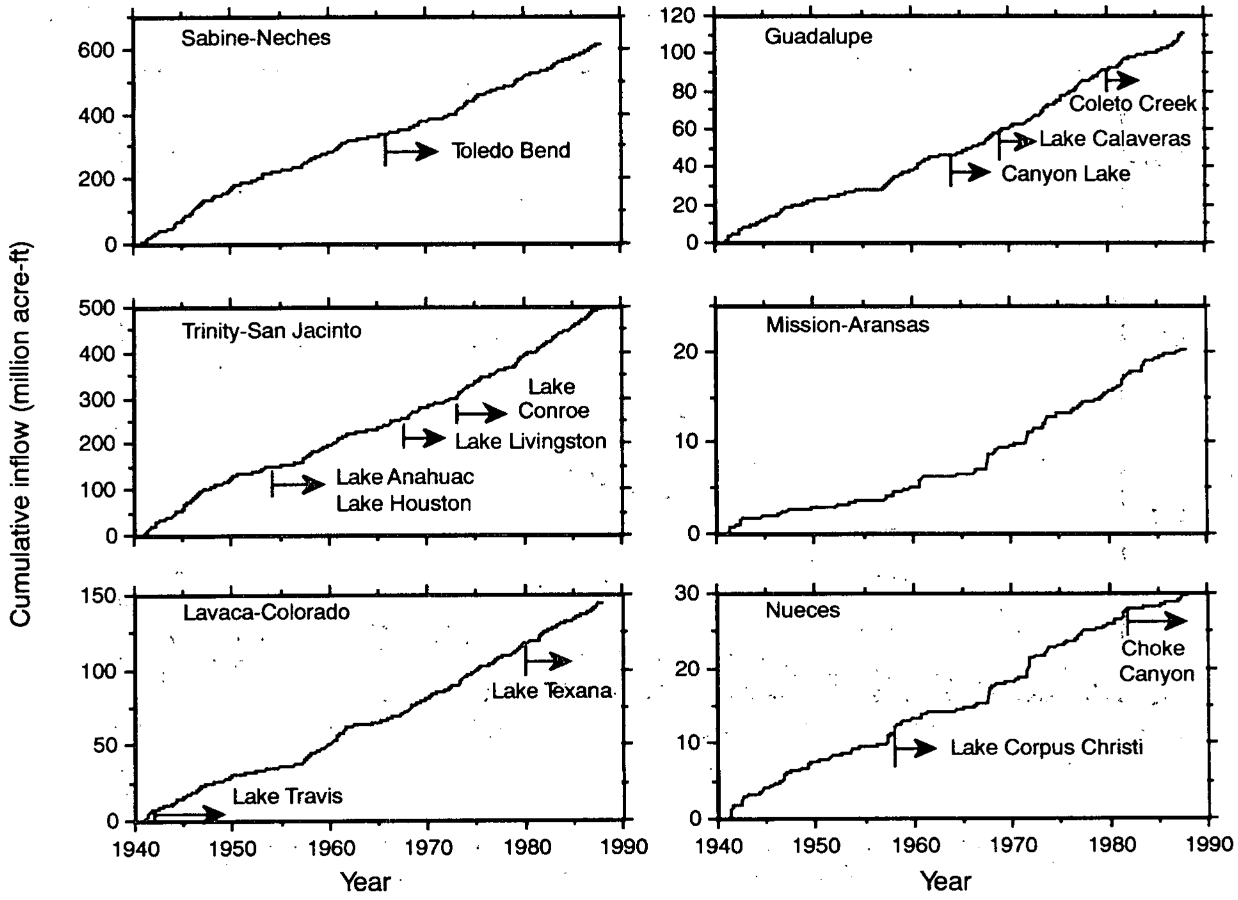


Figure 4.1.12. Cumulative inflow hydrographs (mass curves) for freshwater inflows showing time of completion of reservoir construction.

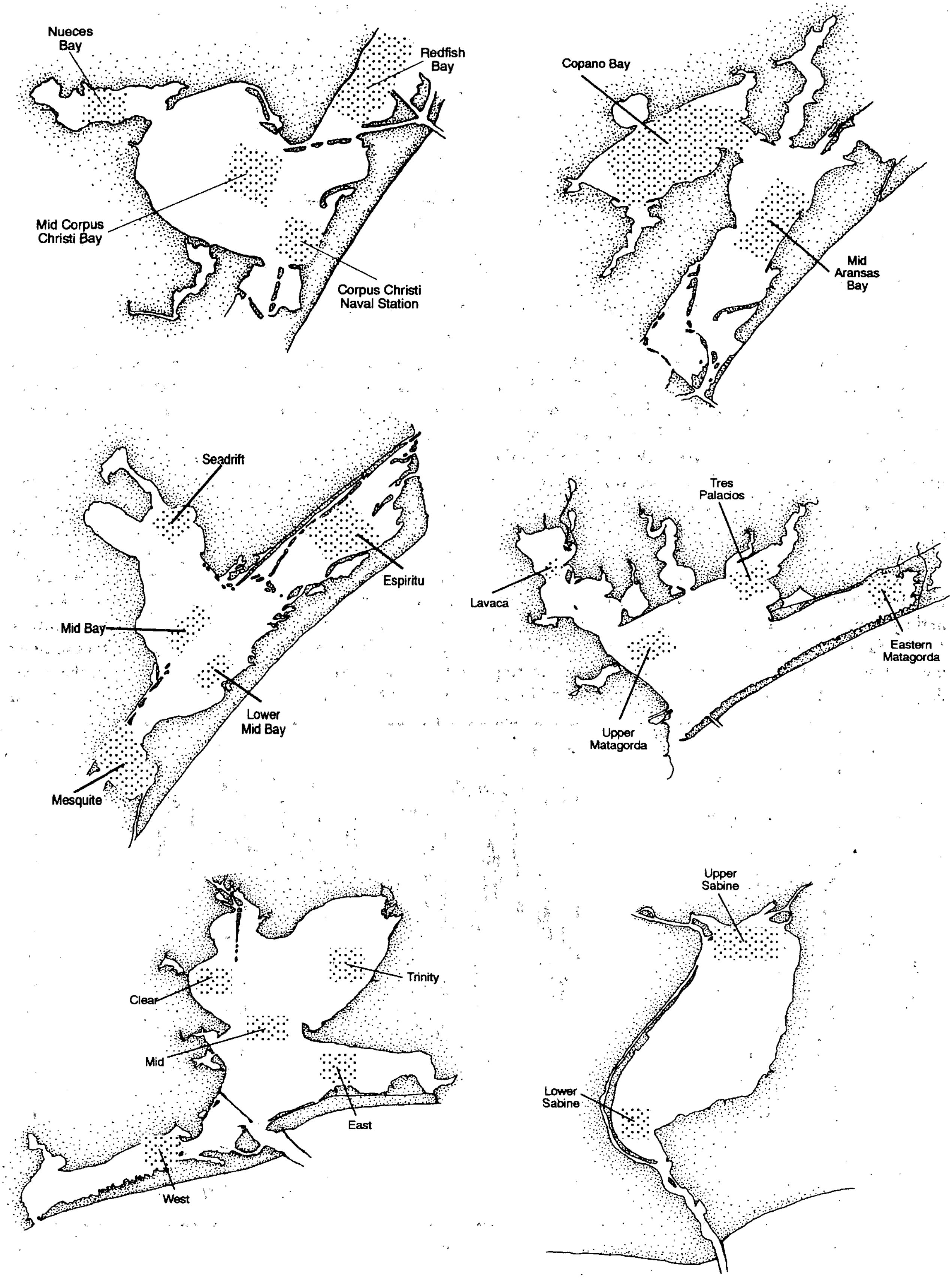


Figure 4.1.13. Salinity measurement sites in the Nueces, Mission-Aransas, Guadalupe, Lavaca-Colorado, Trinity-San Jacinto, and Sabine-Neches estuaries.

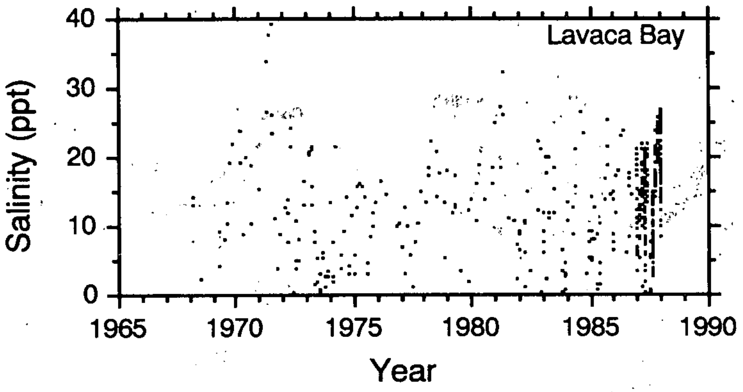


Figure 4.1.14. Typical salinity measurement data set. Daily average salinity measurements in Lavaca Bay.

Typical salinity measurements. Daily salinity measurements for Lavaca Bay from 1968 through 1987, shown in Figure 4.1.14, are typical of the salinity data used here. Prior to 1986, before the TWDB-DN network was operational, very few measurements were taken. Following the implementation of the TWDB-DN network in November 1986, data were taken much more frequently, providing a higher-density data set.

Sample TWDB-DN data are shown in Figure 4.1.15 for Lavaca Bay and Matagorda Bay. Straight line segments in the plot indicate missing data (e.g., June through August, 1987, Matagorda Bay record). In some instances, equipment failure or fouling was suspected. For instance, the steep drop in salinity between January and February 1987 in the Matagorda Bay record is unusual. All datasonde data records were examined for these types of irregularities. Where anomalous data appeared, they were removed to prevent biasing the analysis.

The TWDB-DN data are particularly useful because they are nearly continuous and allow observations not possible with the other data sets. One can determine from this type of data, for example, the magnitude of day-to-day variation and the local response to and recovery times associated with major flood events (May 1987, Figure 4.1.15). Increases and decreases in both the Matagorda and Lavaca bay records follow each other closely. A gradient of roughly 8 to 10‰ is nearly always maintained between these two bays. (The overall average salinity gradient between

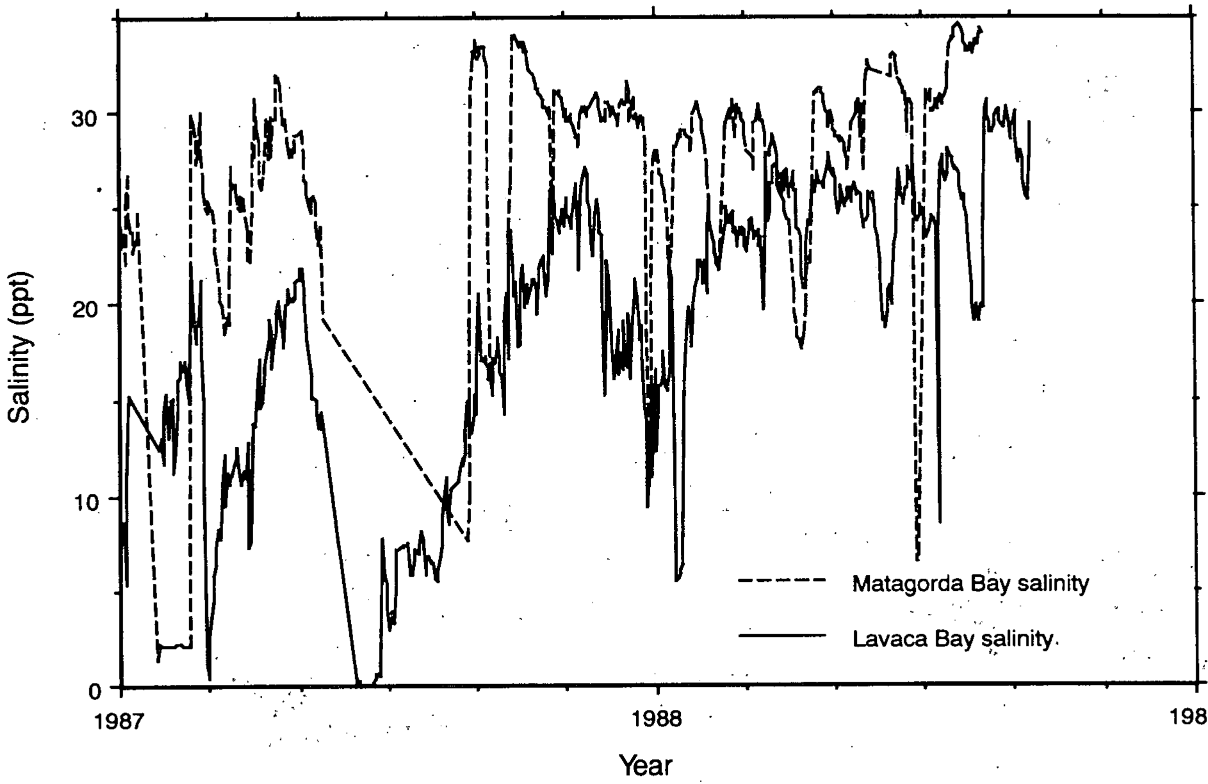


Figure 4.1.15. Typical TWDB-DN (datasonde) data set for Lavaca and Matagorda bays.

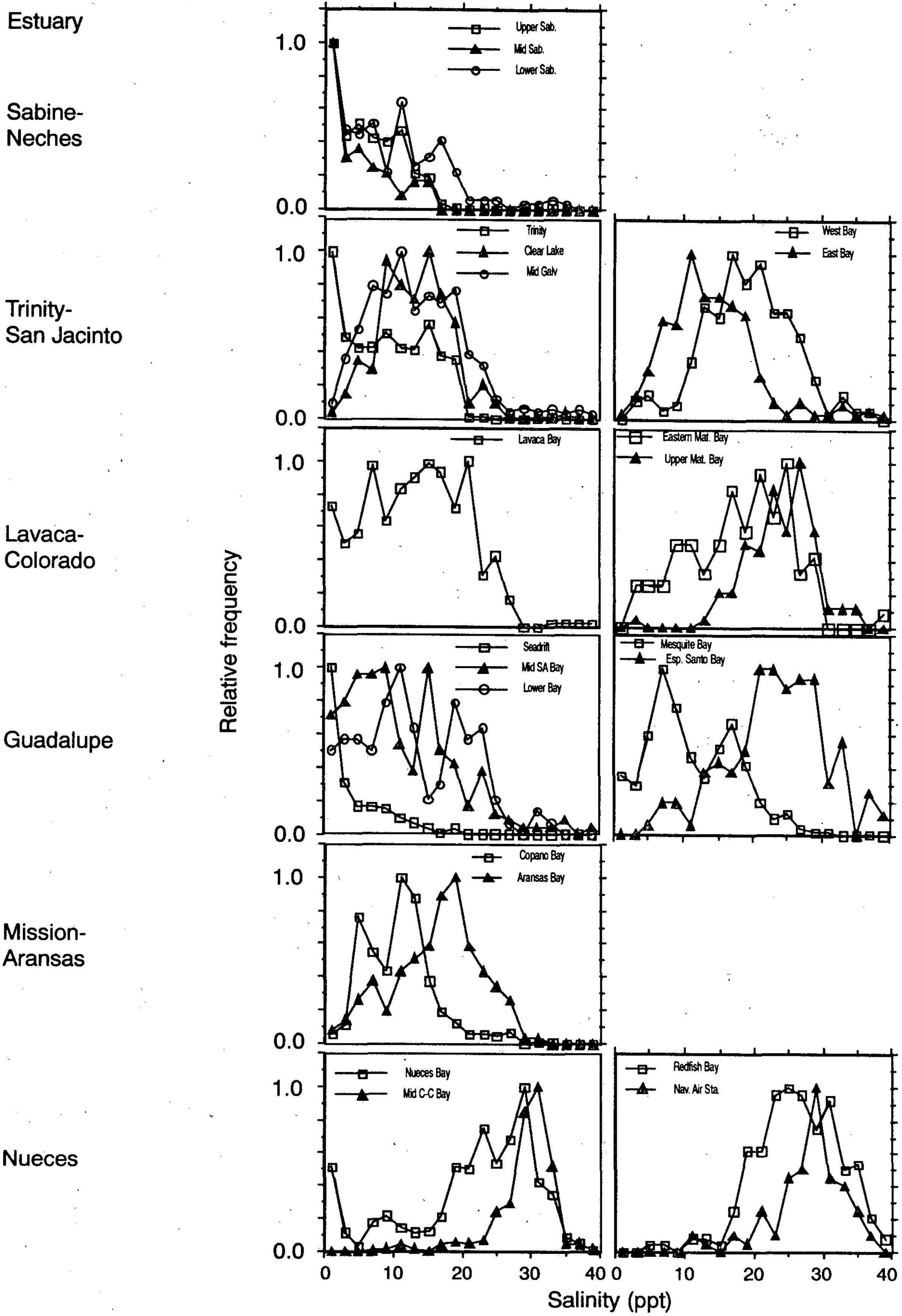


Figure 4.1.16. Frequency distribution of salinity for sites in the Nueces, Mission-Aransas, Guadalupe, Lavaca-Colorado, Trinity-San Jacinto, and Sabine-Neches estuaries.

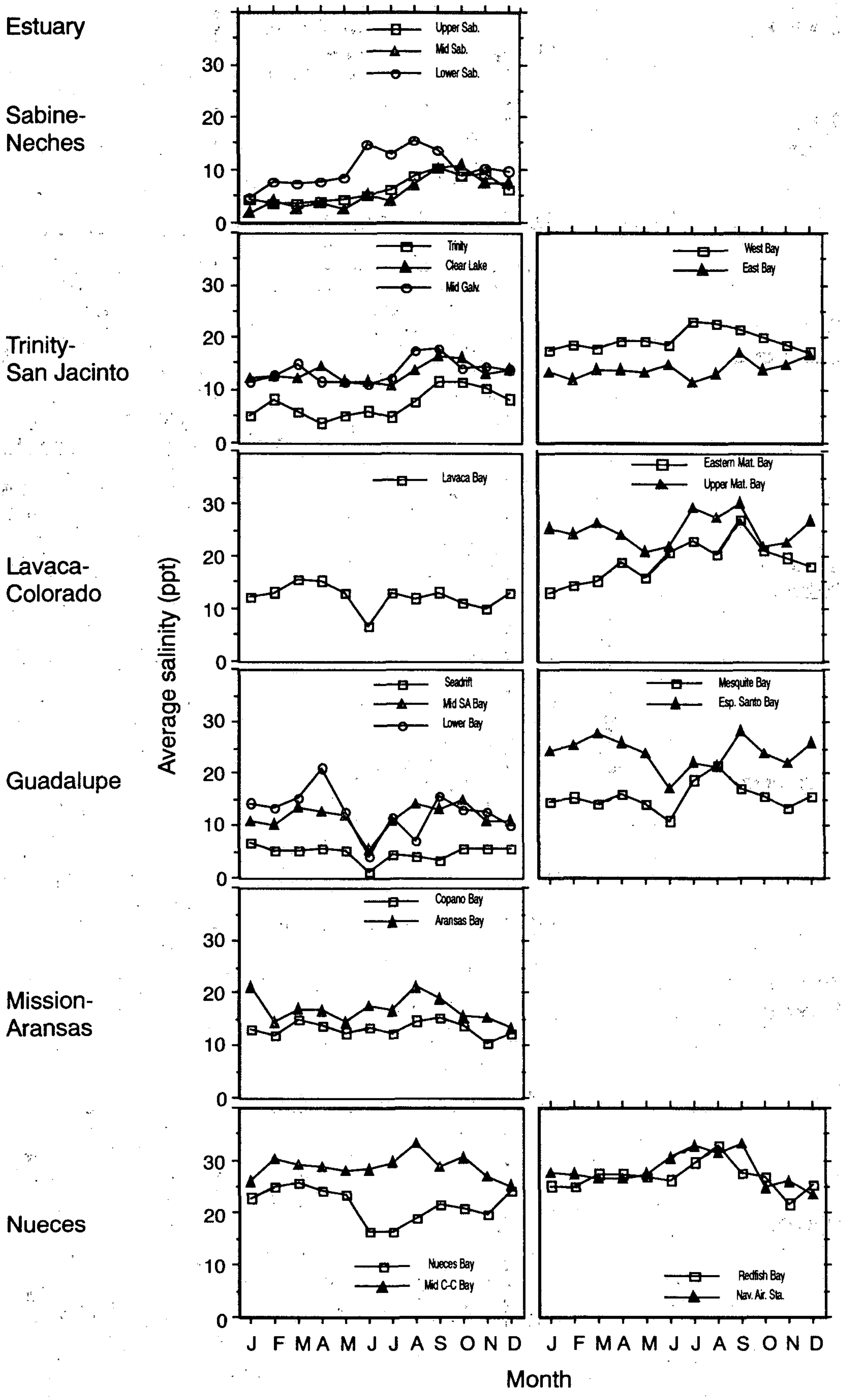


Figure 4.1.17. Monthly salinity distribution for sites in the Nueces, Mission-Aransas, Guadalupe, Lavaca-Colorado, Trinity-San Jacinto, and Sabine-Neches estuaries.

Table 4.1.5. Trend analysis of salinity measurements in Texas estuaries.

Estuary/ Sub-area	z	% change per year	Significant?
Sabine-Neches			
Upper Sabine Lake	-0.99	-1.9	No
Mid-Sabine Lake	0.99	0.2	No
Lower Sabine Lake	-2.36	-3.3	Yes
Trinity-San Jacinto			
Trinity Bay	0.28	0.0	No
Clear Lake	0.38	0.2	No
Mid-Galveston Bay	0.53	0.0	No
West Bay	2.86	1.8	Yes
East Bay	0.69	-0.8	No
Lavaca-Tres Palacios			
Lavaca Bay	-0.04	0.8	No
Eastern Matagorda Bay	-1.08	-0.4	No
Upper Matagorda Bay	0.48	0.1	No
Guadalupe			
Seadrift	1.82	2.7	No
Mid-San Antonio Bay	0.74	0.8	No
Lower Bay	1.97	2.1	Yes
Mesquite Bay	-1.43	-0.9	No
Espiritu Santo Bay	0.43	0.4	No
Mission-Aransas			
Copano Bay	-0.27	-0.4	No
Aransas Bay	1.03	1.8	No
Nueces			
Nueces Bay	2.53	2.1	Yes
Mid-Corpus Christi Bay	1.69	0.1	No
Redfish Bay	2.96	0.8	No
Naval Air Station	2.64	1.1	No

Matagorda and Lavaca bays using the combined data sets is 11.4‰, Figure 4.1.6) However, in some instances, the instantaneous gradient can be much smaller and can even reverse, as during the period March and April 1988. Because measurements from each of the other data sets were taken so infrequently, observations of the type noted above would be very difficult, if at all possible, without the TWDB-DN data set.

Salinity frequency distributions. Salinity frequency distributions, which indicate the likelihood of finding salinity within given limits, are presented in Figure 4.1.16. Frequency distributions are presented for the 22 bay sub-areas. These figures were constructed for each sub-area by sorting the salinity data into 2‰-width bins. The value for each bin was then normalized by dividing the count for that bin by the maximum number found in all bins, so that the maximum for each distribution is one. Very distinct salinity regimes can be seen in these figures. For example, most measurements at each of the sites in the Nueces Estuary are more than 20‰, while in the Sabine-Neches Estuary, most are less than 20‰. Based on these figures, the Nueces Estuary is clearly the most saline, and the Sabine-Neches Estuary the freshest of those studied in this report. The salinity frequency distributions for some sites are broad, indicating that salinities found at these sites are equally likely to fall within a wide range of values. In Lavaca Bay, the distribution is broad and indicates the nearly equal likelihood of finding salinities from 7 to 21‰. At other sites, the distributions are strongly peaked, indicating that salinity is most likely to fall in a narrow range. The frequency distribution peak at 19‰ for mid-Aransas Bay indicates the strong likelihood of finding salinities near that value in mid-Aransas Bay.

Frequency distributions with high, narrow peaks below 2‰, as at Seadrift and at all the Sabine-Neches Estuary sites, indicate the dominant influence of freshwater inflows on salinity at those locations. At other locations, salinity peaks are in the 20 to 30‰ range, as in the Nueces Estuary, Espiritu Santo Bay, and in eastern Matagorda Bay. At these sites, evaporation and mixing with Gulf of Mexico water have a stronger influence on salinity.

Seasonal distribution of salinity. Variations in mean salinity from month-to-month arise from monthly variations in freshwater inflows, evaporation, precipitation, tidal amplitudes, and prevailing wind speed and direction. Because salinity levels are controlled by different mechanisms in each estuary, a wide variety of seasonal salinity patterns exist in Texas estuaries (Figure 4.1.17). For example, maximum salinity levels in the lower Sabine-Neches Estuary occur from June through September, while in Nueces Bay in the Nueces Estuary, maximum salinities occur from December through May. In mid-Sabine Lake, the minimum occurs in January, while in mid-San Antonio Bay, the minimum is in June. Within the Nueces Estuary, such variation exists that minimum salinity levels occur in Nueces Bay, while maximum levels are occurring at the remaining sites in the estuary.

In the Sabine-Neches Estuary, salinity in the winter and spring months (December through May) is lower than in the summer and fall months (June through November). This corresponds reasonably well with freshwater inflow patterns, where inflows are minimum in the summer-fall months and greatest in winter-spring months. This contrasts with the Trinity-San Jacinto Estuary, which has inflow patterns like the Sabine-Neches Estuary. The peak inflow in May (Figure 4.1.8) in the Trinity-San Jacinto Estuary has virtually no effect on salinity in the estuary. Despite tremendous seasonal variation in freshwater inflows to the Trinity-San Jacinto Estuary, the seasonal salinity variation there is among the smallest of all the Texas estuaries.

Average salinity gradient within each estuary also varies throughout the year. In the Sabine-Neches Estuary, the maximum mean salinity gradient between the upper and lower portions of the estuary occurs from May through August (roughly 10‰), and the minimum gradient occurs from September through November (less than 5‰, Figure 4.1.17). The most drastic change in salinity gradient throughout the year occurs in the Nueces Estuary. The gradient between Nueces Bay and all other parts of the estuary from December through May is roughly 5‰. However, in June, the salinity gradient increases to nearly 15‰ and remains at that level through August.

The Guadalupe Estuary exhibits an unusual characteristic in that the seasonal minimum occurs in June at four of five sites in the estuary. This minimum can probably be attributed to the May-June peak in freshwater inflows to the estuary.

Trend analysis. A trend analysis was performed on salinity for each estuary site to determine whether long-term trends (monotonic increases or decreases) in salinity have occurred during the period of record. The seasonal Mann-Kendall test, which is better suited than the Sen test for treating a time series with missing values, was selected for the analysis. The seasonal Mann-Kendall test is described by Van Belle and Hughes (1984).

Output from the seasonal Mann-Kendall test is similar to that from the Sen test; it consists of a statistical parameter, z, which is normally distributed with a zero mean and unit standard deviation. The null hypothesis, that there is no trend, is satisfied for values of z near zero. At the 95% confidence level, the null hypothesis is discounted if z is less than -1.96 or greater than 1.96. Increasing trends are indicated by $z > 1.96$, and decreasing trends are indicated by $z < -1.96$. As in the Sen test, if a trend is indicated by the z statistic, one must next determine whether the magnitude of the trend is significant. The slope of the deseasonalized time series is again used as the test criterion. Generally, trend magnitudes of less than 1 to 2% per year are disregarded as insignificant. The cutoff for significance was taken as 1.5% per year.

Results of the trend analysis are presented in Table 4.1.5. For those cases where a trend is indicated by the magnitude of z, the percent change per year, given by the slope of the deseasonalized time series, is provided. Only in cases where both a trend is indicated by the z statistic and the percent change is greater than 1.5% per year is the trend designated as significant.

The trend analysis indicates that four estuary sites have significant trends in salinity. From 1968 through 1987, mean salinity decreased in the lower region of the Sabine-Neches Estuary at a rate of 3.3% per year, increased in West Bay in the Trinity-San Jacinto Estuary by 1.8% per year, increased in lower mid-San Antonio Bay by 2.1% per year, and increased in Nueces Bay in the Nueces Estuary by 2.1% per year. Although increasing trends were also indicated in Redfish Bay ($z = 2.96$) and Corpus Christi Bay near Laguna Madre ($z = 2.64$), the magnitude of the increases was too small to be considered significant.

The salinity trends discovered by the analysis are not easy to explain. The decreasing salinity trend in the lower Sabine-Neches Estuary cannot be attributed to significant increases in freshwater inflows to the entire estuary, as indicated by the trend analysis on freshwater inflows (Table 4.1.4). Perhaps increases in freshwater inflow from localized areas in the watershed, not large enough to affect the inflow to the entire estuary, have affected salinity only within local regions of the estuary. Similarly, while no trends were found in freshwater inflows to the entire Trinity-San Jacinto Estuary (Table 4.1.4), freshwater inflows may have decreased in the local basin contributing to West Bay, causing local increases in salinity. Although the Sen test indicated a decreasing trend in inflows to the Guadalupe Estuary, the magnitude of the decrease was too small to be considered significant. This again leads to the possibility that localized changes in inflows may have induced the salinity trend at the lower mid-San Antonio Bay site. For Nueces Bay, there are indications in the cumulative inflow hydrograph that inflows decreased over the 1968 to 1987 period, although this is not supported at the 95% confidence level by the Sen test described earlier. Nonetheless, a decreasing trend in freshwater inflows was found at the 80% confidence level, indicating that decreasing freshwater inflows should not be discounted entirely as an explanation for the rising salinity trend in Nueces Bay. Besides local changes in freshwater inflow rates, long-term trends in salinity might also be caused by other factors which influence mixing of fresh water with saline Gulf of Mexico water within the bay, such as the opening or closing of passes to the Gulf.

4.2 DISTRIBUTION OF NUTRIENTS IN TEXAS ESTUARIES

Introduction

Texas estuaries have complicated morphometries. Most Texas estuaries have several sources of inflow, with bays of various sizes. Some have deep channels adjacent to broad shallow expanses. There is broad circulation in some bays while others are isolated from pathways of water exchange. This physical complexity produces spatially varied chemical and biological patterns within each estuary. The

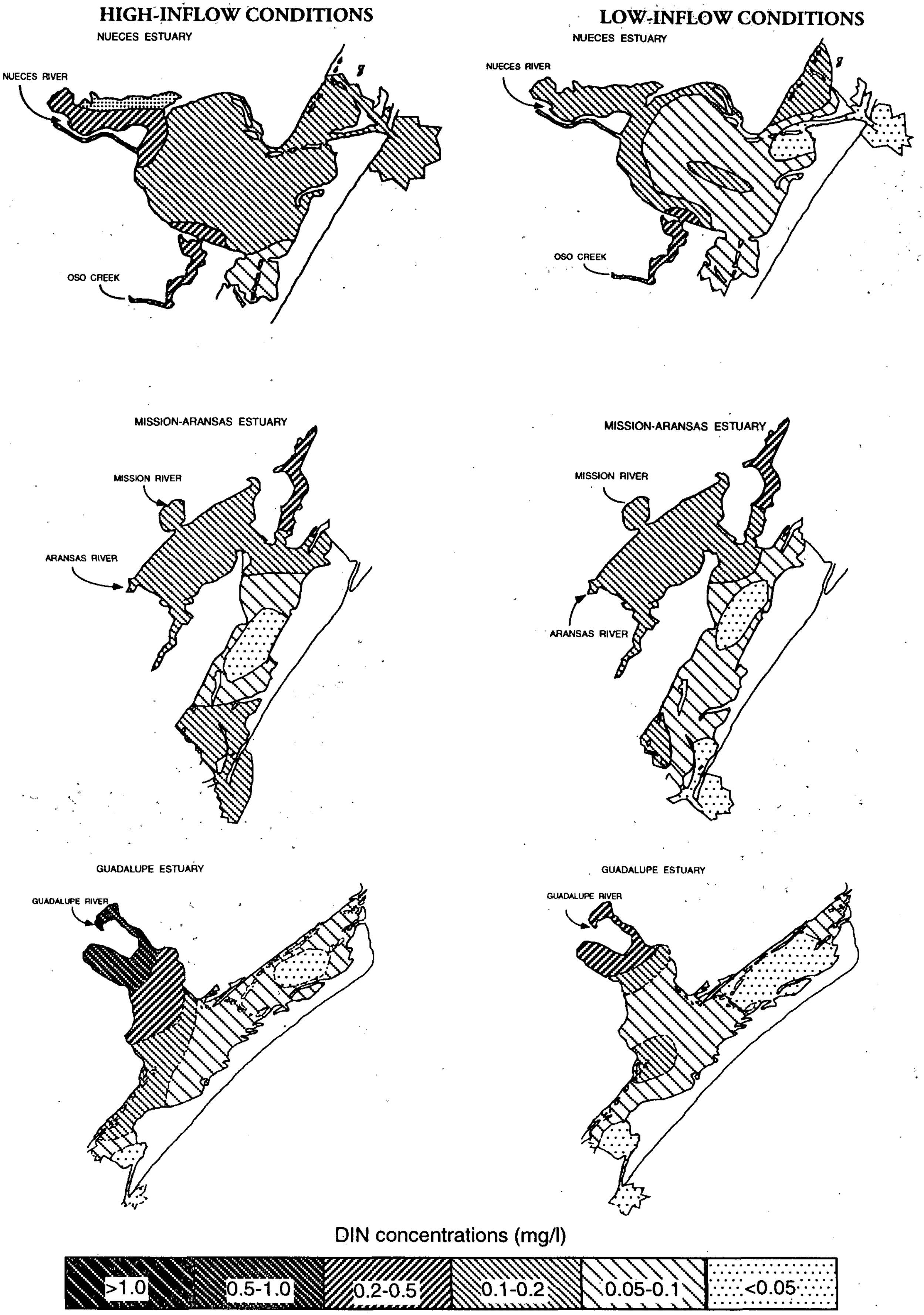
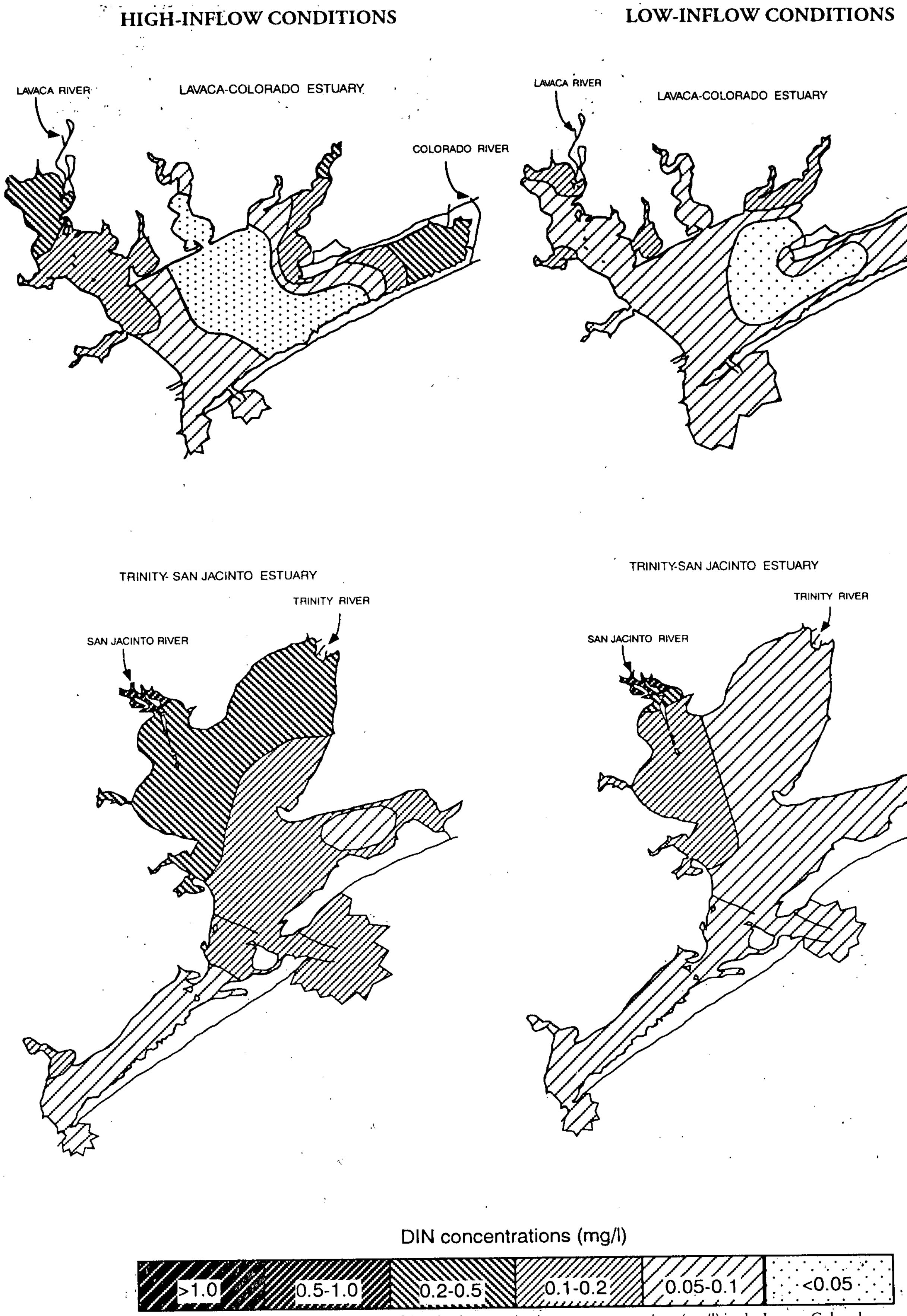


Figure 4.2.1. Average distributions of dissolved inorganic nitrogen concentrations (mg/l) in the Nueces, Mission-Aransas, and Guadalupe estuaries during periods of high and low inflows.



HIGH-INFLOW CONDITIONS

LOW-INFLOW CONDITIONS

Figure 4.2.2. Average distributions of dissolved inorganic nitrogen concentrations (mg/l) in the Lavaca-Colorado and Trinity-San Jacinto estuaries during periods of high and low inflows.

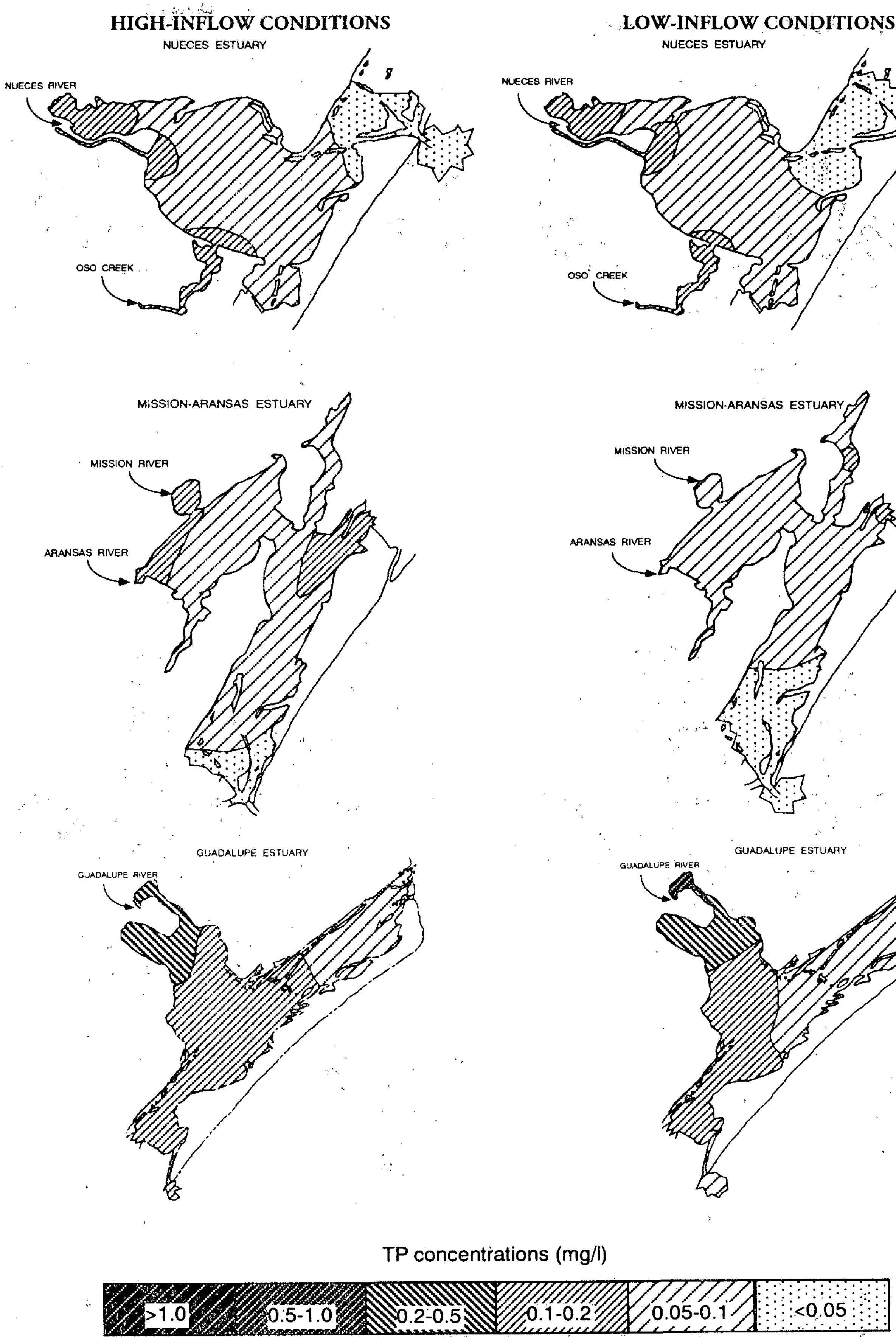


Figure 4.2.3. Average distributions of total phosphorous concentrations (mg/l) in the Nueces, Mission-Aransas, and Guadalupe estuaries during periods of high and low inflows.

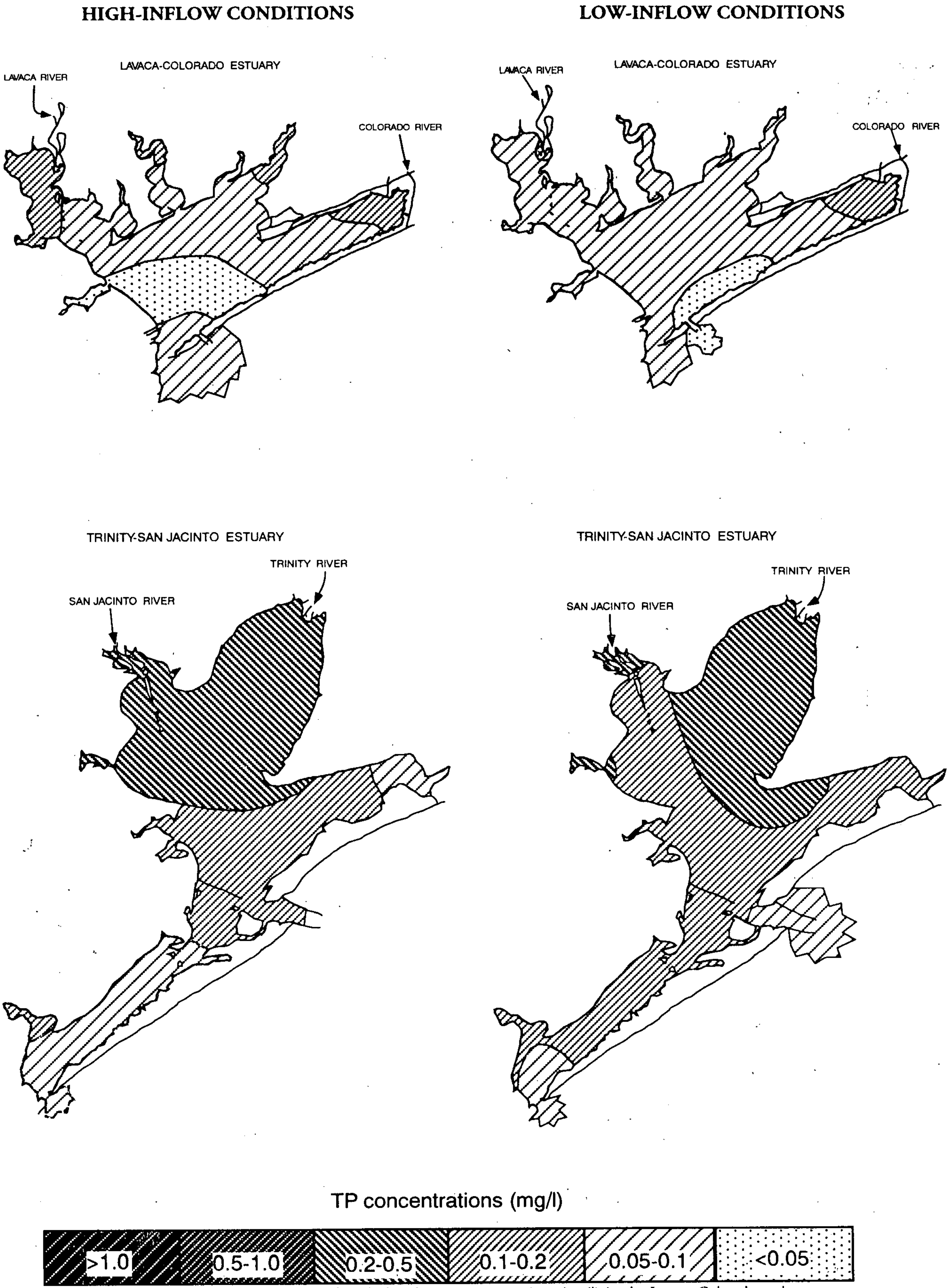


Figure 4.2.4. Average distributions of total phosphorous concentrations (mg/l) in the Lavaca-Colorado and Trinity-San Jacinto estuaries during periods of high and low inflows.

spatial patterns of dissolved and particulate nutrients help determine which parts are productive and what plant and animal communities will be favored in those areas. The pattern of nutrients in the waters also shows the interactive influence of (1) inputs from various sources—including municipal and other wastes, (2) physical transport of water masses, and (3) effects of biotic metabolism—including nutrient uptake by plants.

In this section, maps of historical data from many sampling stations within the bays are presented to illustrate general patterns of nutrient distribution. The map data are categorized with respect to the inflow regime that prevailed at the time of sampling to show how inflow rates determine nutrient patterns. Investigations were carried out to gain more detailed data on the nutrient distributions within several bays and to explain the distributions in terms of the physical and biological processes at work in the systems. Results from these studies demonstrate aspects of nutrient distributions important to all our estuaries. The discussion shows how, in concept, the spatial organization of physical and biological components of the estuary is related to the patterns of nutrient distributions. This organization is influenced by the freshwater inflow an estuary receives in proportion to its size.

Long-term Average Nutrient Distributions

Map data. Data were compiled from the Statewide Monitoring Network of the TNRCC, the Coastal Data System of the TWDB, and from studies funded by the TWDB to investigate specific estuaries. For many of the stations, data are available beginning in 1967 or 1968. Although these data collection programs represent respectable efforts to monitor conditions in all the major bays, the stations are spread rather thinly for the purposes of mapping distributions of dissolved substances. Therefore, the maps presented here offer only approximate boundaries at a coarse scale of resolution.

Maps of nutrient distributions are presented for two categories of freshwater inflow, high and low. Nutrient concentration data were sorted into two categories based on the freshwater inflow volume to the respective estuary during each month that samples were collected. The median monthly inflow volumes from Table 4.1.1 were used as threshold levels for each estuary. For both inflow categories, average concentrations were calculated for each sampling station, combining data over the years and seasons. Stations sampled infrequently (less than 10 times over the period of record) were not included in the mapping.

Dissolved inorganic nitrogen. The concentrations of ammonia, nitrite, and nitrate nitrogen are often summed into a parameter referred to as dissolved inorganic nitrogen (DIN). DIN represents the nitrogen most available to meet nitrogen requirements of phytoplankton production. Ammonia may come from benthic metabolism in the sediments, remineralization in the water column, or from municipal or industrial discharges. Nitrogen in river water is primarily nitrate. A concentration of DIN greater than 0.028 mg/l generally supports maximal growth rates among many phytoplankton species, assuming other conditions are favorable (O'Connor 1981; Whitledge 1989).

In figures 4.2.1 and 4.2.2, the general distribution of dissolved inorganic nitrogen is illustrated for the five central estuaries of the Texas Coast—the Nueces Estuary, Mission-Aransas Estuary, Guadalupe Estuary, Lavaca-Colorado Estuary, and Trinity-San Jacinto Estuary. Enlightening contrasts between estuaries can be made by comparing areas within each estuary at concentrations greater than 0.1, between 0.1 and 0.05, and less than 0.05 mg/l. In both inflow regimes, the Lavaca-Colorado Estuary has the greatest area of water with low DIN concentrations (< 0.05 mg/l). The main body of that estuary does not experience a marked change in nitrogen concentrations with changes in inflow. The upper reaches of all the estuaries maintain a nitrogen-rich zone even in dry months; during wet months, nitrogen concentrations increase in the heads of the estuaries by a factor of two or more. High inflows increase DIN in the upper Guadalupe Estuary by a factor of ten. The lagoonal arms of the lower coast estuaries have zones of low nitrogen concentrations even during months of high inflows. Since the water in these portions of the estuaries tends to be clearer, the phytoplankton of the lagoonal arms are best able to make use of available nitrogen, so the zones of low nitrogen may actually be zones of high nitrogen use and efficient recycling. During high inflow months, outwelling of high nitrogen water (> 0.1 mg/l DIN) to the Gulf of Mexico is characteristic of the Nueces and Trinity-San Jacinto estuaries.

Total phosphorus. Phosphorus comes into the estuary mainly through freshwater inflows and municipal discharges. Exchange between phosphorus dissolved in the water and bound to sediment particles also influences the availability of this major nutrient. Concentrations of total phosphorus (TP) within the estuaries at high and low inflows are shown in figures 4.2.3 and 4.2.4. The Lavaca-Colorado estuary shows low concentrations in Matagorda Bay, which remain or expand during months of high inflow. High inflow produces expanded areas of phosphorus-rich water in upper San Antonio Bay and in Galveston Bay, but the change is not dramatic. Historic distributions of TP do not show areas of significant depletion in any of the estuaries. The Guadalupe and Trinity-San Jacinto estuaries are richer in TP than the others, but concentrations of phosphorus in all these estuaries are typically at levels greater than 0.005 mg/l, which

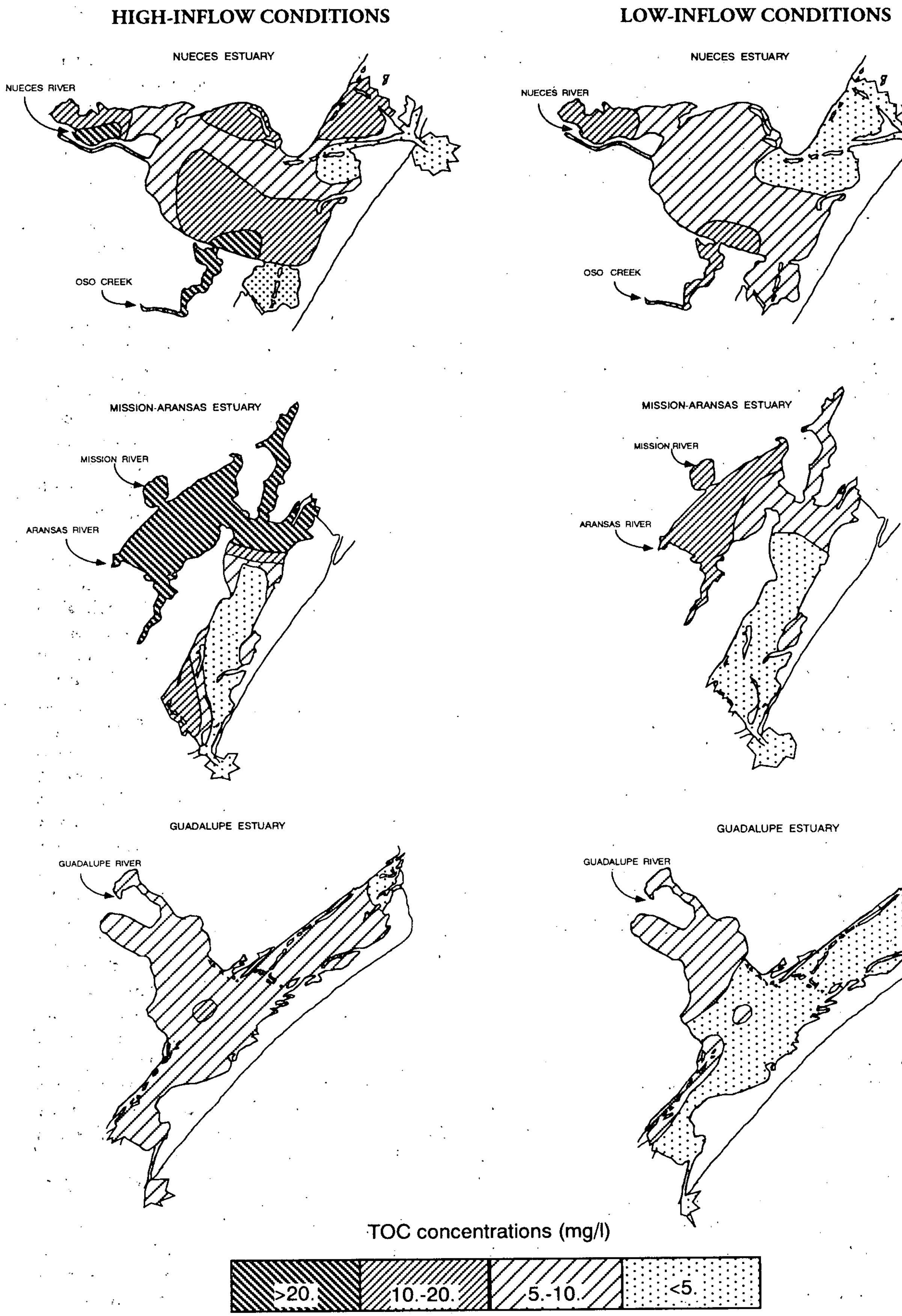


Figure 4.2.5. Average distributions of total organic carbon concentrations (mg/l) in the Nueces, Mission-Aransas, and Guadalupe estuaries during periods of high and low inflows.

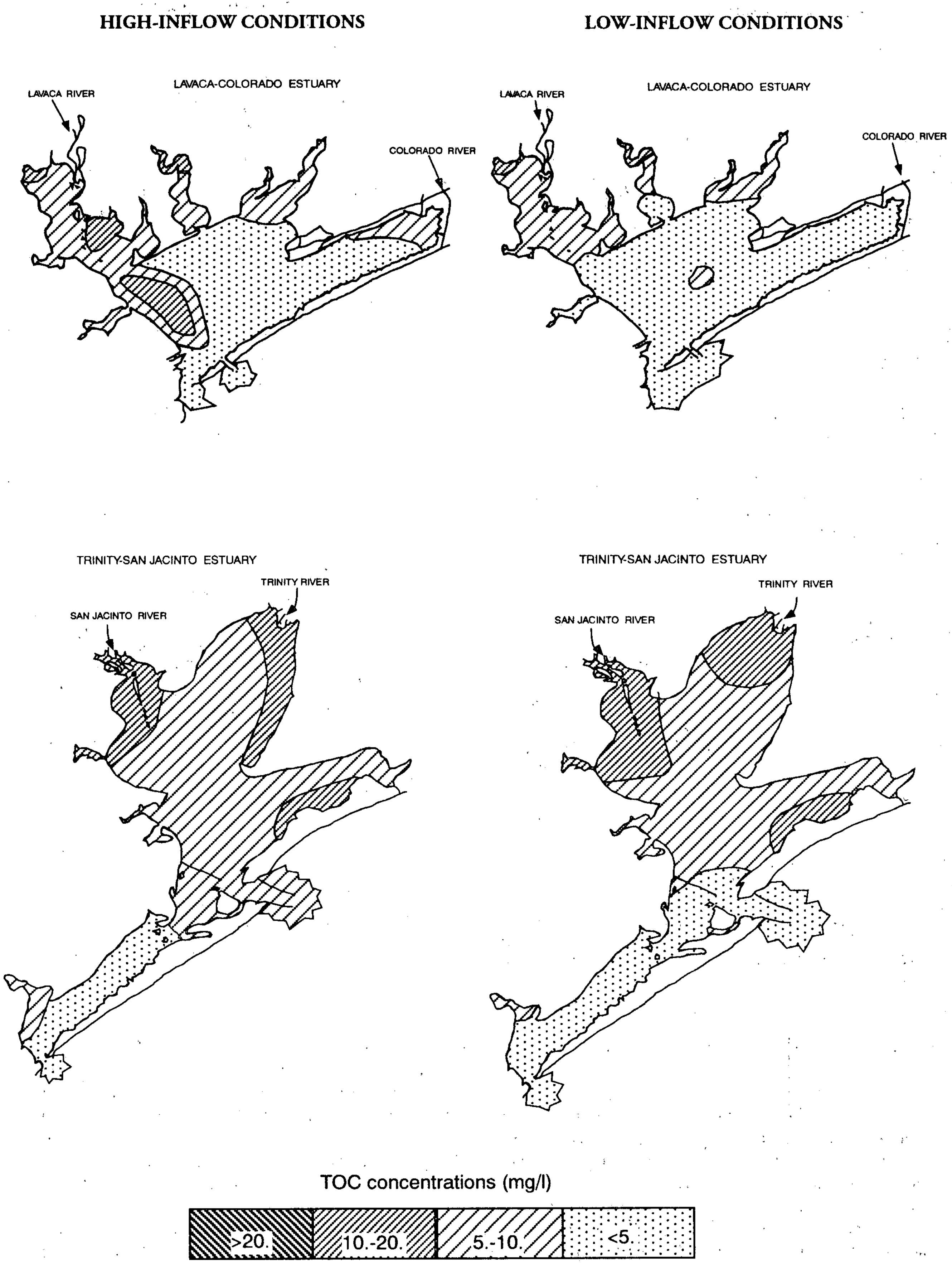


Figure 4.2.6. Average distributions of total organic carbon concentrations (mg/l) in the Lavaca-Colorado and Trinity-San Jacinto estuaries during periods of high and low inflows.

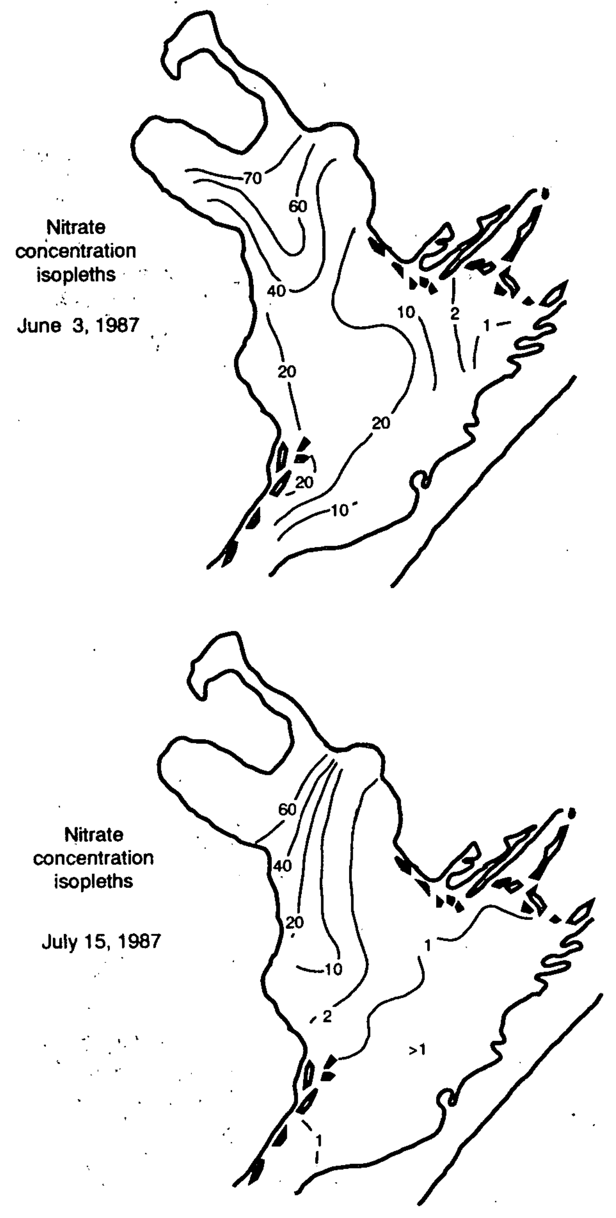


Figure 4.2.7. Isopleths of nitrate concentrations (μmole/l) in San Antonio Bay during high-inflow period of 1987. High concentrations along the western shore show the freshwater plume. Note decline of concentrations from upper to lower bay.

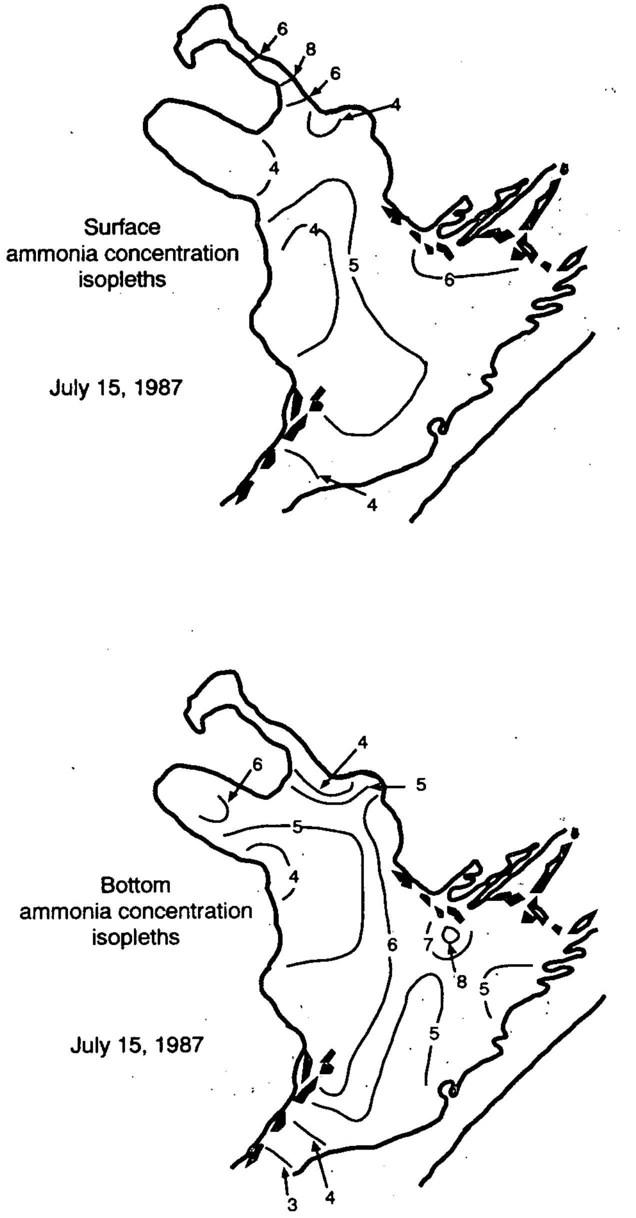


Figure 4.2.8. Vertical stratification of ammonia concentrations (μmole/l) in San Antonio Bay.

would support rapid phytoplankton growth (O'Connor 1981).

The main mechanism controlling phosphorus concentrations may be geochemical. Phosphorus can adsorb to clay particles and to large organic molecules and become bound into particulates which settle or are "salted out" in the upper reaches of an estuary. This process is reversible and its dependence on the ambient phosphorus concentration is such that the bottom sediments of an estuary may buffer the water column phosphorus concentration. This mechanism and the additional involvement of sediment microorganisms in the exchange of phosphorus between sediment and water column have been studied by Pomeroy et al. (1965). Suspension of bay-bottom sediments increases the rate of exchange of phosphorus between particulate and dissolved phases (Pomeroy et al. 1965; Montagna et al. 1989).

Total organic carbon. Dissolved and particulate organic carbon in the estuary derive from organic materials

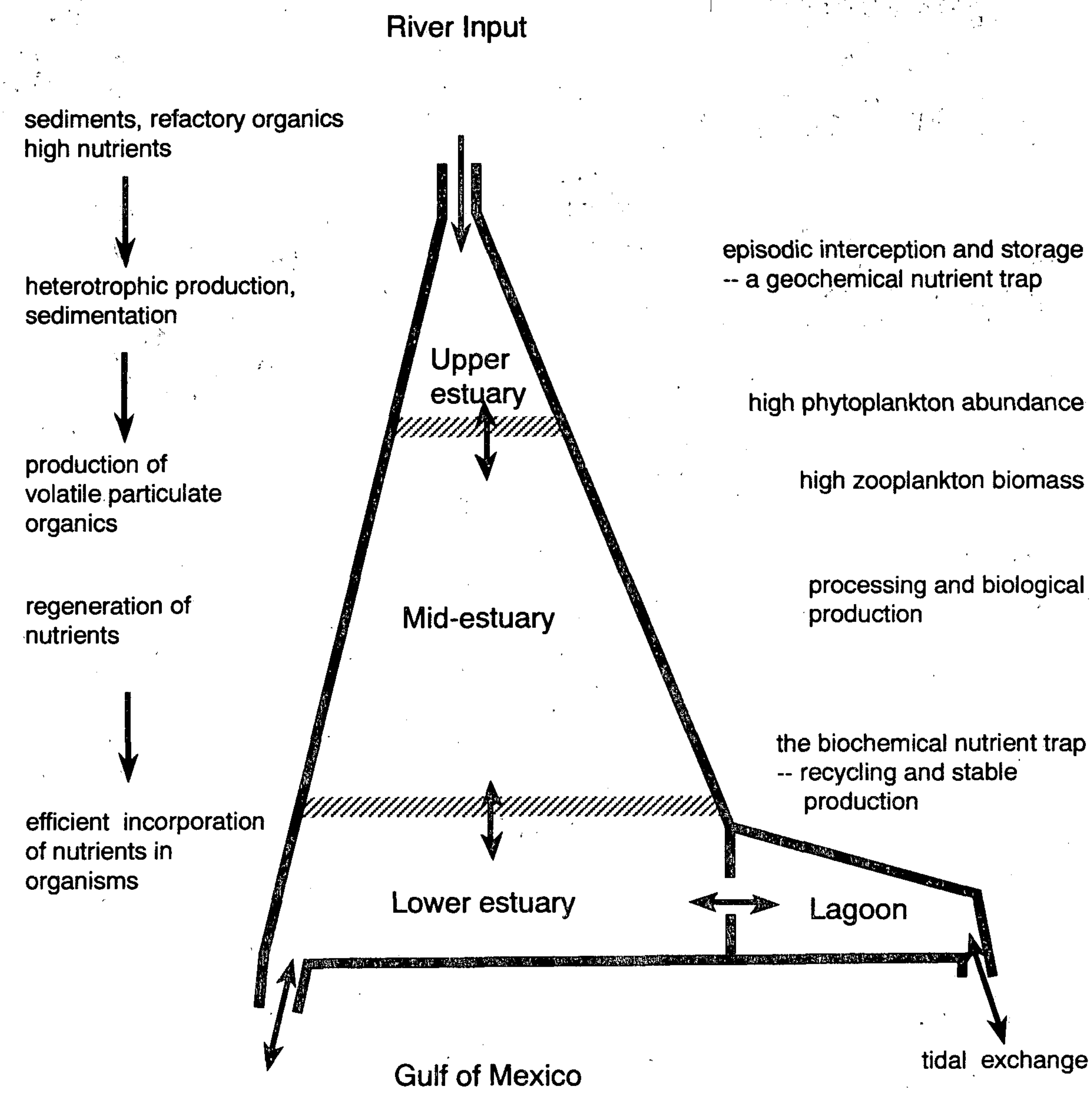


Figure 4.2.9. Conceptual zonation in an estuary with respect to nutrient processing.

carried in by river flows, from primary productivity, and from partial metabolic breakdown of organic materials recycled within the estuary. Section 5.6 summarizes some studies that have sought to determine the relative importance of these possible sources of carbon to the estuarine food chain. The distributions of total organic carbon (TOC) in figures 4.2.5 and 4.2.6 support expectations of higher concentrations near the river mouths. There are also areas of high TOC concentrations adjacent to some barrier island wetlands. Within most of the other estuaries, high flows deliver large concentrations of organic carbon down-estuary. However, the distribution of TOC within the Trinity-San Jacinto Estuary apparently does not depend to a great extent on the monthly inflow volume. In addition, the distribution of TOC in Corpus Christi Bay suggests phytoplankton production responding to inorganic nutrients, rather than a plume of carbon from the Nueces River. This is consistent with findings of the studies of this estuary discussed below.

Nutrient Distributions—Detailed Studies

Of the major studies which have been completed on the estuaries of the middle coast, several were designed to chart the distribution of chemical and biological parameters in the estuary, and to determine the influence of freshwater inflow on these distributions. Many of the results of these studies can be extended to help explain patterns of nutrient concentrations and estuarine productivity in other estuaries.

Spatial variation. The nutrients brought into an estuary by flood flows of a major river do not appear uniformly distributed throughout the receiving bay. In San Antonio Bay, detailed water quality surveys during the high flows of 1987 revealed that the fresh water traversing the bay could be seen clearly as a plume of nutrient-rich water, apparently steered by oyster reefs, dredge cuts, and Coriolis forces, flowing predominantly along the western shore (Figure 4.2.7, from Figure 6 in Whitledge 1989). Within the freshwater plume, biological and geochemical processes rapidly removed nitrogen as the waters moved down the estuary. Nutrient-laden material accumulated in sediment deposits along the track of the plume. In the months following the flood, this deposition could be seen as a zone of ammonia-rich and oxygen-depleted water, indicating areas where high rates of decomposition of deposited material were occurring (figures 12 and 13 in Whitledge 1989).

The influence of the Nueces River on the short-term pattern of nutrient concentrations in the Nueces Estuary is largely confined to Nueces Bay (figures 5, 8C, 40, and 43 in Whitledge 1989). Influence of the delivery of nutrients on the lower bay is indirect, through the production and transport of organisms to the lower bay, or by transport of nutrients regenerated from Nueces Bay sediments. Supporting this idea of indirect effect is data that indicates the main nitrogen source for phytoplankton productivity in Corpus Christi Bay is ammonia rather than nitrate (Whitledge 1989). Ammonia results from recycling of fixed nitrogen, whereas nitrate is indicative of direct river nitrogen supply (McCarthy 1981). Corpus Christi Harbor was often a larger source of nitrogen to Corpus Christi Bay than was the outflow from Nueces Bay (Whitledge 1989, figures 5, 5C, 8, 8C, 40, and 43). Inflow from Oso Bay also influenced nitrogen distribution in a portion of Corpus Christi Bay.

Although deep ship channels are thought to exert some control over the water circulation in some bays, the distributions of dissolved constituents in Corpus Christi Bay do not indicate a consistent influence of circulation along or moving out from the ship channel. Consequently, the gradients and zones of nutrient concentrations within the central bay are more likely the result of the benthic and planktonic metabolism of the waters than circulation-induced patterns (Whitledge 1989).

In the Nueces Estuary, concentrations of nitrogen, phosphorus, and silicate were usually high enough during the periods studied to support maximum phytoplankton productivity. However, minimum levels of all nutrients were occasionally encountered which would limit productivity for short periods at some locations.

Studies in the Lavaca-Colorado Estuary over the period 1984-1986 (Jones et al. 1986) did not focus on determination of spatial distributions of nutrients, but these investigations do show general relationships involving water quality and biological processes in the Lavaca Delta. At moderate river flows, significant processing of nutrients that were delivered to the estuary occurred in the lower stretches of the river and associated deltaic bayous. Occasionally, when a freshwater lens formed over saltier tidal water in the river channel, the rate of metabolism was high enough to deplete dissolved oxygen in the bottom water (Jones et al. 1986). Whitledge (1989) suggested that a similar high rate of nutrient incorporation was occurring in the Nueces River delta region. In both cases, the rate of biological processing of nutrients in the river water appears to be more rapid than the rate of nutrient renewal from river transport. The result is transformation of dissolved nutrients to organism biomass within the lowest reaches of the river and within the river delta. Therefore, in these two systems, the concentrations of dissolved nutrients in the tidal portion of the river at the head of the estuary may not fully represent the river's delivery of nutrients to the system (delta plus estuary).

Temporal and vertical variation. Detailed descriptions of the variability of bay water nutrient concentrations over the diel cycle are presented in Whitledge (1989). Within a water mass, the uptake and regeneration activities of planktonic organisms change the concentrations of dissolved materials during the daily cycle as the activities of the organisms respond to changes in light and temperature. In Corpus Christi Bay, temporal variability was small due to the lack of strong surface salinity gradients. Changes in nutrient concentrations due to wind mixing were more important to short-term variation than biologically mediated effects.

Nutrient and salinity concentrations may also vary with depth, even in shallow Texas bays which are often well mixed by wind-induced turbulence. However, vertical stratification can occur in navigation channels, deeper passes, and bayous where salinity increases with depth. In these channels, tidal waters with higher salinity, lower dissolved oxygen, and moderate nutrient concentrations may flow under fresher waters with higher dissolved oxygen and lower concentrations of nutrients. Detailed surveys of the Nueces and Guadalupe estuaries occasionally revealed a shallow layer of water along the bay bottom in central bay areas, which was more saline than the overlying water. This lower layer of water can develop nutrient concentrations very different from the concentrations in surface layers, due to sediment biochemical processes. The difference in salinity may impede exchange between layers. Figure 4.2.8 (from Whitledge 1989, figures 11, 12) illustrates the differences between surface and bottom concentrations of ammonia in San Antonio Bay. In Corpus Christi Bay, Laguna Madre was identified as the likely source of more saline and therefore denser water flowing along the bay bottom. In San Antonio Bay, Whitledge (1989) suspected that denser water from the estuarine section of the Victoria Barge Canal was flowing out in a thin layer over the central bay bottom.

In many estuaries of relatively simple shape and deep waters, outflowing fresh water is commonly stratified over the denser sea water flowing into the estuary. The flow of the saltier water up the estuary in response to tidal forces and density differences has been cited as a mechanism effective in helping the estuary retain nutrients (Pritchard 1967a). Particulates settling from the overlying water layer are transported back into the estuary instead of continuing out. In the relatively shallow estuaries of the Texas coast, stratification would be expected to be infrequent. Yet, stratification apparently can develop and could act to circulate nutrient-laden bottom waters independently of other circulatory mechanisms.

Discussion

Process zones. Conceptually, a Texas estuary can be divided into major zones of physical and biological processes to explain the observed patterns of nutrient distributions. These zones are illustrated diagrammatically in Figure 4.2.9, building on the conceptual organization of Delaware Bay presented by Sharp et al. (1984), on figures presented in Blanton et al. (1971), and from ideas in Fisher et al. (1988).

Upper bay zone. At the head of the estuary, geochemical mechanisms cause nutrient-laden material to deposit on the sediment surface, lowering the concentrations in the influent plume. Deltaic marsh vegetation may augment this process. In the upper bay, organisms use dissolved nutrients to fuel the breakdown of particulate organics such as bits of leaves and twigs. Wind provides energy to periodically mix and redistribute materials, promoting heterotrophic activity. Regeneration of dissolved nutrients from the sediments occurs (Section 5.4), but turbidity suppresses phytoplankton growth and so prevents phytoplankton from taking up all available nutrients. Therefore, during high-inflow periods, nutrients accumulate in the sediments, or pass through the zone. During low-inflow periods, regeneration of nutrients from sediment storage exceeds uptake and nutrients are passed down to the next zone.

Mid-bay zone. The mid-bay region, with typically greater volume and water depths, has less turbid waters. This allows phytoplankton to use the dissolved nutrients and reduce nutrient concentrations. Benthic metabolism of river-borne materials is also important, with processing rates as high as rates in the upper bay (Montagna et al. 1989). In the mid-bay, however, more of the regenerated nutrients are incorporated into planktonic cells. Zooplankton find salinity levels in the mid-bay region accommodating, and add an important link in the food chain. With the development of a planktonic food chain comes regeneration of dissolved nutrients in the water column. Oyster reefs, which find environmental conditions most suitable in this part of the estuary, also regenerate nutrients from the particulates they consume. An oyster reef may remove a substantial fraction of plankton and particulates which wash over it. Some of this material leaves the reef as dissolved waste products.

Lower bay zone. The lower bay provides a more consistent habitat for marine and estuarine organisms. Favorable habitat promotes higher species diversity, more complete food chains, and more efficient use of resources. Increased efficiency leads to a concomitant decrease in the concentrations of available nutrients. Fixed materials go through longer cycles within biological compartments before becoming available as dissolved nutrients. Light penetration is typically greatest in the lower bay and lagoonal arms of the estuary, so there is a greater volume of phytoplankton capable of growing (and using nutrients) at a high rate. Recycling is important in this section of the estuary. The uptake of dissolved products of nutrient regeneration is rapid, so the concentrations in the water remain low. In the lower bay, tidal exchange with the Gulf provides an additional source and sink for nutrients. Nearshore Gulf waters typically have concentrations of nutrients similar to or slightly less than lower bay concentrations. Therefore, Gulf exchange serves to moderate variation in nutrient concentrations in this zone.

Lagoonal arms. Major inflow events modify conditions in most parts of the typical estuary, but some of the lagoonal arms remain isolated from the mainstream of this flow. The lower level of flushing and salinity change does not remove or impair the biological community to a great extent in these arms. During periods of highest inflow, this area may still be efficient in processing dissolved nutrients, so swings in concentrations are uncommon. In addition, good light penetration in the lower bay and lagoonal arms promotes growth of submergent vegetation. These plants take up nutrients through roots in the sediment, serving to transport nutrients stored in the sediment to active biomass within the system. To a certain extent, the inflow-related delivery of nutrients to the lower bay keeps this area productive. In return, the lower bay serves to restock the upper zones with estuarine species after the occasional flushing effects of floods.

Differences and similarities in Texas bays. The above generalization of estuary organization may help classify Texas bays by dominant functions. Most of Sabine Lake may function as an upper estuary. Lower estuary functions may occur only in a limited zone of the Trinity-San Jacinto Estuary. Conversely, at times of low inflow in the Nueces Estuary, the functions and nutrient processing of the upper

Table 4.3.1. Mass loading (million gm/yr) of major nutrients into Texas estuaries, based on 1977 to 1987 data from the TNRCC and USGS.

Estuary	Total nitrogen	Total phosphorus	Total organic carbon
Nueces	2,300	470	6,600
Mission-Aransas	1,600	340	10,000
Guadalupe	8,900	1,900	28,100
Lavaca-Colorado	6,800	1,020	41,900
Trinity-San Jacinto	44,800	10,000	212,000

estuary may be compressed into the tidal stretch of the Nueces River. Each estuarine zone has prevalent geochemical or biochemical mechanisms to trap nutrients. Therefore, estimating the relative importance of the conceptual zones in an estuary may help explain general features of the nutrient distributions.

Despite the vast differences between loading input rates to the estuaries (Section 4.3), it is remarkable that the range of average nutrient concentrations is similar for all the bays, excluding isolated lagoons. Concentrations are apparently determined more by the differential rates of geochemical trapping in the upper bays versus the rates of regeneration than by the total loading per se. That the concentrations are frequently high with respect to the needs of phytoplankton production can be attributed to the rapid rates of regeneration at ambient water temperatures and to the commonly high turbidities which limit the ability of phytoplankton to exploit all available nutrients.

4.3 NUTRIENT LOADING TO TEXAS ESTUARIES

Introduction

One reason that estuaries support high biological productivity is the nutrients they receive from rivers and tributaries draining terrestrial watersheds. Texas estuaries consist of numerous bays which provide habitats for many organisms to take, use, and recycle incoming nutrients. Although there is considerable variation among estuaries in the timing of riverine inputs, oceanic influence, mixing, and anthropogenic influence, the estuaries which receive more nutrients are more productive in general. However, a high nutrient input rate can cause problems. For some estuaries, there is concern that increasing inputs of nutrients from municipal wastes have contributed or will contribute to problems associated with eutrophication, such as fish kills from low dissolved oxygen levels.

The major nutrient elements that fuel estuarine production include carbon, nitrogen, phosphorus, and silicon. Carbon enters the system via allochthonous inputs from rivers and photosynthetic fixation of dissolved carbon dioxide. Phosphorus and nitrogen input amounts are generally considered most important in determining the productivity of the majority of estuaries (Davis 1973). Silicon is an important nutrient for diatoms and some chrysophytes of the estuarine phytoplankton. The average silica (SiO_2) concentrations in Texas bays—4 to 10 mg/l—suggest a plentiful supply of this nutrient. Since silicon concentrations do not seem to control phytoplankton production in these bays (Whitledge 1989), silicon supply rates to the estuaries were not calculated.

In this section, the rates at which nutrients are delivered to five Texas estuaries (nutrient loading rates) are described for carbon, nitrogen, and phosphorus. In addition, the relative importance of different nutrient sources is discussed for each of the bays.

Background

Nutrients brought into an estuary originate from such sources as the runoff from agricultural fields and forests,

Table 4.3.2. Mass loading of major nutrients into Texas estuaries per unit area and volume. Area loading in gm m^{-2} yr^{-1}; volume loading in gm m^{-3} yr^{-1}.

Estuary	Total nitrogen	Total phosphorus	Total organic carbon
Area loading			
Nueces	3.75	0.77	11
Mission-Aransas	2.7	0.56	17
Guadalupe	15.9	2.96	50
Lavaca-Colorado	7.45	1.01	41
Trinity-San Jacinto	29.7	6.66	141
Volume loading			
Nueces	2.10	0.43	6.3
Mission-Aransas	1.93	0.40	12.0
Guadalupe	10.8	2.25	34.2
Lavaca-Colorado	3.18	0.48	19.6
Trinity-San Jacinto	18.6	4.17	88.1

Table 4.3.3. Nitrogen loading to Texas estuaries in relation to residence time.

Estuary	Residence time[a] (yr)	Nitrogen residence-weighted load[b] (million gm/ bay volume)	Per volume nitrogen load ($\text{gm m}^{-3}\text{ yr}^{-1}$)	Residence-weighted volumetric load (gm/m^{3})
Nueces	0.46	1,060	2.10	0.97
Mission-Aransas	3.02	4,900	1.93	5.83
Guadalupe	0.19	1,690	10.80	2.09
Lavaca-Colorado	0.21	1,430	3.18	0.66
Trinity-San Jacinto	0.11	4,900	18.60	2.05

[a] Residence time is the inverse of the flushing rate per year based on tidal exchange as well as freshwater inflow (Armstrong 1982).

[b] Nitrogen (million gm) input per year times residence time, the amount of nitrogen input associated with each water volume which replaces the estuary volume.

leaching of soils, flows from deep or shallow springs, direct precipitation, discharged cooling waters from chemical and electric power plants, industrial process wastes, and municipal wastes. Many of the inputs are modified in transit in rivers or during flow through marshy river deltas. Much of the nutrient load reaches the estuary in dissolved form; the rest arrives already incorporated into organic particles or adsorbed onto clay particles. Since most nutrients—dissolved, particulate, living, or dead—that enter the estuary become incorporated into the active system, the dissolved and particulate fractions are considered together here as a total quantity. Therefore, the discussion below emphasizes total nitrogen, total phosphorus, and total organic carbon. Inorganic carbon also contributes to aquatic production, but is closely linked to the interaction of dissolved and atmospheric carbon dioxide. The amount of carbon delivered to the estuary that is already incorporated into organic material—as bits of leaves, for example—determines the heterotrophic metabolism of the estuary.

Data and Assumptions

Monitoring data. The quantity of material brought into an estuary is the product of the volumes of inflows from various sources and the concentrations of materials in those inflows. Inflow volumes are those summarized in Section 4.1, summing for each estuary the contributions of many contributing watersheds. Nutrient concentrations from the rivers and many streams are measured and reported through monitoring programs of the TNRCC and the USGS. Data from these two sources were combined and monthly average concentrations calculated for contributing rivers and streams with sufficient representation. For many smaller streams and watersheds, there is no data on inflow concentrations. Data from the nearest monitored stream were substituted for these missing concentrations. Similarity of stream size and land drained were also considered in determining the substitute.

Table 4.3.4. Total nitrogen loading components. Units are million gm/yr.

Estuary	Combined load[a]	River[b]	Return flows[c]	Return/ combined	River/ combined	Rain[d]	Total loading[e]	Rain/ total
Nueces	1,960	760	750	0.43	0.34	340	2,300	0.16
Mission-Aransas	1,240	290	40	0.05	0.24	390	1,600	0.28
Guadalupe	8,680	6,940	1,170	0.13	0.80	230	8,900	0.03
Lavaca-Colorado	6,450	3,280	370	0.07	0.49	370	6,800	0.06
Trinity-San Jacinto	44,300	23,170	6,520	0.16	0.51	490	44,800	0.01

[a] Combined load includes gaged, modeled, and return flow inputs.

[b] River input includes major rivers contributing to the estuary, not all gaged streams.

[c] Return flows included are only those below stream gages or in ungaged watersheds.

[d] Rain inputs are from precipitation directly on water surface.

[e] Total = surface + rain inputs.

Rainfall. The concentration of nitrogen in rainwater is one of the quantities monitored by the National Atmospheric Deposition Program, which maintains sampling sites at Victoria, Beeville, and the Attwater National Wildlife Refuge. Nitrogen inputs from rain were based on Thiessen network estimates of precipitation volume (the quantity falling directly on the bay surface) and on the combined concentrations of nitrate and ammonia nitrogen, averaged over several years from the nearest station.

Return flows. Return flows include volumes from many different sources with widely variant concentration profiles. Concentrations reported as part of waste discharge permit compliance often do not cover all species of dissolved nutrients that can be taken up by aquatic organisms. Staff members of the TNRCC have sampled many wastewater outfalls as part of special studies to determine the capacity of rivers and streams to process these wastes. For the purposes of nutrient loading calculations reported here, the results of these studies were used to derive average return flow nutrient concentrations. Concentrations reported for all outfalls within the first few tiers of counties bordering the coast were combined into average concentrations. A division was made between the upper coast and lower coast (including the Guadalupe Estuary drainage in the lower coast) to account for differing proportions of storm water contributions.

Nutrient Loads

Average loads. Average annual loadings of nitrogen, phosphorus, and organic carbon to Texas estuaries are summarized in Table 4.3.1. These values show a large range from one end of the coast to the other, generally following the pattern of inflow volumes. The table includes all terrestrial sources and rainfall contributions, but does not include an estimate of inputs from tidal exchange with the Gulf. By incorporating other information about the estuaries, it is possible to assess the real significance of these numbers to the ecosystems involved.

Loading by area and by volume. Texas estuaries differ greatly in size, and Table 4.3.2 presents the nutrient loading data in the context of their areas and volumes. These converted loading rates are similar to those reported by Armstrong (1982), although the numbers were compiled from more recent data and were based on independent assumptions about particular components such as rain and return flows. Using this per-unit area or volumetric calculation of nutrient inputs, Texas estuaries can be compared realistically to other major estuaries of the Gulf, Atlantic, and Pacific coasts. Based on data presented in Nixon and Pilson (1983), the Nueces and Mission-Aransas estuaries would be among the least loaded estuaries, while the others rank among major estuaries which receive nutrient inputs from large metropolitan areas. Comparison of estuary nutrient loading by volume and area also puts the wide gap between the loading to the Mission-Aransas and Trinity-San Jacinto estuaries into perspective. Though the absolute loadings for nitrogen range from 1 billion to 40 billion gm/yr, the per-unit-volume range is 2 to 20 gm m^{-3} yr^{-1}. Total phosphorus and carbon also show similar loading differences between bays.

Loading and residence time. The fresh water that flows into the estuaries mixes with estuarine water and becomes a flow of mixed estuarine water to the Gulf. As discussed in Section 4.1, inflows can be represented as flushing the estuary at a certain rate, or conversely, of having a residence time. Clearly, the influence of nutrients delivered to the estuary will depend on the rate at which water in the estuary is replaced. This is actually not a simple physical relationship, because there are other physical and biological mechanisms that can trap entering nutrients. However, the water retention rate can be considered when judging the relative magnitude of inputs between estuaries.

Residence times based on freshwater inflow volumes are given in Section 4.1. Armstrong (1982) provides estimates of water residence times for Texas estuaries based on the combined influence of freshwater and tidal flushing. These are shown in Table 4.3.3 with the resulting yearly average residence-weighted loading rate for nitrogen. This residence loading can be considered the average amount of nitrogen associated with each inflowing water volume which replaces the bay volume. Taking residence times of the water and nitrogen loads into account, the bays do not appear very different in the effective nitrogen loads. The long residence time of the Mission-Aransas Estuary (chiefly Copano Bay) brings its nutrient richness to a level comparable to that of the Trinity-San Jacinto Estuary. This is also shown in the comparable bay nitrogen concentrations in figures 4.2.1 and 4.2.2. The last column in Table 4.3.3 incorporates consideration of the per volume loading with residence time weighting. Valiela and Costa (1988) compiled similar volume and residence-weighted loading rates for nitrogen inputs to 23 estuaries and embayments of the east, west, and Gulf coasts. When compared in this manner, Texas estuaries rank among the most heavily loaded. However, the results Valiela and Costa cited may not have included tidal and organic nitrogen sources; thus, the real ranking of Texas estuaries is uncertain.

Components of Nitrogen Input to Texas Estuaries

Importance of direct precipitation. The input of nitrogen from direct precipitation is similar for all bays (Table 4.3.4). This is partly a reflection of slightly higher nitrogen concentrations in rainwater measured at stations representing the lower estuaries. This also shows the relative importance of summer showers which occur just inland of the Gulf shore. Rain is an important source of nitrogen to the estuaries, particularly those of the lower coast. As a percentage of total nitrogen inputs, rain contributes 16% and 28% to the Nueces and Mission-Aransas estuaries, respectively.

Importance of major rivers. The river/combined column in Table 4.3.4 summarizes the ratio of the nitrogen load brought in by major rivers to the total surface water loadings. Among Texas estuaries, there is an apparent trend in the increasing importance of loadings from ungaged coastal watersheds and associated return flows with decreasing rainfall on the estuaries' entire drainage basins. There is a 43% river contribution to the Trinity-San Jacinto Estuary versus 34 and 24% river contributions to the Nueces and Mission-Aransas estuaries, respectively. The Guadalupe Estuary, with 80% loading from its rivers, is somewhat anomalous, since its principal bay is dominated by the inflow of the Guadalupe and San Antonio rivers and because its coastal drainages are relatively small. The relative importance of river-borne nitrogen to other sources contributing to the Lavaca-Colorado Estuary is dependent in part on an estimate of the portion of Colorado River flow which entered the estuary over the period this data represents. This proportion will increase with recent modifications to the Colorado River mouth. Other rivers included in calculating the proportion of river flow for this estuary were the Lavaca and Navidad rivers. Nitrogen concentrations for calculation of the Navidad load were taken from a site below Lake Texana.

Only the Trinity and San Jacinto rivers were included in the comparison of river nitrogen input to other sources for the Trinity-San Jacinto Estuary. Other gaged sources to this estuary represent watersheds equivalent in area to ungaged watersheds of the lower coast. These gaged and ungaged coastal drainages are very important in contributing nitrogen to the Trinity-San Jacinto Estuary. Tributaries draining the metropolitan areas had the highest average nitrogen concentrations encountered in this coastal watershed survey, with average total nitrogen ranging from 3 to 8 mg/l.

Importance of return flows. For this report, return flows were tabulated as inflow sources only if they were not already included as flows measured at some gage; consequently, the return flow category does not account for the entire contribution of wastewater to the nutrient metabolism of the estuaries. The compilation did not include an estimate of leaching from septic systems, which could be important to some secondary bays. However, the category does represent the influence of metropolitan areas and industries bordering and adjacent to the estuaries. Estimates of the relative contributions of nitrogen to the estuaries from wastewaters are summarized as yearly figures in Table 4.3.4. There are several significant findings, which do not depend on the specific values assumed to represent wastewater concentrations. Nueces Estuary receives approximately 40% of its terrestrial nitrogen from municipal and industrial sources. Industrial outfalls contribute significantly to the Guadalupe Estuary nitrogen load. Finally, the return flows to the Trinity-San Jacinto Estuary provide as much or more nitrogen as the total nitrogen inputs to each of the other estuaries.

Discussion of Loading Patterns

Comparative analysis of nutrient loading rates draws attention to the importance of particular nutrient sources and the need for accurate data about those sources. For example, understanding the nitrogen dynamics in the Nueces Estuary requires good information on contributions from municipal and industrial discharges. Conversely, non-point source loading to the Mission-Aransas and Lavaca-Colorado estuaries is an important component of their nitrogen loads.

The proportional balance of nutrients entering the estuary determines the phytoplankton productivity the estuary can support, and which nutrient exerts most control. From per-volume loading rates and the atomic weights of the principal nutrient elements, the ratio of carbon, nitrogen, and phosphorus inputs can be calculated and compared to a conceptual ratio considered balanced for phytoplankton growth (Redfield et al. 1963). For Texas estuaries, the long-term average C:N:P input ratio ranged from 38:11:1 in the Nueces Estuary to 96:13:1 in the Lavaca-Colorado Estuary. The ratio in all these estuaries indicates more than adequate nitrogen and phosphorus to metabolize input carbon, and more than enough phosphorus to support the potential phytoplankton growth at incoming nitrogen concentrations (Webb 1981). Compared with data compiled by Nixon et al. (1986), the ratio of phosphorus to nitrogen inputs calculated for Texas estuaries can be considered typical of other estuaries that have received study.

Some estuaries in other locations with nutrient inputs similar to those of the Guadalupe and Trinity-San Jacinto estuaries have developed symptoms associated with eutrophication. These symptoms can include zones of low oxygen concentrations harmful to fisheries, fish kills, and blooms of noxious algae. Texas estuaries have not shown these symptoms, probably due to the high turbidity of bay waters that limits productivity of algae in the bays most heavily loaded.

Pomeroy et al. (1972) discussed the contribution of nitrogen, phosphorus, and carbon loading to the development of eutrophication in estuaries. They concluded that in turbid estuaries, increased loadings of nitrogen and phosphorus were less likely to be damaging than equivalent loadings in clear estuaries. The loading of organic matter into clear estuaries produces a eutrophication succession in the biota. In turbid estuaries, this same loading would be more likely to produce anoxic zones in the bays most heavily affected. In Texas, the clearer estuaries of the lower coast receive a high nitrogen and phosphorus input with respect to inputs of carbon. Therefore, typical symptoms of eutrophication could appear sooner in these estuaries, although they receive less total nutrient loading than the more turbid upper coast estuaries. Current monitoring programs in Texas are designed to detect nutrient overloading, whether manifested as problems with low dissolved oxygen or as a trend of increasing phytoplankton and dissolved nutrients.

4.4 INFLUENCE OF INFLOW ON SEDIMENT DEPOSITION IN DELTA AND BAY SYSTEMS

Introduction

Water is the agent primarily responsible for loosening and transporting soil and rock particles to the sea. Soil and rock erosion by water can occur through one of the following processes (Hudson 1971): (1) splash erosion, which occurs when raindrops break down soil aggregates that are then removed by the flow; (2) sheet erosion, which occurs when sheets of water move over a broad area and remove loose soil particles; (3) rill erosion, which occurs when water moves in a more concentrated area, such as in small grooves on the soil surface, and removes soil particles from the grooves' perimeter; and (4) gully erosion, which occurs when water moves over an area with enough energy to cause erosion and create a channel.

Erosion rates depend on the erosion process, soil type, topography, land use, and cover type, and can vary by orders of magnitude. For example, erosion rates in similar soils measured in a depleted hardwood forest, an abandoned field, a pasture, and on cultivated land averaged 264 (0.118), 316 (0.141), 3,609 (1.610), and 53,500 kg/ha/yr (23.87 ton/acre/year), respectively (Schreiber and Duffy 1982). The quantity of sediment eventually reaching a watershed's outlet can be significantly less than that which is eroded due to particle deposition. In some cases, less than 25% of eroded soil reaches the ocean (Vanoni 1975).

Sediment Characteristics

Sediment transported in streams and rivers generally includes non-living organic and inorganic materials such as clay and organic colloids, organo-clay complexes, silt, sand, gravel, twigs, etc. These materials transport nutrients that are crucial to the survival of estuarine ecosystems. In some cases, the nutrients are absorbed or adsorbed to the materials; in other cases, the materials themselves are the nutrients once they are degraded. In addition, the quantity of sediment transported and the particle size distribution of the sediment control several properties of and processes within fluvial systems.

Sediment carried in streams and rivers can be characterized by its organic material content, particle size distribution, and by whether the sediment is suspended or part of the bed load. Suspended load consists of sediment supported by the upward components of turbulent currents and stays in suspension for an appreciable length of time, whereas bed load sediment is primarily confined to the region just above the bed and moves by jumping, rolling, or sliding (Simons and Senturk 1976). Bed load is not easily captured because of the above characteristics, and is therefore not commonly measured. As a result, bed load is generally estimated simply by taking a percentage of the suspended sediment load, where the percentage is a function of channel characteristics (Lane and Borland 1971). Sediment load data presented later in this section that is based on TWDB and USGS measurements do not include bed load.

Chemical composition. The total organic content in suspended sediment is generally estimated by measuring the fraction of volatile solids in a sediment sample or, using more sophisticated techniques, by detailed chemical analysis for organic carbon. The ratio of volatile solids to total solids measured in Texas rivers by TNRCC staff during the period 1969 to 1989 is presented in Figure 4.4.1. River-to-river variation and within-river variation in the volatile solid to total suspended solid ratio ranges roughly from 0.20 to 0.55. No consistent trends are evident in this data. In some cases, the volatile solids fraction increases in the downstream direction toward the coast (Trinity River), while in other cases, it decreases (Sabine River, Arroyo Colorado). At most stations, the maximum measured volatile-solid to total suspended solid ratio was 100%, indicating that, at times, most sites are saturated with organic material.

Particle size distribution. Sediment is classified according to its particle size as either clay, silt, sand, or gravel. Several common schemes are used to characterize particle size distributions. Geologists most often use the Wentworth scheme; engineers generally use the AASHTO scheme; and agricultural scientists generally use a third scheme devised by the SCS. The particle size distributions used in these three schemes are significantly different (Table 4.4.1) in determining silt and clay content, but, in general, are similar in determining sand content.

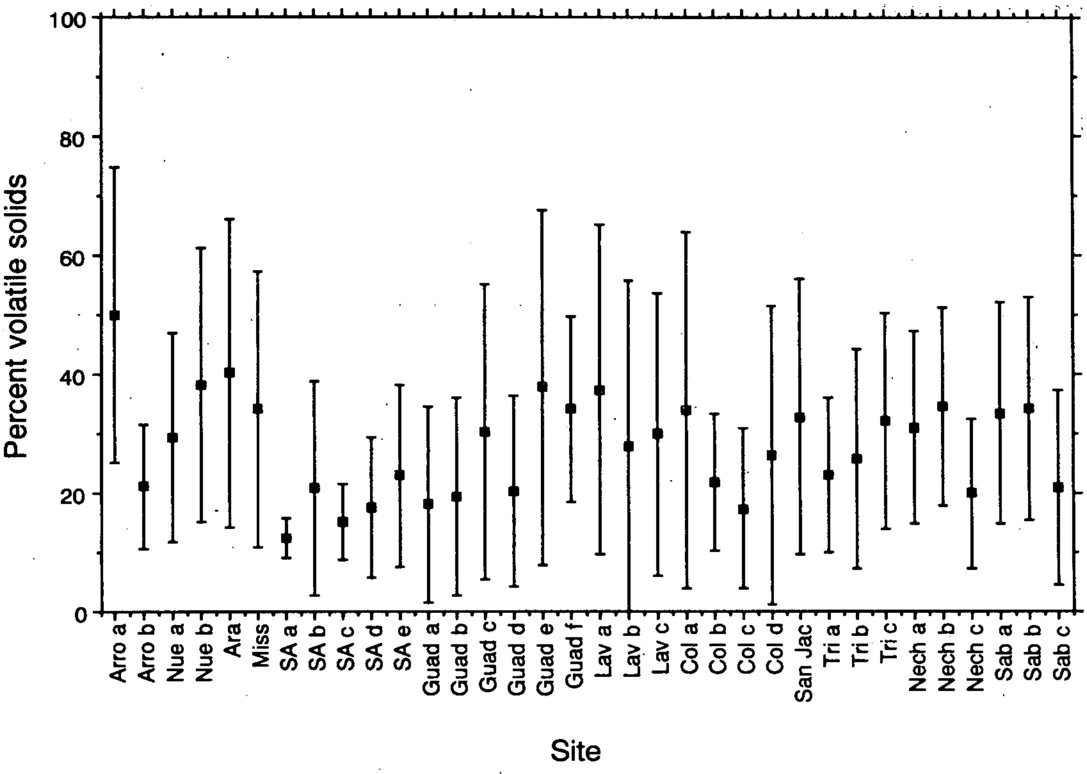


Figure 4.4.1. Mean and one-standard-deviation error bars for volatile solids measured near mouths of Texas rivers. Sites identified by TNRCC Statewide Monitoring Network numbers below.

River	Designation identifier	TNRCC-SMN
Arroyo Colorado	Arro a	22010100
	Arro b	22020200
Nueces River	Nue a	21010100
	Nue b	21020100
Aransas River	Ara	20040100
Mission River	Miss	20020100
San Antonio River	SA a	19010050
	SA b	19010100
	SA c	19010105
	SA d	19010130
	SA e	19010145
Guadalupe River	Guad a	18010100
	Guad b	18030025
	Guad c	18030100
	Guad d	18030150
	Guad e	18030200
	Guad f	18030220
Lavaca River	Lav a	16010100
	Lav b	16020100
	Lav c	16020180
Colorado River	Col a	14010100
	Col b	14020035
	Col c	14020050
	Col d	14020300
San Jacinto River	San Jac	10010100
Trinity River	Tri a	8010100
	Tri b	8020100
	Tri c	8020180
Neches River	Nech a	6010100
	Nech b	6010300
	Nech c	6020100
Sabine River	Sab a	5010100
	Sab b	5010300
	Sab c	5030100

Table 4.4.1. Sediment particle sizes (mm) under various classification schemes.

Type	Classification Scheme		
	Wentworth	AASHTO	USDA
Gravel	> 2.0	> 2.0	> 2.0
Sand	0.06 - 2.0	0.05 - 2.0	0.04 - 2.0
Silt	0.002 - 0.06	0.002 - 0.05	0.002 - 0.04
Clay	< 0.002	< 0.002	< 0.002

The breakdown of total suspended load into sand, silt, and clay fractions, using the AASHTO scheme, is presented in Figure 4.4.2 for several major rivers in Texas (Welborn 1967). The data indicate a decrease in sand content and a corresponding increase in clay content moving from the Louisiana border to the Mexican border of Texas. Although this trend reflects the variation in soil type and characteristics across the state, some of the measurements were taken significantly upstream from the coast. Thus, the particle size distributions one would find at the coast, affected by particle deposition in reservoirs and elsewhere throughout transit, might be somewhat different than those shown in Figure 4.4.2.

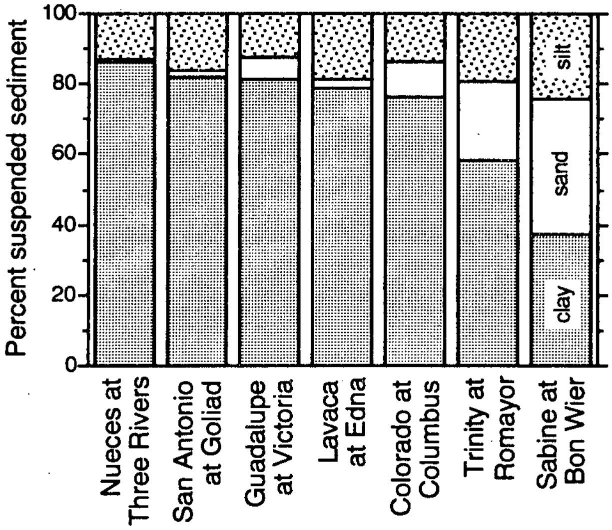


Figure 4.4.2. Fraction of sand, clay, and silt in suspended load carried by Texas rivers. Data from Welborn (1967).

Sediment Loading

Sediment loading in rivers can be expressed in terms of the absolute loading over time (tons or kg of sediment/yr), in terms of long-term sediment concentration (tons of sediment/acre-ft of river discharge), in terms of the sediment yield factor (tons of sediment/acre of drainage area/yr), or in terms of shorter-term sediment discharge relationships (tons/month of sediment input versus acre-ft/month of river discharge). Sediment load can be measured directly at the watershed outlet by analyzing water samples for suspended sediment, or can be estimated indirectly by other means. Estimates of sediment loading by rivers do not generally include bed load.

Measurement techniques. Sediment load can be measured directly by analyzing samples collected at a stream gage site. The TWDB and the USGS, both of whose data are used in this section, operate a network of collection sites for measuring suspended sediment loads in Texas rivers.

The TWDB and the USGS use different devices for measuring sediment load. The difference between results obtained with the Texas sampler, the device used by the TWDB, and the depth integrating sampler, used by the USGS, were investigated by Welborn (1967) and Andrew (1989). Over a 14-year study period (1966 to 1979), the USGS estimated a suspended sediment load 3.4% greater than the TWDB, with individual year differences ranging from 6.9% less to 15.3% more than the TWDB's measurements (Table 4.4.2). Caution should be exercised when making direct comparisons between TWDB and USGS data because of small differences that can arise strictly due to differences in the measurement techniques.

In the above studies, all samples were collected at equal frequency, although the USGS generally collected samples at irregular intervals while the TWDB collected samples every day.

Table 4.4.2. Comparison between TWDB and USGS methodologies for measuring suspended sediment (Andrews 1989).

Year	TWDB[a] (ton)	USGS[a] (ton)	Difference (ton)	Difference (%)
1966	18,484,000	19,899,594	1,415,594	7.7
1967	982,600	1,071,952	89,352	9.1
1968	29,618,000	30,799,205	1,181,205	4.0
1969	14,341,000	15,635,604	1,294,604	9.0
1970	8,705,000	9,202,515	497,515	5.7
1971	1,044,000	1,172,919	128,919	12.3
1972	3,919,000	3,943,243	24,243	0.6
1973	11,800,000	12,140,975	340,975	2.9
1974	11,400,000	10,813,593	-586,407	-5.1
1975	23,055,000	25,146,463	2,091,463	9.1
1976	8,585,000	9,154,709	569,709	6.1
1977	17,211,000	16,023,057	-1,187,943	-6.9
1978	993,900	1,146,300	152,400	15.3
1979	23,243,000	23,045,200	-197,800	-0.9
Total	173,381,500	179,195,329	5,813,829	3.4
Mean	12,384,393	12,799,666	415,274	3.4

[a] Data from Richmond, Texas, on the Brazos River.

Sediment discharge relationships. Searey and Hardison (1960) suggest the use of a double-mass curve, where cumulative sediment input is plotted against cumulative river discharge, to identify the occurrence of changes in the sediment-discharge relationship. Changes in the sediment-discharge relationship can occur due to man-made influences such as reservoir construction upstream of the gage site, or due to natural influences such as a major loss of topsoil due to extreme erosion. Double-mass curves for nine Texas rivers are presented in Figure 4.4.3; the slopes of the double-mass curves between break points, which give long-term average sediment concentrations, are presented in Table 4.4.3. These figures are based on data taken by both the TWDB and the USGS. Caution should be exercised in interpreting the USGS data since these data were taken at irregular intervals. For example, the total cumulative sediment input (and total cumulative discharge) indicated on the figures based on USGS data are incorrect since several months of data are missing. Nonetheless, the slopes of the double-mass curves provide reasonable estimates for long-term average sediment concentrations.

Breaks in the slope of the double-mass curves occur in several instances following the construction of reservoirs. However, breaks in the slopes which are not associated with reservoir construction also occur. For example, the double-mass curve for the Trinity River indicates two significant breaks—the first occurring in 1947 and the second in 1968. No explanation is evident for the 1947 break. The second break appears after construction of Lake Livingston in October 1968, reducing the sediment input rate from 0.732 tons/acre-ft (0.538 kg/m^3) to 0.152 tons/acre-ft (0.112 kg/m^3).

Breaks in the double-mass curve for the Lavaca River cannot be explained by reservoir construction, as no reservoirs have been built on the Lavaca River. Breaks for the Lavaca, Guadalupe, and San Antonio rivers appear to be associated with large inflow events following heavy precipitation, suggesting that large amounts of sediment are possibly flushed from the system during floods, leaving little behind and eventu-

Table 4.4.3. Slope of double-mass curves relating suspended sediment input to discharge.

River	Date	Average sediment concentration[a] (ton/acre-ft)	Sediment load (10^6 ton/year)	Sediment yield factor(10^6 ton /acre/year)	Comments
Sabine at Ruliff	1974-1987	0.057	—	—	USGS data
Neches at Evadale	1960-1987	0.690	—	—	USGS data
Trinity at Romayor	1936-1946	1.076	7.523	0.684	
	1947-1968	0.732	3.081	0.280	
	1969-1986	0.152	0.800	0.073	Following construction of Lake Livingston
Colorado at Wharton	1977-1987	0.327	—	—	USGS data
Lavaca at Edna	1946-1957	1.170	0.160	0.305	
	1958-1986	0.492	0.147	0.282	Following 1950's drought
Guadalupe at Victoria	1946-1957	0.593	0.516	0.155	
	1958-1986	0.359	0.513	0.154	Following 1950's drought
San Antonio at Goliad	1945-1957	1.427	0.544	0.217	
	1958-1986	0.728	0.440	0.175	Following 1950's drought
Mission at Refugio	1979-1987	0.116	—	—	USGS data
Nueces at Mathis	1961-1971	0.094	0.575	0.054	
	1972-1986	0.034	0.156	0.015	

[a] Average concentrations based on slope (from least-squares best-fit line) of double-mass curve for period indicated.

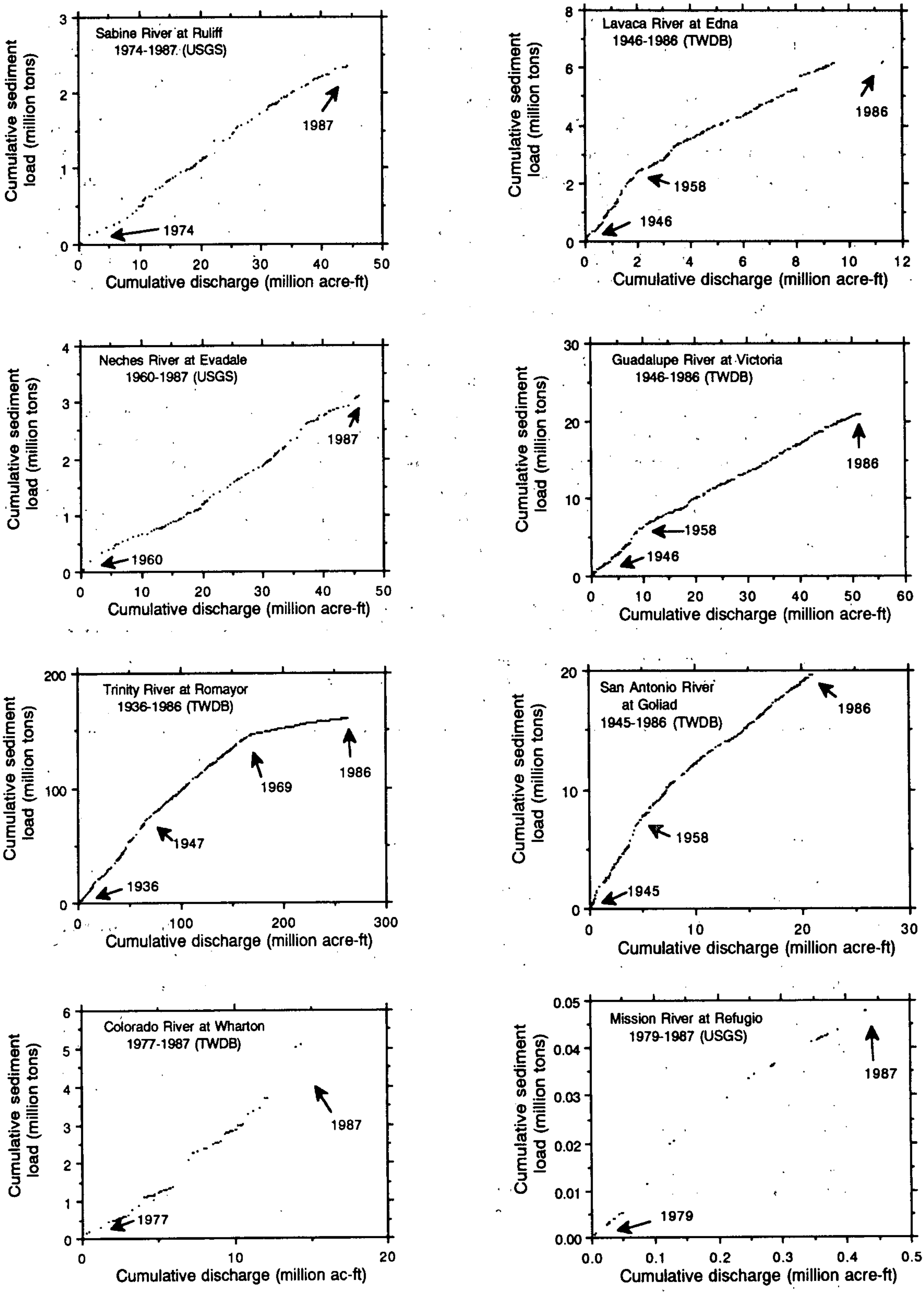


Figure 4.4.3. Double-mass curves—cumulative sediment load versus cumulative river discharge—in Texas rivers. Slope between break points (average suspended sediment concentration) provided in Table 4.4.4.

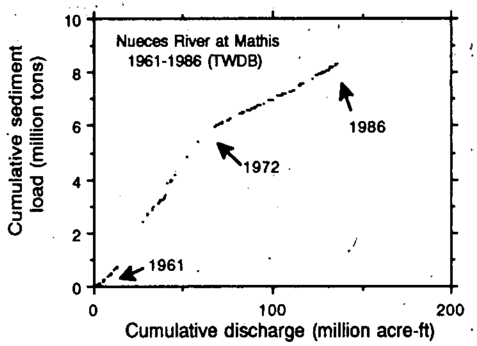


Figure 4.4.3. Continued from previous page.

ally reducing the sediment-discharge slope. There is no explanation for the break in the double mass curve for the Nueces River in 1972.

The long-term average sediment concentrations, given by the slope of the double-mass curve, decrease for all TWDB stations. The largest decline occurred in the Trinity River where concentrations fell from 1.076 tons/acre-ft (0.791 kg/m^3, 1936-1946) to only 0.152 tons/acre-ft (0.112 kg/m^3, 1969-1986) (Table 4.4.4). The highest average sediment concentrations currently occur in the San Antonio River (0.728 tons/acre-ft, 0.535 kg/m^3), while the lowest are in the Nueces River (0.034 tons/acre-ft, 0.025 kg/m^3), a difference by more than a factor of 20. It should be noted that Nueces River sediment concentrations are extremely low due to measurements being made just downstream of Lake Corpus Christi, where significant particle deposition takes place.

Absolute sediment load in terms of tons of sediment per year have also generally declined. (Data are presented for TWDB stations only.) In the Trinity River, the absolute sediment load decreased from 7.52 million tons/year (6,820 million kg/year, 1936-1946) to only 0.80 million tons/year (7.26 million kg/year, 1969-1986) (Table 4.4.4). Despite the decrease, the Trinity River still delivers the largest absolute sediment load, while the Lavaca River delivers the smallest (0.147 million tons/year, 133 million kg/year). Sediment yield factors for TWDB sites vary from 0.282 million ton acre^{-1}yr^{-1} (632 million kg ha^{-1}yr^{-1}, Lavaca River at Edna) to 0.015 million ton acre^{-1}yr^{-1} (33.6 million kg ha^{-1}yr^{-1}, Nueces River at Mathis).

Short-term sediment-discharge relationships of the form $S = aQ^b$, where S is sediment input in tons/month, Q is river discharge in acre-ft/month, and a and b are site-dependent constants, were computed for the TWDB data sets for each of the periods between break points on their respective double-mass curves. Similar relationships were also computed for USGS data sets using all available data for each river. Monthly values for USGS data were obtained by assuming that the instantaneous values measured (in mg/liter) were applicable through the entire month, and were appropriately converted to a monthly value (in tons/month). Results are presented in Table 4.4.4 and in Figure 4.4.4. These equations can be used to estimate monthly sediment input given monthly river discharge.

Estimates of sediment yield in ungaged areas. The technique used to quantify sediment yield in ungaged watersheds, which are generally located closest to the river mouth where streamflow gages are tidally affected, is based on erosion calculated by the universal soil loss equation and on gully and streambank erosion (SCS 1976, 1977, 1978, 1979). Based on the above technique, Greiner (1982)

Table 4.4.4. Coefficients a and b for monthly suspended sediment-monthly discharge relationship for Texas rivers ($y = aQ^b$ where y = monthly suspended sediment load [tons/month], Q = monthly discharge [acre-ft/month], n=number of data points, and R^2 = correlation coefficient).

River	Period	a	b	R^2	n	Data source
Sabine, Ruliff	1974-1987	$3.46*10^{-3}$	1.19	0.68	101	USGS
Neches, Evadale	1960-1987	$5.80*10^{-2}$	1.00	0.66	113	USGS
Trinity, Romayor	1936-1946	$2.22*10^{-3}$	1.46	0.91	125	TWDB
	1947-1968	$3.97*10^{-4}$	1.56	0.90	264	TWDB
	1969-1986	$3.50*10^{-4}$	1.43	0.88	212	TWDB
Colorado, Wharton	1977-1987	$8.56*10^{-6}$	1.82	0.80	85	USGS
Lavaca, Edna	1946-1957	$1.27*10^{-2}$	1.42	0.86	148	TWDB
	1958-1986	$1.68*10^{-2}$	1.32	0.83	344	TWDB
Guadalupe, Victoria	1946-1957	$1.24*10^{-4}$	1.70	0.86	144	TWDB
	1958-1986	$1.01*10^{-4}$	1.65	0.81	348	TWDB
San Antonio, Goliad	1945-1957	$9.17*10^{-5}$	1.89	0.83	156	TWDB
	1958-1986	$6.51*10^{-4}$	1.62	0.71	348	TWDB
Mission, Refugio	1979-1987	$6.67*10^{-2}$	1.04	0.91	43	USGS
Nueces, Mathis	1961-1971	$6.71*10^{-3}$	1.21	0.89	126	TWDB
	1972-1986	$2.34*10^{-2}$	1.02	0.90	177	TWDB

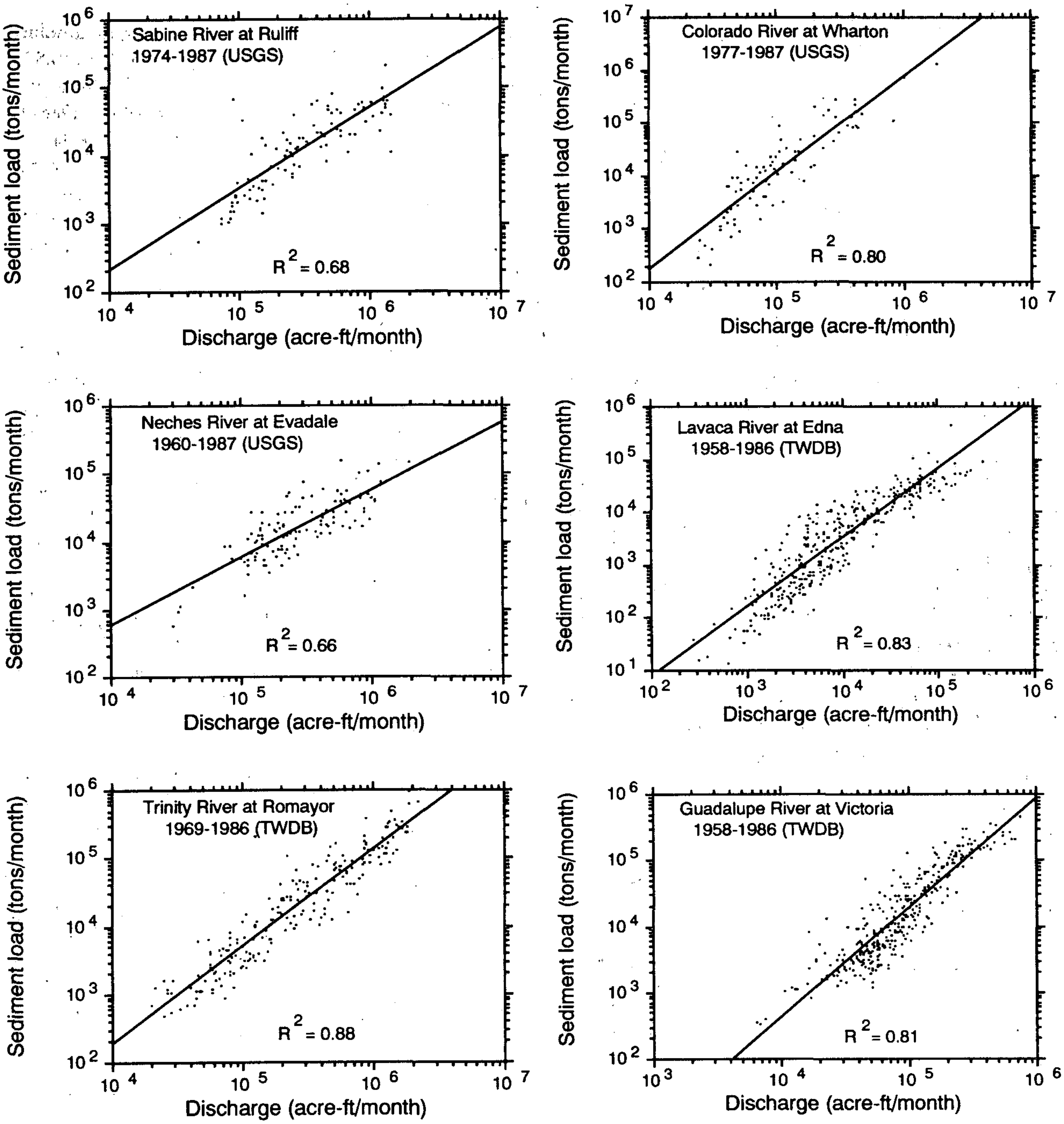


Figure 4.4.4. Monthly sediment load versus monthly river discharge for Texas rivers.

calculated sediment yield factors for 300 points in Texas in 1979. These analyses are of limited regional application, and the derived relationships show a considerable scatter of data points because estimated or measured rates of annual sediment production vary widely (Petts and Foster 1985). Sediment yield factors were found to vary widely across Texas (0.13 to 2.33 ton/acre, 291.36 to 5,222 kg/ha) reflecting the wide range of soils, land type, and land management practices across the state. Sediment yield factors and sediment load estimates are provided for ungaged coastal basins on the Texas coast in Table 4.4.5.

Delta Development

Rivers carry sediment to the bays. When a river flows into a bay, the velocity of the water suddenly decreases. Some sediment held in suspension due to the water's movement settles to the bottom. If local waves and currents are not strong enough to carry the sediment away from its site of deposition, the sediment will eventually build up and create a delta in what was formerly open water.

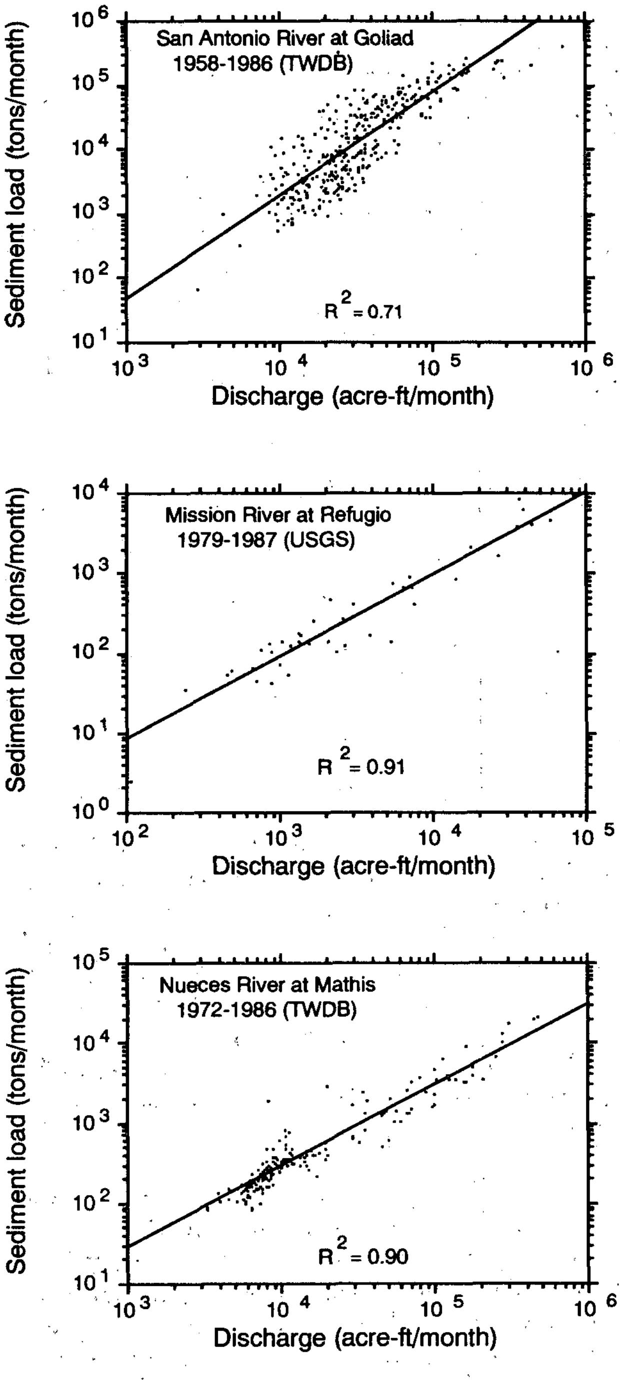


Figure 4.4.4. Continued from previous page.

Deltas and delta habitats. Deltas form seaward of the river mouth and consist of the subaqueous sediment deposited in the bay near the river mouth, the river channels and levees that contain the river flow, and the shallow areas that occur on the opposite sides of the levees from the river channels. The river channels and levees are called distributaries, while the shallow areas between them are called interdistributaries. It is the interdistributary areas and shallow edges of the delta that develop habitats attractive to estuarine fish and shellfish. In these areas, marsh plants and occasional seagrasses may grow because the water is shallow and protected from excessive wave action, and the sediment contains abundant inorganic nutrients. These shallow, vegetated areas provide significant habitat for larval and juvenile forms.

Fisher (1969) identified the major factors affecting delta development. From the landward perspective, factors include sediment type, amount, delivery rate, variation in delivery rate, and relative amount of bedload. These variables depend on river basin characteristics and may be influenced by events that occur hundreds of kilometers inland of the delta. Other factors affecting delta development are more local and include: the nature of the bay-bottom sediments; the depth of the water in which the delta builds; the energy environment of the bay where deposition occurs including waves, currents, and tides; and the structural characteristics of the underlying bay-bottom substrate. Physical and chemical properties of the water may influence the transport and deposition of sediment. For example, water density differences between river and bay waters may minimize the mixing of water masses, influencing the distribution of some suspended sediment. Alternatively, chemical processes resulting from the mixing of fresh and salt water may cause flocculation and settling of clay particles from the river water.

A delta consists of subdelta units that undergo a regular process of growth and decay. The effect of some factors listed above on the subdelta cycle is discussed in more detail below.

Subdelta construction. Consider a newly opened river channel to a bay. Water carrying suspended sediment and some bed load passes through the river mouth and into the bay. As water enters the bay, the levee banks cease confining the flow and channeling it in a single direction; the water can spread out in a wide horizontal plane. Since the river water reaches the base water level of the bay, the force of gravity no longer drives the river water flow. The kinetic energy of the river water encounters the mass of the stationary or slowly moving bay water and dissipates in turbulence as the river water begins to mix.

The kinetic energy of the river water keeps sediment suspended in the water column while the river is flowing between its banks. With the loss of kinetic energy, suspended sediment begins to settle to the bottom. The ability of flowing water to keep various sizes of sediment particles in suspension is proportional to the water velocity: big particles require greater water velocities, smaller particles require lesser velocities. When the velocity of the water decreases, the heaviest particles, such as sands or fine sands, fall out

Table 4.4.5. Sediment yield in 1979 for ungaged coastal river basins (Griener 1982).

River	Sediment yield (ton acre^{-1} yr^{-1})	Area (acre)	Sediment mass (ton)	Sediment volume (acre-ft)
Trinity below Romayor	0.16	505,103	80,816.5	53.0
San Jacinto below Lake Houston and Buffalo Bayou below Addicks	0.64	526,277	336,817.3	220.9
Colorado below Columbus	1.36	461,481	627,614.2	411.7
Navidad above Lake Texana	1.12	895,620	1,003,094.4	657.9
Lavaca (all)	0.66	581,492	383,784.7	251.7
Guadalupe below Victoria	0.67	505,856	383,923.5	251.8
San Antonio below Goliad	0.36	168,960	339,749.1	110.8
Mission (all)	0.29	647,154	187,674.6	123.1
Aransas (all)	0.33	524,832	173,194.6	113.6
Nueces below Lake Corpus Christi	1.51	164,925	249,036.8	163.3
Oso (all)	0.51	177,141	88,570.5	58.1

first. With further velocity decreases, smaller-sized particles begin to drop out in order: silts, fine silts, and then large clay particles.

As the river water enters the bay, it slows immediately. It spreads out from the river mouth and drops sand, fine sand, and silt laterally. This forms the base of what will become future river banks and levees. The river water also deposits sand and fine sand across the channel opening into the bay creating a bar. As river water travels beyond the bar, slows, and spreads laterally, sandy silt and silty clay settle out, creating a subaqueous fan-shaped area called the delta front. Beyond the delta front, silty clay and clay settle in a prodelta that lays directly upon the original bay bottom. The deposition of clay particles is more complicated than sand and silt. In river water, fine clay particles have large molecular surface areas and adsorb ions so that they become negatively charged. These charged particles repel each other, and the repulsion may be great enough to overcome the force of gravity that would otherwise cause the particles to settle. Therefore, they remain in suspension. When fine clays reach the bay waters, positively charged cations such as sodium—abundant in estuarine waters—neutralize some charged particles, causing them to flocculate and settle to the bottom.

Eventually, the bar across the mouth of the river channel collects enough sediment that the water starts to pass around either side of the bar. The bar becomes a submerged mid-channel shoal, directing the river flow to split in two separate water channels (Russell 1967). During periods of high inflow, enough sediment may be deposited at the mid-channel shoal and along the submerged levee deposits lateral to the main flow direction that the shoals and levees become subaerial as the water level returns to normal. This creates two distinct channels separated by land. In this manner, the mouth of the river begins to advance out beyond the old shoreline into the bay. This slow advancement of the river channel, levees, delta front, and prodelta into the bay is called progradation.

As the subdelta progrades further, the levees isolate the interdistributary area between the river channels from normal river flow and protect the area from currents and wave action. During periods of high inflow, water and sediment enter this area from the flood waters that spill over the levees. As this water slows, sand and silt deposit on or near the levee, while silt and clay are carried into the interdistributary. During normal flow periods, turbid water from the bay may enter the interdistributary area and deposit clays. Measurements from the Colorado River Delta by White and Calnan (1990b) confirm this pattern. Clay constituted up to 80% and silt no more than 30% of the sediment in the low marshes on interdistributary areas, while high marshes on river levees contained more silt (up to 50%) and less clay (no more than 40%).

Even during flooding, the interdistributary area is sheltered from currents and wave action so the silts and clays settle more rapidly than if they were in less protected open water. The interdistributary area acts as a settling tank for sediment. This results in a shallow open water area with fertile submerged soil—the right conditions for marsh plant and submerged seagrass growth. As the plants become established, the stems and blades of the vegetation further promote the deposition of suspended sediment by retarding flow and wave action. Bottom dwelling suspension-feeding invertebrates in the interdistributary may add to the removal of fine particles from the water column and their deposition at the sediment surface (Smith and Frey 1985).

Besides shallowing the interdistributary area, the sediment carried by the river supplies material for construction of new subaqueous levees and bars across the mouths of the channels that had divided earlier. When enough sediment has gathered, these channels divide again in the manner described above. This process produces a geometric sequence of channels as the subdelta progrades. How far and how rapidly the subdelta advances into the bay depends on many factors including depth of water, quantity of sediment supplied, pattern of channels, and relative amounts of transported sand, silt, and clay. Sediment from flood waters and organic matter from marsh plant production allows the interdistributary land surface to reach sea level and rise above it. Small ponds and bayward-flowing creeks frequently remain in the central interdistributary area. Since there is an adequate supply of sediment for the levees and interdistributaries, the subdelta will continue to grow and prograde into the bay.

Subdelta deterioration. The rate of vertical accretion (aggradation) of the interdistributary areas varies inversely with the land elevation and distance from the water source for flooding (Redfield 1972). Since levee banks may become high enough to reduce the frequency of flood waters reaching the interdistributary areas, the rate of aggradation eventually declines. The elevation of the interdistributary marshes begins to depend more on the deposition of organic material from the plants than inorganic sediment.

The subdelta has several layers. At the base is the bottom sediment of the bay, followed by the prodelta layer that has silty clay and clay. On top of that is the delta front layer that is sandy silt and silty clay. Higher layers may consist of silty clays in interdistributary areas or sandy silts along levees. The weight of the upper layers compacts the clays and silts of the prodelta and delta front layers. Deeper sediments below the bay bottom may also compact as the result of the weight. The entire upper bay area may be affected by larger scale processes such as subsidence from water and mineral extraction, and the Gulf Coast Geosyncline (a downwarping of the Gulf shoreline from Mexico to Mississippi that has occurred during the past 150 million years). In addition to subsidence, a global (eustatic) rise in sea level independent of land surface changes is also occurring at the present time (Gornitz et al. 1982). The land surface elevation in the subdelta exists in a balance between the forces adding to the elevation and those decreasing it. The subsidence and eustatic sea-level change combine to produce a relative rise in sea level. When the relative rise overbalances the aggradation forces, the delta begins the process of deterioration and abandonment.

Inevitably, the river creates another pathway to the bay that has a greater slope than the present route. Less water flows through the river channels in the recently constructed delta, and they begin to accumulate sediment and shallow. Flood waters pass through the new opening to the bay and it begins to build its own delta. Because the source of inorganic sediment to the recently constructed delta diminishes, the balance between aggradation and relative sea-level rise tips toward land submergence. A sure sign of deterioration is the widening of small ponds and creeks in the interdistributaries and the appearance of new shallow open-water area.

Estimates of sea-level rise during the next century that result from global warming range from 0.15 to 2m (Schmandt et al. 1992). Subdelta deterioration will accelerate even if only modest climate change-related rises in sea level occur.

Subdelta abandonment. Sometime after the river has changed course, the channels in the delta shoal so they cannot deliver sediment to the interdistributaries and levees. Organic material from the marsh plants is the only new particulate matter available to counteract the relative rise of sea level. Organic production may result in a thick layer of organic sediment, but the relative rise in sea-level and wave action cause the open water areas to spread wider. The interdistributary area and the levees eventually sink below the water level and create a new open-water area.

Subdelta cycle. The cycle of subdelta construction, deterioration, and abandonment occurs often during the building of a delta. Small subdeltas are built and abandoned. New subdeltas build next to deteriorating subdeltas or may be constructed on top of old subdeltas that have subsided and become bay bottom. At some point in this repeating process, the rate of subsidence slows. The particulate matter delivered and organic matter produced on site effectively counter the relative sea-level rise, and the land surface remains subaerial. This has occurred in most of the river basins that drain into Texas bays. For example, during the past 4,500 years, the Guadalupe Delta has prograded about 24 km (15 mi) into the bay since the end of the late Wisconsin Glaciation (McGowen et al. 1976b).

A few rivers have carried so much sediment that the estuaries into which they flowed have completely filled. The

Brazos and Colorado rivers both flowed into a common estuary located between what is now East Matagorda Bay and West Bay by Galveston Island. The estuary reached inland 35 km (22 mi) about 4,500 years before present (BP), but has completely filled since then (McGowen et al. 1976a). More than 4,500 years BP, the Rio Grande built a delta in its estuary that extended beyond the present Gulf shoreline. When sea level rose to its present elevation, erosion of sediment from that delta helped to form the modern South Padre Island (Brown et al. 1980).

Assessment of Sediment Input to Deltas and Bays

Nueces Delta. The Nueces Delta prograded into Nueces Bay during the period 1867 to 1982, but recent photographs show that marsh progradation ended between 1930 and 1959 (Morton and Paine 1984). The delta has continued to grow slightly, through accretion into Nueces Bay, but open water and barren flat areas have increased faster than the delta has accreted. White and Calnan (1990b) measured net vegetated area decreases of 54 ha (133 acres) between 1930 and 1959, and 75 ha (185 acres) between 1959 and 1979. The rate of decrease in the latter 20-year period is about double the rate of decrease in the former 29-year period.

From sediment load records, White and Calnan (1990a) showed that the fluvial sediment load below Lake Corpus Christi was much smaller from 1961 to 1980 than from 1942 to 1957. Wesley Seale Dam, which created Lake Corpus Christi (capacity about 280,000 acre-ft), was completed in 1958. White and Calnan (1990b) attributed the reduced fluvial sediment loading to this reservoir. Before Wesley Seale Dam was built, the smaller Corpus Christi Reservoir was impounded behind Mathis Dam built near Wesley Seale Dam. This reservoir, completed in 1934, impounded 54,000 acre-ft, but decreased in capacity to 39,400 acre-ft in 1948 due to rapid sedimentation (Buckner et al. 1986). The end of delta progradation was probably related to the construction of Corpus Christi Reservoir as evidenced by the rapid 30% loss of reservoir capacity in just 14 years.

The Nueces Delta now receives much less sediment than before reservoir construction. More than 95% of the Nueces River Basin that contributes sediment to the delta is upstream of Lake Corpus Christi. In a recent study for the period 1972 to 1985, Leibbrand (1987) showed that 97% of the sediment entering Lake Corpus Christi was retained in it. Still, it is not possible to make an unequivocal statement about the effect of reservoir construction on delta area decreases since the Nueces Bay area has also experienced subsidence (Brown et al. 1974; Ratzlaff 1980), probably due to the Saxet oil and gas field. In addition, the reduction in Nueces Delta vegetated area is the smallest decrease of all the major deltas, and is less than the decrease in the Guadalupe Estuary that has not experienced sediment reductions from nearby reservoir construction.

Nueces and Corpus Christi bays. Shepard (1953) compared the sediment load of the Nueces River with the fill of Nueces and Corpus Christi bays and found a close correlation. However, reservoir construction has decreased the sediment load to the delta and bay to only 4% of Shepard's annual estimate (White and Calnan 1990a). White and Calnan stated that it is possible that sedimentation rates have fallen behind the relative rise in sea level in Corpus Christi Bay and the bay is becoming deeper. Since Nueces Bay and Nueces Delta receive most of the small sediment load delivered by the river, it seems unlikely that much river-borne sediment reaches Corpus Christi Bay.

Guadalupe Delta. The Guadalupe Delta is undergoing a major shift in development. Traylor Cut, dredged in 1935, now carries half or more of the discharge from the Guadalupe River into Mission Lake. A new subdelta is building at Traylor Cut with the prodelta and delta front extending into Mission Lake. The quantity of water and sediment carried through the north and south forks of the river to Guadalupe Bay has decreased because of Traylor Cut. White and Calnan (1990b) measured vegetated area changes using photographs from 1930, 1957, 1974, and 1979. The area next to the north and south forks of the river and Mission Lake has been modified by levees and Traylor Cut. New vegetation in this area has reduced the loss to only 10 ha (25 acres) from 1930 to 1979. The vegetated area west of the Guadalupe River above Hynes Bay decreased by 190 ha (475 acres) from 1930 to 1979.

The lower delta below the south fork of the river consists of several subdeltas whose river channels were abandoned. These consolidated subdeltas are subsiding without a source of sediment; water bodies within this area are widening and deepening, and the edge of the delta by Hynes Bay, San Antonio Bay, and the lower Guadalupe Bay is eroding from wave action. Vegetation coverage in this area decreased by 307 ha (759 acres) from 1930 to 1979 (White and Calnan 1990b).

Information presented earlier in this section showed a trend of decreased sediment loads in the Guadalupe River beginning in 1958. The break in the double-mass curves (Figure 4.4.3) predates construction of Canyon Lake. There are no obvious upstream developments that can account for the change. The portion of the delta next to Mission Lake and the north and south forks of the Guadalupe River is the only part of the delta that regularly receives flood waters and sediment from the river (TDWR 1980b); the delta areas west of the river and below the south fork are effectively cut off from river sediment due to the way the delta has developed. So, changes in the Guadalupe Delta seem to be more attributable to the subdelta construction and decay cycle than changes in sediment load.

San Antonio Bay. About 21% of the sediment load of the Guadalupe River is deposited in Mission Lake or as part of the Traylor Cut delta (Section 7.5). The sediment load of the river is 2% sand, 27% silt, and 71% clay (Welborn 1967). Since sand and silt settle out first, and most of that deposition occurs in the Mission Lake area, most of the sediment carried to the bay is clay and fine clay. Much of this material settles, but some may be carried to the passes and into the Gulf. Based on sediment trap efficiency curves, White and Calnan (1990a) estimated that Texas estuaries (except for Sabine Lake) trap 95% of their sediment load. They also noted that other studies have estimated lower trap efficiencies of 60 to 70%.

Several studies reviewed by White and Calnan (1990a) concluded that the bay is filling under the current loading regime. One study hypothesized that sediment delivery and accumulation are in equilibrium. White and Calnan (1990a) noted that estimates of relative sea-level rise from the Port Aransas area may show that relative sea-level rise is greater than other studies have anticipated. If so, San Antonio Bay may be getting deeper. The lack of detailed information about subsidence, relative sea-level rise, bathymetry, and sediment distribution by currents does not allow resolution of these conflicting conclusions about San Antonio Bay sedimentation.

Lavaca Delta. The Lavaca Delta protrudes about 3.7 km (2.3 mi) into Lavaca Bay. McGowen and Brewton (1975) compared the shoreline and wetland areas in the entire Matagorda Bay region between the mid-1850's and mid-1950's using U.S. Coast Survey charts for the years 1856 to 1859, and later topographic maps and photographs. They found little change in the lower delta during the 100-year period.

Farther up the river valley at Menefee Flat, some old vegetated areas are now open water and barren flats. White and Calnan (1990b) determined that vegetated wetland areas in the river valley decreased by 153 ha (378 acres) from 1930 to 1958, and by 430 ha (1,061 acres) from 1958 to 1979. They noted that levees and canals had altered the area where this submergence occurred, and local subsidence had been as great as 30 cm (11.8 inches) in 55 years. They also cited the slumping of the western valley wall and possible brine disposal from oil and gas fields as contributing to the vegetation loss.

These vegetation losses occurred before 1980, the year Lake Texana was completed, just upstream of the area. Although reservoir completion had no bearing on the documented vegetation changes, there is information to suggest that sediment supply to wetlands will be diminished now that Lake Texana is complete. Leopold et al. (1964) and Vanoni (1975) noted that reservoirs with capacities greater than 10,000 acre-ft can trap 95 to 100% of the incoming sediment. White and Calnan (1990a) presented a modified Brune trap efficiency curve, an empirical relationship between the percent of sediment trapped and the ratio of reservoir capacity to inflow. As the capacity-inflow ratio increases, water remains in the reservoir longer, providing greater opportunity for sediment deposition. Entrapment of sediment in several Texas reservoirs is consistent with the Brune curve, which estimates a trapping efficiency for Lake Texana of 95%.

Studies on the environmental impact of Lake Texana (TWDB 1974) estimated a much lower retention rate (32%) than suggested above. This estimate may be based on the texture analysis of the sediment carried by the Navidad River. The final environmental impact statement for the project (U.S. Department of the Interior 1972) states that 68% of the sediment carried by the Navidad River is in the clay-size range. The assumption may have been made that no clay-size particles would settle during the water's transit through the lake.

Blanton and Ferrari (1992) completed a sediment survey of Lake Texana covering the period May 1980 through June 1991. The average annual accumulation rate in the reservoir was 341 acre-ft/yr, and the net sediment accumulation rate from the contributing basin was 0.243 acre-ft $mi^{-2}yr^{-1}$. On this basis, Lake Texana retained about 43% of the sediment that flowed into the reservoir. However, flows through the reservoir during the period were about 30% higher than the long-term mean annual inflow, so 43% may be an overestimate.

The Lavaca River and Lake Texana Basin yield 273 and 784 acre-ft/yr of sediment (Greiner 1982), respectively. With the reservoir in place and using the 43% trapping efficiency estimate, the sediment load to the wetlands, delta, and Lavaca Bay may be reduced by as much as 32% compared to the loading before Lake Texana was constructed. White and Calnan (1990a) concluded that entrapment of the sediment probably will result in more extensive submergence of upriver wetlands in the future.

Colorado Delta. The Colorado River Delta developed as the result of the removal of a logjam in 1929 that extended 74 km (46 mi) upriver from the town of Matagorda. The logjam trapped most of the sediment load of the river. Within six years after the removal of the logjam, the delta prograded 6.4 km (4 mi) across the bay. A channel dredged through the delta and Matagorda Peninsula allowed the

Colorado River to discharge directly into the Gulf. The construction phase of the delta abruptly stopped in 1941 with the closing of most distributaries. This was attributed to impoundment of the Highland Lakes in Central Texas, construction of a farm road along the eastern arm of the delta, and connection of the river to the Gulf. Ward et al. (1980) noted that the mean sediment load to the delta is an order of magnitude less than it was during the period of rapid delta construction. Although no sediment load information is available before 1945, the Colorado River was considered a high sediment-load river until the Highland Lakes were built (Ward et al. 1980). White and Calnan (1990a) note that Lake Buchanan, at the head of the Highland Lakes, traps about 98% of the sediment flowing into it. Since there are six more dams downstream of Lake Buchanan and less than 10% of the basin area remains below the last lake in the chain to feed sediment to the river, the reservoirs probably have a strong influence on the amount of suspended sediment reaching the delta.

Manka and Steinmetz (1971) thought the eastern side of the delta was in a destructional phase, but White and Calnan (1990b) determined that the eastern side was stable between 1974 and 1982. On the western side, subdeltas have developed at Culver Cut, near the Intracoastal Waterway (ICWW), and at Tiger Island Cut, about 1.6 km (1 mi) from the Gulf entrance. According to Van Beek et al. (1980), the sediment for the Culver Cut subdelta is probably supplied from the river; it must pass southwest along the ICWW and out the cut. The sand contributing to the Tiger Island Cut subdelta comes largely from the Gulf (White and Calnan 1990a) since an upstream silting basin traps most of the sand transported by the river. Tiger Island Cut added 66 ha (135 acres) between 1979 and 1987.

The delta is now undergoing a major change. The mouth of the Colorado River has been diverted into Matagorda Bay west of the existing delta. It flows directly into Matagorda Bay, and will form a series of new subdeltas. Van Beek et al. (1980) have made the only quantitative predictions for any of the coastal deltas. They predict it will prograde into the eastern arm of Matagorda Bay at a rate of 25 ha/yr (37 acres/yr). About 8% of the annual sediment load will be retained in the delta. It will be instructive to follow the development of new subdeltas formed by the Colorado River to find out whether the prediction methods used by Van Beek et al. (1980) are accurate and can be applied to other delta sites.

Lavaca and Matagorda bays. Wilkinson and Byrne (1977) studied the depositional history of Lavaca Bay and showed that the bay has experienced slight shoaling in the past 3,000 years. They found that 73% of the bay fill sediment came from river flow, 24% came from shoreline erosion, and the remainder came from Matagorda Bay. Sedimentation during the past century occurred at twice the rate as in the previous 10,000 years, probably due to agricultural development in the basin (White and Calnan 1990a). With the completion of Lake Texana on the Navidad River, the major source of bay sediment decreased significantly. The input of river sediment will be only about 68% of the pre-Lake Texana sedimentation rate, based on information from White and Calnan (1990a) and Blanton and Ferrari (1992). White and Calnan (1990a) state that it is possible that sedimentation will fall behind relative sea-level rise resulting in a deepening bay system.

While the depth of Matagorda Bay has kept pace with sea-level changes over the past 10,000 years, most of the central portion of the bay has not changed in depth over the past century. The depth of the eastern arm of Matagorda Bay, influenced by the Colorado River, has decreased in the past century. With the diversion of the Colorado River into Matagorda Bay, Van Beek et al. (1980) predicted 1,370 acre-ft/yr of sediment will be transported into the bay, nearly twice as great as the load received by Lavaca Bay before Lake Texana. How the sediment will be distributed in Matagorda Bay is unknown.

Trinity Delta. The Trinity Delta that protrudes into Trinity Bay is 500 to 1,000 years old and has undergone 30 to 60 cm (1 to 2 ft) of subsidence during its construction (Failing 1969). It has actively prograded during the past century (Shepard 1953). In the past 50 years, the most southern portions of the delta with active distributaries have accreted. Older portions of the delta, below Old River Lake, have retreated slightly.

In the interior portions of the delta, White et al. (1985) documented the conversion of some marsh areas to open water; this may suggest the effects of subsidence or reduced sediment inflow. The delta experienced 22.5 cm (0.75 ft) of subsidence between 1943 and 1978. The U.S. Army Corps of Engineers (1981) stated that Lake Livingston, about 100 river miles upstream, traps all but 2% of the sediment that enters the reservoir. Figure 4.4.3 shows the effect of Lake Livingston in reducing suspended sediment at Romayor. For the period 1969 to 1986, the annual average suspended sediment load measured at Romayor was 526 acre-ft/yr. This compares with a 39-year average from 1936 to 1971 of 2,573 acre-ft/yr (Dougherty 1979). While sediment load measurements are not available before 1936, White and Calnan (1990a) cited studies that explained the current configuration of the delta and Trinity River valley, assuming long-term sediment loads of 3,000 acre-ft/yr. It appears that the present sediment load is significantly less than the long-term load that provided material for delta construction.

The effects of reduced sediment load on the delta's vegetated wetland areas are not clear-cut. White and Calnan (1990b) determined that wetland area increased by 600 ha (1,481 acres) between 1930 and 1956, but decreased by 414 ha (1,023 acres) from 1956 to 1974 (not including the conversion of 1,030 ha [2,543 acres] of wetland into a power plant cooling reservoir). The rate of wetland increase in the early period approximately equaled the rate of decrease from 1956 to 1974. Six of the 18 years during the latter period occurred after the completion of Lake Livingston, so the wetland area decrease could be related to reduced sediment loading. Between 1974 and 1988, vegetated wetland area decreased by 90 ha (222 acres). While the sediment load decreased by 75% during the most recent period, the rate of wetland loss was only 25% of the rate of the previous 18-year period, 12 years of which occurred before Lake Livingston was completed. If the loss of wetland area is due to decreased sediment, the relationship is not direct. The rate of wetland loss was higher during the 1956 to 1974 period, when the average sediment reduction was less than during later years. Dredging of the Anahuac Channel and partial construction of Lake Wallisville overflow dam and locking facilities within the delta are complicating factors in assessing the changes in delta area and relating the changes to reduced sediment inflow.

While subsidence, dredging, and Wallisville construction confound a clear interpretation of the effects of sediment loading on the Trinity Delta, White and Calnan (1990b) conclude that relative sea-level rise is surpassing sedimentation. This is consistent with a decrease in deltaic wetland area. They conclude that subsidence and reduced sediment loading from the Trinity River appear to have contributed to the decline of the deltaic wetlands. If the recommended alternative for Lake Wallisville is completed, the suspended sediment will decrease by an additional 35% and aggradation and progradation of the delta will occur only during high flows (U.S. Army Corps of Engineers 1981).

Galveston and Trinity bays. The major source of sediment for the Galveston Bay system has been the Trinity River (Paine and Morton 1986). White et al. (1985) report that sediment from the river is carried past Smith Point as far as East Bay, based on trace metal information. Between 1854 and 1933, the Galveston Bay system decreased in depth at a rate of about 44 cm/100 yr (1.4 ft/100 yr) (Shepard 1953), but, more recently, bay depth has increased. Morton and McGowen (1980) compared depth soundings taken in 1977 with 1968 National Ocean Survey bathymetric data. Depth increased in Galveston, Trinity, and East bays by 0 to 1.5 m (4.9 ft); this rapid subsidence is due to groundwater and mineral extraction. West Bay also deepened, but less than the other bays.

Paine and Morton (1986) evaluated shoreline changes in Galveston Bay and noted an average rate of shoreline retreat of 67 cm/yr (2.2 ft/yr). While erosion may have increased in recent years and produced additional bay fill, it probably does not offset the reduction in sediment from the Trinity River due to Lake Livingston (White and Calnan 1990a). So, the Galveston Bay system will continue to increase in depth.

Summary

Data and analytical methods. The quantitative data on suspended sediment is adequate to relate inflow and sediment load. However, there is not much information on sediment texture and practically no information about bed load. A significant amount of data in maps and photos exist to evaluate the areal extent of shoreline changes during periods of 50 to 100 years.

No conceptual model exists yet to provide a framework for quantitative analysis of sediment loading. While researchers have made some geologic measurements in deltas and upper bay areas, the measurements are not extensive enough to provide quantitative relationships between loading and delta or bay fill volume or area change. To complicate matters, there is evidence to suggest that some sediment movement into deltas is episodic (White and Calnan 1990a, b).

Conclusion. We can determine the amount of sediment transported by most rivers and relate it to inflow conditions, but we can provide only a qualitative assessment of what will happen to the sediment when it reaches the estuary. Areal loss of vegetation and gain of open water and barren flats have been measured for the major deltas, and trends defined. Translating these gains and losses into sediment volumes and loading requirements will require more extensive and accurate measurements of bathymetry, progradation, aggradation, sediment thickness, compaction, transport, and relative sea-level rise. Site-specific measurements of several of these critical variables are not available or are just approximations. A substantial research effort will be needed to relate sediment loading to maintenance of existing deltas or shallow water areas.

Sediment loads to the Nueces, Lavaca, and Trinity deltas are already influenced by upstream land uses. Reductions in sediment loads to these deltas have been documented. The effects of these reductions are not as clear, however. Subsidence, local geologic changes, levee and canal construction, and upstream agricultural practices all confound the assessment of changes in sediment loads on the deltas and bay areas. Nevertheless, it seems unlikely that areas that are submerging will regain their former elevations with reduced sediment loads, and it is possible that they will deteriorate further. The Guadalupe Delta is still actively undergoing the delta growth and decay cycle. The changes in its area do not seem directly related to sediment load from the Guadalupe and San Antonio rivers. Though the construction of the highlands lakes reduced sediment load to the Colorado River, the Colorado Delta probably will continue to grow on the western side when the diversion project is complete.

4.5 CONCLUSIONS

Hydrology. Inflow to Texas estuaries varies widely. From north to south, there is a general decrease in freshwater inflow. On the average, the volume of fresh water received by the Mission-Aransas and Nueces estuaries is less than each estuary's volume. Estuaries farther up the coast receive more inflow on a per-volume basis, with the Sabine-Neches Estuary receiving more than 50 times its volume each year.

Inflow varies from year to year, but the variation for the Mission-Aransas and Nueces estuaries is much greater than for systems farther up the coast. These two estuaries have more periods with very low flows than the other systems studied. All estuaries show monthly inflow variations, with the lowest inflows occurring during August in each estuary. The Sabine-Neches and Trinity-San Jacinto estuaries have peak flows during the spring. The Nueces and Mission-Aransas inflow peaks historically have occurred with storm events during the fall. Middle-coast estuaries have both spring and fall inflow maxima.

Only the Mission-Aransas estuary had a significant trend in inflows during the past 47 years, an increase of 2.1%/yr. Two periods ending in droughts (1941 to 1958 and 1958 to 1966) showed significant decreases in inflow for nearly all estuaries. During the period 1966 through 1987, however, there were no statistically significant trends. The Nueces Estuary showed a large decrease (-4.33% per year) in inflows over the latter period, but the large variability of the inflow record prevented the decrease from being statistically significant.

Average salinities of Texas estuaries are directly related to the number of annual inflow volumes each estuary receives. Bays with lower salinities generally receive a greater number of inflow volumes than those with higher salinities. All estuaries display a salinity gradient that increases from the upper to the lower portion of the estuary.

A trend analysis for various areas of Texas estuaries showed that the salinity of the lower Sabine-Neches Estuary decreased by about 3%/yr from 1968 to 1987. At the same time, salinity increased by around 2%/yr for West Galveston Bay and lower mid-San Antonio Bay. Several portions of the Nueces Estuary had increases in salinity, but Nueces Bay was the only portion of the system in which the change was large enough to be considered real.

Nutrients. Inflows provide the majority of nutrient loading to Texas estuaries. The proportion of the load carried by river flow varies with respect to amounts delivered by return flows and direct precipitation. Gaged flow provides 24 to 34% of the nitrogen load to the Mission-Aransas and Nueces estuaries, about 43% to the Trinity-San Jacinto, 49% to the Lavaca-Colorado, and 80% to the Guadalupe estuary.

In all estuaries, there is a decreasing nutrient concentration gradient from the head to the mouth. The gradient exists under both high- and low- inflow conditions. The magnitude of nutrient loading varies substantially from one estuary to another, but the residence time of the inflowing water lessens the loading differences. Even under low-inflow conditions, it does not appear that large areas of the bays are nutrient-limited. The lagoonal arms of the bays have the lowest concentrations of nitrogen during low flow conditions, but are zones of efficient use and recycling. Heterotrophic regeneration and high turbidity in the upper reaches of estuaries coupled with lower turbidity and efficient benthic regeneration in the lower estuary allow nutrients to move through the system and be reused without encountering problems of eutrophication.

Sediment. There is adequate information about suspended sediment loads for the major rivers flowing to Texas estuaries, but almost no information about bed load. The relative importance of bed load compared to suspended load in providing sediment to deltas and bay areas is not known. Suspended sediment consists of sand, silt, and clay. Among the rivers flowing to Texas estuaries, the proportion of silt does not vary much. Rivers with high flows, such as the Trinity and Sabine, carry high proportions of sand (20 to 38%) compared to the inflowing rivers to the south (0 to 5%). The Trinity and Sabine rivers also have relatively low levels of clay (38 to 58%) compared to the other river systems (more than 70%). Rivers with much lower flows, such as the Nueces, carry more than 85% of their suspended sediment as clay, with practically no sand.

The records for suspended sediment load for the Sabine, Colorado, and Nueces rivers do not predate the construction of large upstream reservoirs, so the effects of reservoir construction on sediment loading cannot be measured. Suspended sediment load of the Trinity River was measured from 1936 to 1986. The completion of Lake Livingston on the Trinity River in 1968 brought about a clear decrease in suspended sediment load that has continued through 1986. Other rivers show changes in suspended sediment load that do not appear to be related to human activities. The Lavaca, Guadalupe, and San Antonio rivers displayed significant changes in sediment loading in 1958, at the end of a prolonged drought. The decreased sediment load is still evident. The Nueces River had a substantial decrease in suspended sediment load beginning in 1972. The decrease, following the largest annual inflow on record in 1971, was still obvious in 1986.

Recent studies have documented reductions in the Nueces and Trinity delta areas that are most likely related to reservoir construction. Reductions in the Lavaca Delta are probably not reservoir-related, and changes in the Guadalupe Delta seem to be part of the normal delta construction-destruction cycle. While we can relate inflow and sediment load quantitatively, we have only a qualitative understanding of the relationship between sediment loading and the building of deltas or maintenance of bay-bottom bathymetry.

CHAPTER 5: INFLOW EFFECTS ON PRIMARY PRODUCTION, CONSUMERS, NUTRIENT CYCLING, AND ORGANIC CARBON USE

5.0 INTRODUCTION

The previous chapter presented information about the effects of freshwater inflow on the hydrology, nutrient loading, and sediment loading of Texas estuaries. The quantities and patterns of supply of these basic ingredients are the features of estuarine ecosystems most directly affected by upstream uses of fresh water. This chapter links these essential inputs to the organisms that form the basis of the estuarine food web: the plants and primary consumers. These species constitute a substantial part of the "estuarine life on which . . . fish and shellfish are dependent," which, in turn, is an important managerial concern for the TNRCC in making permit decisions concerning water rights [TEXAS WATER CODE 11.147(a)]. The major goal of this chapter is to document and, where possible, to quantify the relationships between inflow and various fundamental biological components and processes in estuarine ecosystems. An additional goal is to evaluate the relationships and identify those components or processes that can be used quantitatively to determine freshwater inflow needs.

Phytoplankton. Section 5.1 reviews information about phytoplankton standing crop and primary production in Texas bays. Results from older studies as well as two recent studies in the Guadalupe and Nueces estuaries are included.

Submerged grasses and marsh plants. Estuarine vascular plant communities—submerged vegetation and marsh plants—are the subject of Section 5.2. Salinity relationships, underwater light requirements, inundation needs, and substrate conditions are the major inflow-related factors affecting vascular plant distribution and abundance that are discussed in the section.

Zooplankton. The relationships between freshwater inflow and the abundance and productivity of zooplankton are investigated in Section 5.3. This section includes information about micro- and macrozooplankton, although the emphasis is on the latter. An analysis of macrozooplankton abundance under high- and low-inflow conditions in Lavaca Bay, the Guadalupe Estuary, and the Nueces Estuary is included.

Benthos. Section 5.4 details the effects of freshwater inflow on benthic organisms. Recent studies have provided information on inflow effects on microbenthic, meiobenthic, and macrobenthic groups. Differences in diversity and abundance with respect to salinity regimes are discussed for several estuaries. In addition, the effect of inflow on regeneration of nutrients by benthic organisms is also reviewed.

Nutrient cycling. Special studies on the Guadalupe and Nueces estuaries have provided an in-depth understanding of nutrient cycling in Texas bays. Section 5.5 compares the cycling rates and storage of materials in the two estuaries under contrasting conditions of high and low inflow. Differences in nitrogen cycling are presented in some detail.

Terrestrial and delta carbon use. Section 5.6 considers the distribution and use of organic material from terrestrial and river delta marsh sources, as determined by stable carbon isotope tracers. The extent of use of delta and terrestrial organic matter is described from carbon measurements on sediment, particulate organic material, and consumer and predator species in three Texas estuaries.

Table 5.1.1. Major categories of phytoplankton common in Texas estuaries, ranked by order of importance.

Estuary	Upper bay	Lower bay	Reference
Sabine-Neches	greens diatoms blue-greens	diatoms greens	TDWR (1981e)
Trinity-San Jacinto	diatoms greens blue-greens		TDWR (1982b)
Lavaca-Colorado	cryptophytes greens diatoms		Gilmore et al. (1976), Jones et al. (1986)
Guadalupe	cryptophytes greens diatoms		Matthews et al. (1975)
Mission-Aransas	blue-greens greens	diatoms dinoflagellates greens	Holland et al. (1975)
Nueces	blue-greens diatoms	diatoms dinoflagellates	Holland et al. (1975)
Laguna Madre	diatoms dinoflagellates	diatoms greens	TDWR (1983)

5.1 ESTUARINE PHYTOPLANKTON, PRIMARY PRODUCTIVITY, AND FRESHWATER INFLOWS

Introduction

The production of biomass by phytoplankton (suspended micro-algae) inhabiting the estuaries is a major source of organic material entering the estuarine food chain. This section describes current information on phytoplankton in Texas estuaries, the importance of their productivity, and how the phytoplankton are influenced by environmental factors including the quality and quantity of freshwater inflows.

Plankton Groups and Biomass

Composition. The phytoplankton comprise a diverse assemblage of algal species, sizes, and shapes, with various capacities for photosynthetic conversion of sunlight and dissolved nutrients into organic matter. Marine species and freshwater species mix in the estuaries. The relative composition of these species in a bay varies seasonally, to some extent, and as the salinity gradient changes. For the purpose of assessing the general character of estuarine primary production, phytoplankton species are often grouped by major taxonomic divisions. The relative importance of these groups may indicate the ecological health and functioning of the system, how accessible the production is to consumers, and whether the system may be dominated by marine or freshwater species.

Table 5.1.1 presents an overview of the predominant phytoplankton groups in each estuary, summarized from the references listed. The table illustrates only the most general level of contrast. However, differences in relative dominance of phytoplankton groups among the estuaries may determine which pathways are most important in moving the photosynthetic carbon into higher trophic levels. Diatoms, for example, are generally considered a more available food source for zooplankton than blue-green or many green algae (Ryther and Officer 1981). Marine diatoms are more prevalent in the bays of the lower coast than in the other bays, and are generally most prevalent in the portions of the bays proximal to the barrier islands. Therefore, phytoplankton productivity in these bays contribute directly to the zooplankton link in the food chain. Freshwater algal species may dominate the upper bays during times of high inflow. Because some freshwater phytoplankton species are not the preferred food of zooplankton, they may enter the estuarine food chain through benthic filter feeders, rather than through the planktonic food chain.

Standing crop. Realistic comparisons of estuaries based on phytoplankton standing crop would require more abundance and biomass measurements than are currently available (see discussion on variation below). In addition, recent studies demonstrate that more attention should be directed to accurately sample the smallest phytoplankton species, as these may constitute a large proportion of the total biomass (Jones 1986; Stockwell 1989).

Measurement of chlorophyll extracted from algal cells has long been used as a quantitative index of phytoplankton biomass. Chlorophyll data from the TWDB Coastal Data System are presented in Table 5.1.2. Monthly to quarterly samples were taken at several locations in each bay, from 1968 to 1989. The average chlorophyll concentrations among the seven Texas estuaries range from 12.9 µgm/l in the Trinity-San Jacinto Estuary to 3.1 µgm/l in the Mission-Aransas system. The average chlorophyll levels in these estuaries fall in the mid-range of averages for river-dominated estuaries listed by Boynton et al. (1982). Concentrations are generally higher in the upper portions of five of the estuaries, as might be expected in response to higher nutrient levels there. The other two estuaries, Laguna Madre and Mission-Aransas, have higher chlorophyll concentrations in the vicinity of passes to the Gulf. These areas may receive nutrient additions other than through their upper bays, Aransas Bay from the Guadalupe Estuary, and lower Laguna Madre from the Arroyo Colorado.

Table 5.1.2. Average (Avg.) chlorophyll concentrations (µgm/l) and standard deviations (St.D.) for total, upper, and lower regions of seven Texas estuaries. Values are from TWDB and TNRCC monitoring programs.

Estuary	Total		Upper		Lower	
	Avg.	St.D.	Avg.	St.D.	Avg.	St.D.
Sabine-Neches	5.5	4.6	6.3	5.4	4.9	3.6
Trinity-San Jacinto	12.9	18.3	15.9	17.9	9.6	10.3
Lavaca-Colorado	6.4	7.2	8.2	9.1	4.5	3.6
Guadalupe	9.9	14.4	18.0	23.6	6.3	6.0
Mission-Aransas	3.1	2.9	1.0	0.9	3.9	2.9
Nueces	5.3	5.8	8.6	6.8	3.9	3.2
Laguna Madre	9.6	33.4	6.4	6.7	13.3	47.5

Variation. Phytoplankton are not uniformly abundant in estuaries, either temporally or spatially. Figure 5.1.1, from Armstrong and Hinson (1973), shows typical variation in numbers of phytoplankton suspended in the water column. Phytoplankton were sampled using a fine-mesh plankton net, so the abundance data show the distribution primarily of large marine diatoms in the estuary. Whitledge (1989a, 1989b) presents detailed descriptions of the spatial variation in phytoplankton chlorophyll for the Guadalupe Estuary and the Nueces Estuary. Figure 5.1.2 (from Whitledge 1989a) illustrates the high phytoplankton biomass occurring between pulses of freshwater inflow to San Antonio Bay in 1987.

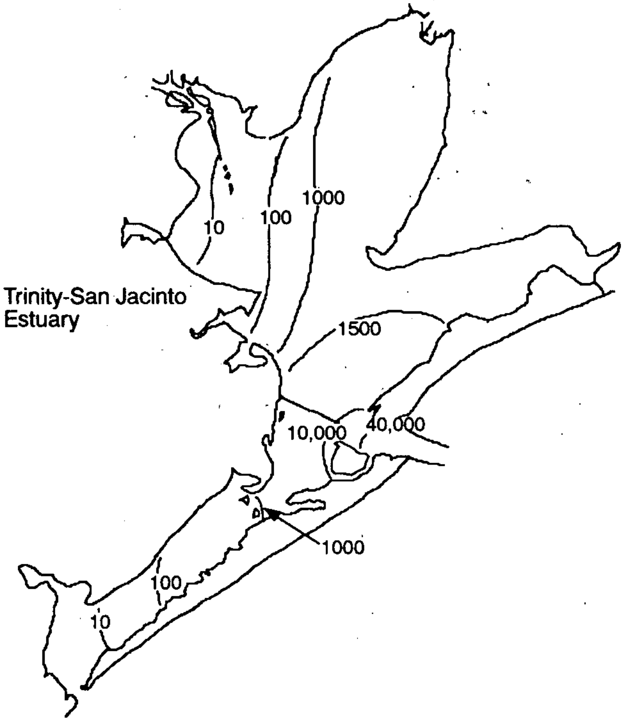


Figure 5.1.1. Phytoplankton concentration (cell numbers/ml) in the Trinity-San Jacinto Estuary, April 15 to 24, 1969 (from Armstrong and Hinson 1973).

In any bay, the numbers and kinds of phytoplankton characteristically vary with the seasons and over shorter periods (e.g., week to week) as some species bloom at the expense of others (Holland et al. 1975; Copeland and Fruh 1969; Gilmore et al. 1976). As a particular algal species finds the combination of temperature, salinity, and nutrients favorable within a bay, its population may grow to tremendous numbers, producing a characteristic color in the water. Some blooms attract planktivores, while in other cases, blooms make the water distasteful or even toxic to other species (red tides). Some blooms continue for months, while others last for only weeks. Therefore, the history of the phytoplankton over a year frequently shows dramatic population fluctuations. A number of the studies cited in this section provide data on the seasonal variation encountered.

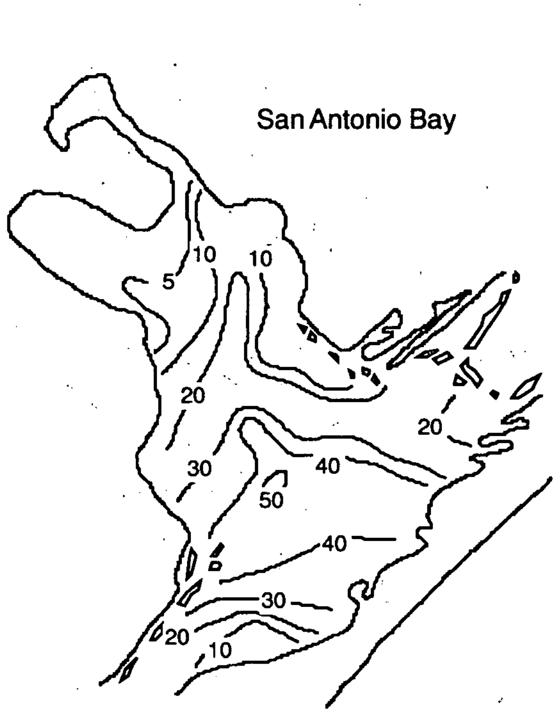


Figure 5.1.2. Chlorophyll-*a* concentration (μgm/l) in San Antonio Bay, March 4, 1987 (from Whitledge 1989a).

Within these large-scale shifts, daily changes may also occur. Stockwell (1989) documented three- and four-fold day-to-day changes in surface chlorophyll at sites in Nueces Bay. Because of the typical variation in phytoplankton populations, many samples of the plankton community are required before the estuarine planktonic biomass can be determined with statistical confidence.

Primary Productivity

Primary productivity of Texas estuaries have been measured by techniques to assess community-wide production using the diurnal curve method (e.g., Odum and Wilson 1962) and by using carbon-14 methods. Table 5.1.3 presents average rates of production from studies of most major Texas estuaries. These figures show Texas estuaries to be among the more productive estuaries in the U.S. in comparison with data presented by Boynton et al. (1982) for river-dominated estuaries. This productivity comparison corresponds to similar comparisons of nutrient richness presented in Section 4.3. Productivity measurements from these estuaries are not yet numerous enough to establish a true range of variation. However, from the data available, it appears that the rates of production among these estuaries are similar. Although these estuaries differ in the amount of freshwater inflow received, average salinity, and other characteristics, the adaptive responses of the phytoplankton community in each estuary lead to similar levels of production.

The importance of pelagic production. Most of the organic production in the bays of the upper coast may originate with phytoplankton. However, in the lower coastal bays, seagrasses and their epiphytes play an increasing role (Pulich 1980, see Section 5.2). The rates of production reported for upper and lower Laguna Madre in Table 5.1.3 were measured by the diurnal curve method and may include substantial seagrass production. Most of the phytoplankton contribution to Laguna Madre occurs in Baffin Bay (Hedgpeth 1967).

In addition to the carbon from phytoplankton photosynthesis, other sources of organic carbon in estuaries include river-borne detritus, material transported into the bay from tidal marshes, wastewater discharges, and runoff. Flint (1984) estimated that other sources of fixed carbon besides phytoplankton account for 48% of the Nueces Estuary total annual carbon input. Table 5.1.4 compares average areal primary production to average inputs of organic carbon from combined river, runoff, and waste discharge input (Section 4.3). Phytoplankton production may provide an order of magnitude more fixed carbon than external sources for a few estuaries all of the time. Phytoplankton production exceeds the inputs of other sources of carbon in Texas estuaries where measurements have been made. Section 5.6 discusses other measures of the relative importance of various carbon sources.

Table 5.1.3. Average primary production rates in Texas estuaries in gm C m^{-2} day $^{-1}$. An asterisk (*) marks values standardized by the procedure in Flint (1984).

Estuary	Rate	Reference
Trinity-San Jacinto	2.18	Armstrong and Hinson (1973)*
Lavaca-Colorado	0.502	Ward and Armstrong (1982)
	2.41	Davis (1973)*
Guadalupe	1.18	MacIntyre and Cullen (1988)
Nueces	1.76	Odum and Wilson (1962); Odum et al. (1963)*
	0.48	Flint (1984)
	1.22	Stockwell (1989)
Upper Laguna Madre	2.68	Odum and Wilson (1962); Odum et al. (1963)*

Table 5.1.4. Phytoplankton productivity and external total organic carbon (TOC) loading. Loading data presented in Section 4.3, values are gm C $m^{-2}day^{-1}$.

Estuary	Primary production	TOC load
Trinity-San Jacinto	2.2	0.390
Lavaca-Colorado	0.50 to 2.4	0.130
Guadalupe	0.70 to 1.2	0.120
Mission-Aransas		0.047
Nueces	0.50 to 1.0	0.030

Factors controlling primary productivity. The investigation of environmental controls on phytoplankton productivity has been a focus of studies of Texas estuaries. Armstrong and Hinson (1973) investigated the potential limitation of algal growth by toxic chemicals, but the emphasis of most studies has been on potential limitation by nutrient concentrations. Phytoplankton of the Trinity-San Jacinto Estuary were stimulated with additions of nitrate and phosphate, but productivity was not correlated with those nutrient concentrations in the water column (Armstrong and Hinson 1973). Productivity was assumed to be limited by light penetration. Simple correlation analysis found that chlorophyll concentrations in the Guadalupe Estuary were positively associated with inflow rate and nitrite and negatively associated with salinity and water temperature (Matthews et al. 1975).

Table 5.1.5. Average (Avg.) Secchi disk depths (cm) with standard deviations (St.D.) for Texas estuaries, from TWDB and TNRCC monitoring data.

Estuary	Total		Upper		Lower	
	Avg.	St.D.	Avg.	St.D.	Avg.	St.D.
Sabine	72	47.2	67	33.3	70	58.3
Trinity-San Jacinto	59	35.6	50	25.2	63	41.1
Lavaca-Colorado	58	40.6	49	34.2	85	49.6
Guadalupe	52	36.5	33	19.9	70	40.6
Mission-Aransas	74	44.8	58	32.3	87	48.8
Nueces	86	50.9	33	21.1	102	46.9
Laguna Madre	78	47.9	88	59.8	69	34.6

For the Nueces Estuary, Flint et al. (1983) developed a regression equation which predicted daily productivity from salinity, water temperature, ammonia nitrogen concentration, average total daily sunlight for the sample month, Secchi disc depth, station water depth, and average water surface sunlight for the day of sampling. Stockwell (1989) found that salinity, temperature, and total nitrogen per square meter were useful in explaining chlorophyll variations in the Nueces Estuary. The best predictor of primary productivity developed from his data, however, was an equation based on phytoplankton biomass and light availability:

$$P = -217.8 + 2.49\ (B^*Z_p{}^*I_0), \quad R^2 = 0.71,$$

where P is the rate of production in mg C $m^{-2}day^{-1}$, B is phytoplankton biomass (chlorophyll-*a*, in mg/m^3), Z_p is the depth of 1% isolume, and I_0 is the surface irradiance in units of Einstein $m^{-2}day^{-1}$ (an Einstein is the radiant energy required to effect a photochemical transformation of one mole of reactant and varies with the frequency of the radiant energy).

The bays of Texas estuaries are frequently very turbid, usually from clays kept suspended in the water by wind-induced turbulence. A common measure of turbidity is Secchi disk depth, the depth at which a white disk lowered into the water just disappears. Average Secchi disk depths range from 52 to 86 cm among the estuaries (Table 5.1.5). In contrast, Secchi disk depths in the Gulf at the Port Isabel jetties are commonly 300 to 400 cm, and occasionally down to 800 cm (data from the TNRCC Statewide Monitoring Network). The rate of light extinction with depth is inversely proportional to Secchi disk depth. Thus, light available for phytoplankton production in water with a Secchi disk depth of 25 cm is only 1/500 of the light available in water with a 250 cm Secchi depth. Turbidity, seasonal temperature variation, and flushing rate are physical parameters important in determining primary production in estuaries (Boynton et al. 1982). Additionally, algal growth rates may be controlled by concentrations of dissolved nitrogen falling below optimal levels. Boynton et al. (1982) regressed productivity data from many estuaries against nutrient loading per unit area. The results suggested that nitrogen input to an estuary has a stronger influence on system production than does the input of phosphorus. Whitledge (1989a, 1989b) presents evidence that nitrogen is the chief nutritional limit on phytoplankton growth in the Guadalupe and Nueces estuaries.

Freshwater Inflow Effects

Taxonomic groups. Inflow rates definitely affect the relative abundance of various species of phytoplankton in the estuary, especially in the upper estuary. Gilmore et al. (1974) reported blooms of small diatoms and microflagellates in Lavaca Bay following periods of high inflow, whereas marine diatoms became important when river inflows remained below 500 cfs. A similar situation was observed by Holland et al. (1975) in Copano Bay.

Biomass. The relationship between estuarine chlorophyll concentrations and inflow rates may indirectly indicate the relationship between inflow and phytoplankton productivity for a given bay. Regression of San Antonio Bay chlorophyll data from Stockwell (1989) with freshwater inflow volume during the month of sampling showed that increased inflows produced higher chlorophyll concentrations, but explained only 39% of the variation. Surface chlorophyll data collected in routine TWDB and TNRCC monitoring of stations in mid-San Antonio Bay do not show a simple relationship with inflows. Figure 5.1.3 displays the relationship between surface chlorophyll sampled during 1980-1988 period and the surface inflows summed during the 30 days preceding sampling. There is an apparent increase in chlorophyll concentrations from very low to moderate inflows, but a slow decrease in concentrations occurs as flows continue to increase.

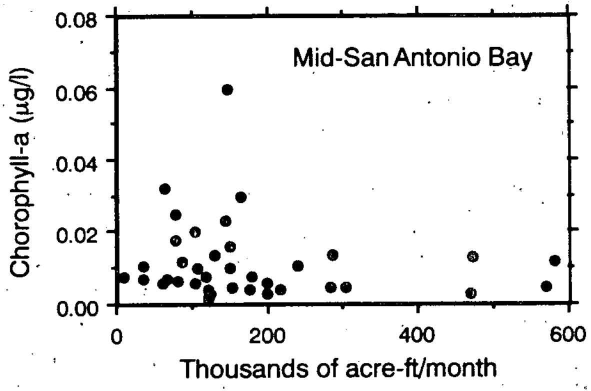


Figure 5.1.3. The relationship between chlorophyll-*a* (mg/l) in mid-San Antonio Bay (from TWDB and TNRCC monitoring programs) and the combined freshwater inflow to the Guadalupe Estuary summed during the preceding 30 days.

Productivity. Freshwater inflows bring nutrients into the bay, but high flows may physically flush the upper estuary of estuarine phytoplankton. The point at which the effects of flushing overcome increased productivity cannot be determined from the data presently available. High rates of primary production exist in both the Trinity-San Jacinto Estuary, with its high inflow rates and high turbidity, and the Laguna Madre with low inflow rates and higher water clarity. In general, data discussed here suggest that all Texas estuaries have productive phytoplankton. Future concerns for these important primary producers of Texas bays will focus on the kinds of algae that are productive and the conditions favorable for species which contribute to secondary production. Studies that identify conditions for growth of noxious species should also be done (Dean Stockwell, UTMSI, personal communication; cf. Ryther and Officer 1981).

5.2 EFFECTS OF FRESHWATER INFLOWS ON DISTRIBUTION AND PRODUCTIVITY OF ESTUARINE WETLAND FLORA AND SUBMERGED VEGETATION

Introduction

The wetland plant communities important to estuarine fishery and wildlife organisms for nursery or feeding habitat consist of salt marshes, brackish marshes, freshwater/intermediate delta marshes, and submerged vascular vegetation. Each community is defined by the occurrence of certain characteristic plants and their associated physical-hydrological environment (Cowardin et al. 1979). Common vascular wetland species are listed in Table 5.2.1 according to community types and the Texas estuaries where they frequently reach dominance.

Ecological gradients. Along the Texas Coast, a well-defined north-south gradient is evident in the amount of these habitat types within the seven major estuarine systems. Intermediate marshes are most extensive along the upper coast in the Sabine and Trinity River deltas [13,060 ha (32,258 acres) in Trinity delta (USFWS 1987)], and decrease dramatically going southward. The Guadalupe Delta contained about 4,833 ha (11,942 acres) of low-salinity brackish marsh in 1976, while the Nueces Delta had 5,350 ha (13,220 acres) at this time (Adams 1977). Permanent freshwater marsh assemblages typically occupy river drainage bottomlands and frequently are interspersed with fluvial woodland species [e.g., willow (*Salix*) and cypress (*Taxodium*)].

The dominant species composition of brackish marsh communities consists of bulrush (*Scirpus* spp.), marsh hay cordgrass (*Spartina patens*), and saltgrass (*Distichlis spicata*) on the upper coast around Sabine Lake, Trinity Bay, and Galveston Bay. This changes to fringe smooth cordgrass (*Spartina alterniflora*) saltmarsh on the central coast (Matagorda, San Antonio, Copano, and Aransas bays), and eventually yields to succulent halophytes including glass-

Table 5.2.1. Dominant plants of Texas estuaries and respective communities (from Pulich 1990).

Scientific name	Common name	Community type Submerged	 Salt marsh	 Brackish marsh	 Fresh/ intermediate marsh	Estuary where dominant
Thalassia testudinum	Turtlegrass	S				M-A[e]
Syringodium filiforme	Manateegrass	S				M-A, N[f], LM[g]
Halophila engelmanni	Clovergrass	S				L-C[h], M-A, N, LM
Halodule wrightii	Shoalgrass	S[a],B[b]				All except S-N[c]
Ruppia maritima	Widgeongrass	S, B, F[d]				All
Vallisneria americana	Wild celery	B, F				T-SJ
Najas guadalupensis	Water nymph	F				T-SJ[i], L-C, G[j]
Heliotropium curassavicum	Beach heliotrope		X			All
Salicornia bigelovii	Annual glasswort		X			All
Aster tenuifolius	Salt marsh aster		X	X		All
Avicennia germinans	Black mangrove		X	X		G, M-A, N, LM
Batis maritima	Saltwort		X	X		All
Borrichia frutescens	Sea oxeye		X	X		All
Distichlis spicata	Seashore saltgrass		X	X		All
Iva frutescens	Sumpweed		X	X		All
Lycium carolinianum	Wolfberry		X	X		All
Monanthochloe littoralis	Shoregrass		X	X		All
Salicornia virginica	Perennial glasswort		X	X		All
Spartina alterniflora	Smooth cordgrass		X	X		All except LM
Spartina patens	Marsh-hay cordgrass		X	X	X	All
Spartina spartinae	Gulf cordgrass		X	X	X	All
Juncus roemerianus	Needlerush			X		S-N, T-SJ, L-C
Scirpus maritimus	Saltmarsh bulrush			X		All except LM
Scirpus olneyi	Olney bulrush			X		S-N, T-SJ, L-C
Bacopa monnieri	Water hyssop			X	X	All
Hydrocotyle spp.	Pennywort			X	X	All
Paspalum vaginatum	Seashore paspalum			X	X	S-N, T-SJ
Phragmites australis	Common reed			X	X	All except LM
Scirpus americanus	Three-square bulrush			X	X	All
Typha domingensis	Narrowleaf cattail			X	X	All
Alternanthera philoxeroides	Alligator weed				X	S-N, T-SJ
Eichhornia crassipes	Water hyacinth				X	T-SJ, L-C, G
Sagittaria spp.	Arrowhead				X	S-N, T-SJ, L-C, G
Scirpus californicus	Bulrush				X	S-N, T-SJ, L-C, G
Zizaniopsis miliacea	Giant cutgrass				X	S-N, T-SJ, L-C, G

[a] S: Submerged saltwater habitat
[b] B: Submerged brackish water habitat
[c] S-N: Sabine-Neches Estuary
[d] F: Submerged freshwater habitat
[e] M-A: Mission-Aransas Estuary
[f] N: Nueces Estuary
[g] LM: Laguna Madre Estuary
[h] L-C: Lavaca-Colorado Estuary
[i] T-SJ: Trinity-San Jacinto Estuary
[j] G: Guadalupe Estuary

worts (*Salicornia* spp.), saltwort (*Batis maritima*), and black mangrove (*Avicennia germinans*) in salt marshes on the lower coast (Corpus Christi Bay and Laguna Madre). Diener (1975) listed acreage of estuarine marsh vegetation as: 171,995 ha (425,000 acres) for Sabine Lake; 93,646 ha (231,400 acres) for Galveston Bay; 48,563 ha (120,000 acres) for Matagorda Bay; 10,117 ha (25,000 acres) for San Antonio Bay; 18,211 ha (45,000 acres) for Copano-Aransas bays; and 18,211 ha (45,000 acres) for Corpus Christi Bay. As calculated from additional data in Diener, the ratio of submerged vegetation acreage to emergent marsh acreage is 0.078 in Galveston Bay, 0.059 in Matagorda Bay, 0.654 in San Antonio Bay, 0.092 in Copano-Aransas bays, 0.283 in Corpus Christi Bay, and 0.764 in Laguna Madre. These ratios express quantitatively the increasing abundance of submerged vegetative habitat from north to south, and the concurrent decreasing dominance of emergent marsh habitat.

These latitudinal shifts in abundance and dominance of estuarine plant communities are attributable to the pervasive influence of freshwater inflows on Texas coastal wetlands (Copeland 1966). Texas estuarine systems vary along well-defined geographic gradients of precipitation and temperature (Thornthwaite 1948; TDWR 1982c; Bomar 1983), and vegetation develops in response to the unique topography and hydrologic conditions existing in each estuary. The vascular plant species persist at bay locations only if they are adapted to the salinity and inundation regimes associated with the precipitation and temperature gradients. For the most part, the bays from Matagorda Bay southward along the Texas coast are located in arid regions which experience net annual water deficits (annual evapotranspiration exceeds rainfall). Because precipitation, both inland and on the bay, controls the amount and timing of freshwater inflows, the latter ultimately controls the salinity conditions in the bay waters. Dominant species comprising the plant communities are those which are most competitive over basic inundation cycles (i.e., hydroperiod) and salinity, nutrient, and sediment-loading regimes caused by tides and freshwater inflows, coupled with temperature regimes (Copeland 1966; Chabreck 1972).

Environmental factors and vegetation tolerance limits. Because the immediate effect of altered inflows is usually on bay water salinities, response of many estuarine plant species to variations in salinity has been extensively investigated. Table 5.2.2 lists information on salinity limits and optimal ranges observed for common Texas estuarine vascular plants (seagrass references from Table 9-1 in TDWR 1983 by Gary L. Powell, with emergent marsh plant and additional seagrass references added by section author). These data were derived from both laboratory and field investigations under widely variable situations and over broad geographic ranges. Examination of these limits indicates that physiological responses can differ significantly from ecological or growth responses for a given species. Generally, physiological optima are well defined, while ecological limits cover a wider range of salinities. This reflects the interaction between salinity and other growth factor requirements which can ameliorate the salinity effect. An estuarine species will grow abundantly, although not at maximum rate, where the combination of growth factors is in a favorable range, which may be quite broad.

Another consideration is the distinction between water column salinity and soil water or root zone salinity. Roots exist in a totally different environment from aerial shoots and leaves. These tissues, which have a distinctly different physiology, are adapted to the special characteristics of either

Table 5.2.2. Extreme salinity limits and optimum ranges for selected Texas estuarine-dependent plants.

Group/species	Limits min. (‰)	 max. (‰)	Optimum (‰)	Remarks
Submerged vegetation				
Halophila engelmanni			37	Plants flowered profusely in March following transplantation to 14-hour photoperiod in laboratory (McMillan 1974).
			25 to 36	Plants observed flowering in Redfish Bay, Texas, from April to June at 25 to 29 °C temperatures (McMillan 1976).
	13 to 18	50	23 to 37	Plants transferred from 37 to 13‰ and 18‰ salinity retained some green tissue for at least a week; plants did not survive at 50‰ past eight weeks; 23 to 37‰ salinity range of good survival for 13 weeks (McMillan 1974).

(continued)

Table 5.2.2. Continued from previous page.

Group/species	Limits		Optimum (‰)	Remarks
	min. (‰)	max. (‰)		
Manateegrass *Syringodium filiforme*		40		Plant leaves showed height increase to 40‰ as salinity was increased during 55-day test (McMillan and Moseley 1967).
		44		Plants were rated not vigorous at 44‰ salinity and died after three-week laboratory test at 52.5‰ (McMahan 1968).
	10	50		Salinity range of green tissue survival for two weeks or more in laboratory; plants did not survive at 5‰ and 60‰ salinity (McMillan 1974).
Shoalgrass *Halodule wrightii*	3.5	52.5		Range of survival after six-week test in laboratory (McMahan 1968).
	<5	80	23 to 37	Plants transferred from 37‰ to tap water and 5‰ conditions retained green tissue (leaves) for two weeks; plants retained green tissue in evaporated artificial seawater salinities up to 80‰; 23 to 37‰ range of good survival for 13 weeks (McMillan 1974).
		>72		Plant leaves showed continuous height increases to 72‰ as salinity was increased during 55-day test (McMillan and Moseley 1967).
	10	60		Salinity range reported for this adaptable tropical species which also tolerates 7 to 32 °C temperatures in northern Gulf of Mexico (Phillips 1980).
	21	35	22 to 31	Maximum production (biomass) in Redfish Bay, Texas, occurred at water temperatures of 28 to 31 °C and salinities of 22 to 31‰ during July to September of two years (Pulich 1985).
	25	50	30 to 36	Maximum production (biomass) in Laguna Madre, Texas, occurred at water temperatures of 30 °C and salinities of 30 to 36‰ during June to August (Pulich 1985).
Turtlegrass *Thalassia testudinum*			28 to 32	Flowering of Florida plants in laboratory (Marmelstein et al. 1968).
			37	Flowering (staminate only) of plants from Port Isabel, Texas, observed in April following transplantation to 14- and 16-hour photoperiods in laboratory (McMillan 1974).
		48	33 to 38	Maximum field distribution in Florida; salinity range of common occurrence (Phillips 1960).
		<60		Plant leaves showed no further height increases beyond 60‰ as salinity was increased during 55-day test (McMillan and Moseley 1967).
	10	50		Salinity range of green tissue survival for two weeks or more in laboratory; plants did not survive at 5‰ salinity for more than a few days (McMillan 1974).
			20 to 35	Optimum salinity range reported for this restricted tropical species (Phillips 1960).
Widgeongrass *Ruppia maritima*			<28	Flowering and seed formation reported to occur in Florida at less than 28‰ salinity (Bourn 1935).
		33.2	<25	Maximum field distribution in Florida; flowering and greatest plant occurrence in areas <25‰ salinity (Phillips 1960).

(continued)

Table 5.2.2. Continued from previous page.

Group/species	Limits		Optimum (‰)	Remarks
	min. (‰)	max. (‰)		
	0		<30	Plants survived indefinitely in tap water; flowering in laboratory of Redfish Bay, Texas, plants observed in tap water and salinities <30‰ (McMillan 1974).
		74	<46	Plants tolerated salinities up to 74‰ for brief periods in laboratory; plant leaves showed height increases to 46‰ as salinity was increased in outdoor tank during 55-day test (McMillan and Moseley 1967).
	21	35	22 to 32	Maximum production (biomass) in Redfish Bay, Texas, occurred at water temperatures of 22 to 30 °C and salinities of 22 to 32‰ during April through June of two years (Pulich 1985).
	25	50		Moderate production (biomass) in Laguna Madre, Texas, occurred at water temperatures of 21 to 28 °C and salinities of 31 to 48‰ during April through June (Pulich 1985).
Water nymph *Najas guadalupensis*	0	10	0 to 0.17	Laboratory growth experiments (Haller 1974).
	0	9	3	Reported for Chesapeake Bay area populations (Stevenson and Confer 1978).
Emergent Marsh Plants				
Smooth cordgrass *Spartina alterniflora*	6	34	19.92 ± 8.06 (n = 61)	Salinity range of bay waters for highly productive Matagorda Island salt marshes from 1978 to 1988 (Pulich 1990).
	2	28	12.12 ± 6.88 (n = 53)	Salinity range of bay waters for extensive Aransas Wildlife Refuge salt marshes from 1978 to 1988 (Pulich 1990).
			15.19 ± 7.78 (n = 86)	Mean water column salinity from statewide coastal marsh survey of Louisiana done in August 1968 (Chabreck 1972).
			10.10 ± 4.38 (n = 95)	Mean soil water salinity from Chabreck (1972).
	0	50	5 to 10	Plant growth experiments under greenhouse conditions at 26 to 30 °C (Mooring et al. 1971; Seneca 1974).
Perennial glasswort *Salicornia virginica*	0	35	0 to 16	Results of one-month seed germination tests in laboratory using sea salt solutions. Plant material studied from southern California (Zedler and Beare 1986).
	<10	>40	>20	In three southern California marshes, flooding of previously high-salinity soils with fresh water caused decline or elimination of *Salicornia* during several months (Zedler 1983).
	7	41	10 to 37	Laboratory observations on shoot growth of San Francisco Bay plants between temperatures of 25 to 27 °C (Mahall and Park 1976).
	24.5	89		Salinity calculated from soil conductivity measured in San Diego, California, marsh. Based on 34‰ salinity equivalent to 15 mmhos/cm conductivity (Zedler 1982).
Black mangrove *Avicennia germinans*			25.72 ± 13.65 (n = 6)	Overlying water column salinity (Chabreck 1972).
			11.94 ± 6.07 (n = 8)	Soil water salinity (Chabreck 1972).
	0	>50	ca 9 to 10	Plants in 1/4 strength seawater produced more leaves and roots than plants in undiluted seawater (Waisel 1972).

(continued)

Table 5.2.2. Continued from previous page.

Group/species	Limits		Optimum (‰)	Remarks
	min. (‰)	max. (‰)		
Saltmarsh bulrush *Scirpus maritimus* (syn. *robustus*)	0	10	0 to 5	Results of one-month seed germination tests in laboratory using sea salt solutions. Plant material studied from southern California (Zedler and Beare 1986).
	14	32	<26	Seed production greatest at this salinity range and submergence period at least six months (50% of year) (Josselyn 1983).
	5	25		Good growth of field populations in San Francisco Bay marshes (Josselyn 1983).
	0	28	4.39 ± 7.0 (n = 24)	Mean calculated during five-year period (1983 to 1988) for Guadalupe Delta area, Texas, populations (Pulich 1990).
			8.90 ± 5.30 (n = 61)	Overlying water column salinity (Chabreck 1972).
			6.78 ± 3.59 (n = 68)	Soil water salinity (from Chabreck 1972).
Saltgrass *Distichlis spicata*			13.32 ± 6.70 (n = 80)	Overlying water salinity (Chabreck 1972).
			8.81 ± 4.03 (n = 94)	Soil water salinity (from Chabreck 1972).
	22.6	45.2		Soil salinity calculated from conductivity measurements in San Francisco area marsh (Josselyn 1983). Based on 15 mmhos/cm equivalent to 34‰.
Needlerush *Juncus roemerianus*	0	30	0	Growth (leaf elongation) experiments during three months under Mississippi summer greenhouse conditions and continuous salinity of overlying water (Eleuterius 1984).
	5	30		Soil salinity measured in lush, healthy Mississippi field populations (Eleuterius 1984).
	30	300		Soil salinity measured in dwarf plant populations under stress (Eleuterius 1984).
			13.89 ± 8.27 (n = 63)	Water column salinity (Chabreck 1972).
			9.20 ± 4.33 (n = 70)	Soil water salinity from Chabreck (1972) Louisiana coastal marsh survey.
Common reed *Phragmites australis*			3.33 ± 3.96 (n = 29)	Water column salinity (Chabreck 1972).
			3.62 ± 3.68 (n = 35)	Soil water salinity from Chabreck (1972) Louisiana coastal marsh survey.
	0	28	4.39 ± 7.0 (n = 24)	Mean calculated during five-year period for (1983 to 1989) for Guadalupe Delta area, Texas (Pulich 1990).
Saltwort *Batis maritima*			23.60 ± 9.97 (n = 23)	Overlying water salinity (Chabreck 1972).
			10.55 ± 6.38 (n = 16)	Soil salinity (Chabreck 1972).

(continued)

Table 5.2.2. Concluded.

Group/species	Limits min. (‰)	 max. (‰)	Optimum (‰)	Remarks
	5	50	10 to 34	Adams (1963).
Marsh-hay cordgrass *Spartina patens*	0	40	5	Range of salinity for plant growth of field populations (Seneca 1974).
			5 to 10	Optimum salinity for seed germination (Mooring et al. 1971).
			8.55 ± 6.33 (n =173)	Water column salinity (Chabreck 1972).
			6.81 ± 4.04 (n = 190)	Soil water salinity (Chabreck 1972).
Bulrush *Scirpus californicus*			1.63 ± 1.22 (n=20)	Water column salinity (Chabreck 1972).
			2.35 ± 1.28 (n=21)	Soil water salinity (Chabreck 1972).
Arrowhead *Sagittaria falcata*			1.70 ± 1.59 (n = 64)	Water column salinity (Chabreck 1972).
			1.79 ± 1.49 (n = 71)	Soil water salinity (Chabreck 1972).
Water hyacinth *Eichhornia crassipes*			0.37 ± 0.10 (n = 4)	Overlying water salinity (Chabreck 1972).
			0.68 ± 0.44 (n = 7)	Soil water salinity (Chabreck 1972).
Narrow leaf cattail *Typha angustifolia*	11.3	22.4		Soil water salinities measured for San Francisco area populations. Based on 15 mmhos/cm equivalent to 34‰ salinity (Josselyn 1983).
			3.93 ± 4.06 (n = 50)	Overlying water salinity (from Chabreck 1972).
			3.64 ± 3.65 (n = 56)	Soil water salinity (Chabreck 1972).
var. *domingensis*	0	10	0	Results of one-month seed germination tests in laboratory using sea salt solutions. Plant material studied from southern California (Zedler and Beare 1986).
	0	10	0 to 3	Based on soil water measurements in San Diego, California, marsh after flooding with fresh water. *Typha* production (biomass) was greater at 3‰ salinity compared to plants growing at 2 to 10‰ salinity regime (Zedler 1983).

Table 5.2.3. Some indicator plants in Texas estuarine submerged communities and tolerance ranges for environmental factors (Pulich 1990).

Environmental factor	Submerged community		
	Water nymph	Widgeongrass	Shoalgrass
Geographic region	Delta	Delta to lower bay	Middle to lower bay
Salinity regime			
Optimum salinity [a]	0 - 4	0 - 25	20 - 40
Salinity range [a]	0 -10	0 - 60	6 - 60
Water clarity	Moderate	High	Moderate to high
Substrate type	Mud to sandy mud	Muddy sand to sand	Muddy sand to sand
Nutrient loading	High	Moderate	Low
River inflow	High	Moderate	Low

[a] Salinity given in ‰.

the soil or overlying water media (Flowers et al. 1977). Response to salinity, therefore, greatly depends on the variation in salinity between the two environments.

As previously mentioned, ecological tolerance limits of estuarine flora will reflect the interaction between combinations of growth factors. Examples of the range of environments inhabited by submerged and emergent estuarine communities and their indicator species have been compiled and compared in Tables 5.2.3 and 5.2.4. As these environmental factors, especially hydrologic conditions, change in various parts of the estuary, species composition of the plant community can change according to each species' tolerance limits and preferred environmental regimes. Usually, spe-

Table 5.2.4. Some indicator species in Texas estuarine marsh communities and tolerance ranges for environmental factors (Pulich 1990).

Environmental factor	Estuarine marsh community			
	Arrowhead	Saltmarsh bulrush	Needlerush	Smooth cordgrass
Geographic region	Delta bay	Delta and upper bay	Delta and upper bay	Upper to lower
Inundation frequency [a]	90 - 100	<10	<8	15 - 98
Salinity regime				
Optimum salinity [b]	<3	1 - 10	1 - 20	5 - 25
Salinity range [b]	0 - 10	0 - 25	1 - 30	1 - 40
Marsh elevation [c]	Low <0.3	High >0.3	High >0.3	Low <0.5
Nutrient loading	High	Moderate	Moderate	Moderate
River inflow	High	Moderate	Moderate	Moderate to low

[a] Percent of the year inundated by tidal action.

[b] Salinity given in ‰.

[c] Elevation in m relative to mean sea level.

cies replacement in a community occurs gradually unless an episodic event occurs; eventually, an entire community may be replaced (e.g., salt marsh may displace brackish marsh). For certain species to become dominant, favorable inflow conditions need to occur in a bay system over a substantial time period since plants, being immobile, become established and proliferate over a season or more (Correll and Correll 1975; Cowardin et al. 1979; Zedler 1983). Such estuarine plant communities thus represent time-integrated indicators of inflow conditions during longer periods of several months.

Plant Distribution and Productivity Studies

A limited number of baseline studies have verified species distribution and wetland community associations for actual Texas coastal areas, but none has been as complete or extensive as the model by Chabreck (1972) for Louisiana wetlands. Such data are extremely important for assessing the effects of altered freshwater inflow or other impacts on vegetated habitats of bays and estuaries.

Mapping studies. Field surveys by the TPWD in the 1960's and early 1970's (McMahan 1966; West 1971, 1972, 1973; C. E. Bryan, Coastal Fisheries Division, pers. comm.) were conducted to map the coastwide distribution of seagrasses (submerged grassbeds) and coastal marshes. These maps were later used by Diener (1975) to compile the acreage figures for coastal habitats published in his report. Later studies by Benton et al. (1977, 1979) for the TDWR demonstrated the application of color infrared aerial photography at a scale of 1:40,000 to map wetland vegetation distribution. Species composition was verified by extensive ground truth. The Benton project examined seasonal changes in wetland communities at selected sites in all Texas estuaries and produced a series of detailed vegetational community maps for these areas for the mid-1970's.

The U.S. Fish and Wildlife Service (USFWS) National Wetland Inventory Program has mapped wetlands in Texas from 1956 aerial photography. Coastal marsh habitat, estimated in 1956 at 379,361 ha (937,400 acres) (Shaw and Fredine 1956), serves as a baseline for comparison with two recent studies. TPWD determined coastal wetlands during the mid-1970's from Landsat imagery (TPWD 1988). Their figure of 247,576 ha (611,760 acres) is somewhat high due to inclusion of undifferentiated rice fields in the area. The USFWS National Wetland Inventory has initiated, but not completed, an inventory of coastal marshes from 1979 color infrared photography. However, the preliminary National Wetland Inventory Program results for this time period appear to be significantly lower than the TPWD area above (USFWS 1987). Neither study delineated submerged vegetation. Because of inconsistencies between various workers in wetland classification techniques, areas of wetlands listed by Diener (1975) for the early 1970's cannot be satisfactorily compared with the USFWS National Wetland Inventory or the TPWD inventory. This illustrates the need to standardize mapping protocols and vegetation classification techniques in order to perform wetland change analysis. If a figure of 242,817 ha (600,000) acres is used from the TPWD study, coastal marshes in Texas have decreased by more than 35% between the mid-1950's and mid-1970's.

Coastal studies during the 1980's by W. A. White and colleagues at the Bureau of Economic Geology, University of Texas at Austin, provide another basis for a comprehensive, recent inventory of wetlands in the seven Texas estuaries. Distribution of plant communities was mapped based on color infrared NASA photography (scale 1:65,000) taken in November 1979. Six atlases produced by the University of Texas project include detailed wetland maps, and discussion of plant relationships to estuarine zones and hydrologic regimes (White et al. 1983, 1985, 1986, 1987, 1988, 1989). The ground truth and species distribution work encompassed in these reports constitutes the most accurate description of wetland plant associations for the entire Texas coast to date. However, quantitative analysis of the wetland maps from these reports remains incomplete. A major need exists to calculate the total acreage of the various wetland habitats mapped in each estuary for this project.

Production dynamics. Hoese (1960) reported on the biotic changes in Mesquite Bay (lower part of Guadalupe Estuary) associated with the end of the most severe drought in Texas history. The flora and fauna of the high salinity period in 1956 to early 1957 were compared with those of the low salinity period after heavy rainfall in the spring of 1957. Hoese noted a complete change in the submerged vegetative communities, from shoalgrass [*Halodule* (=*Diplanthera*) *wrightii*] during high salinity (26 to 45‰) to widgeongrass (*Ruppia maritima*) during the low-salinity post-drought period (0.5 to 18‰). This replacement occurred during six months in 1957 and demonstrated the significant effect of salinity regimes on regulating populations of these submerged grasses.

An extensive study was conducted on the Guadalupe Estuary by Childress et al. (1975) for the period 1971 to 1974; this estuary is influenced by inflows from the Guadalupe and San Antonio rivers. While the study dealt primarily with abundance of fishery species and relationships to freshwater inflow parameters such as salinity, turbidity, and nutrient loadings, some information was provided on seasonal status of vegetated habitats and potential contribution of freshwater inflow to plant productivity. Significant observations included the widespread dominance of two classical higher salinity species, smooth cordgrass and shoalgrass, around much of the San Antonio Bay shoreline, and the common reed (*Phragmites australis*) in the brackish river delta. During the study period, the overall bay environment showed an average annual salinity in the range of 1.6 to 9.5‰ for the upper bay, 4.1 to 13.0‰ for the middle bay, and 7.4 to 19.1‰ for the lower bay. Childress et al. inferred from their data that a major decrease in the quantity or timing of freshwater inflows from an annual gaged minimum of 1.6 million acre-ft would cause a major alteration in the estuary's ecology.

In work on the Nueces Delta in the Corpus Christi area, Henley and Rauschuber (1978) determined zonation of various marsh plants with respect to elevation and salinity, and measured net above-ground primary production. Production was monitored during 1977 along transects through the marsh using methods that sum biomass changes during different seasons. The productivity and distribution information (Table 5.2.5) was used in conjunction with published data to assess effects of freshwater inflow alterations on plant habitats in Nueces and Corpus Christi bays.

Table 5.2.5. Elevation and annual production data for saltmarsh plants in the Nueces Bay Delta during 1977. Summarized from Henley and Rauschuber (1978).

Species	Elevation (ft relative to mean sea level)	Productivity (gm dry matter $m^{-2}yr^{-1}$)
Smooth cordgrass	+0.9	554
Sea oxeye	+2.1	1,405
Glasswort	+2.2	1,369
Saltwort	+2.2	1,383
	+2.7	1,160
Gulf cordgrass	+2.2	2,114
Seashore saltgrass	+2.3	
Annual glasswort	+2.9	
Shoregrass	+2.9	631

Freshwater Inflow-related Factors and Estuarine Marsh Communities

Tables 5.2.3 and 5.2.4 list the major factors regulating distribution and abundance of estuarine plants (viz., elevation, inundation frequency, salinity regime, and nutrient and sediment loadings). Mechanisms by which these factors influence plants can be complicated when freshwater inflow also varies. A legitimate question is: which of these factors constitutes the primary parameter to key on, if freshwater inflows to the estuary are to be managed for plant habitat? To answer this, we need to consider the growth requirements of wetland plant species and then correlate plant community dynamics under varying inflows with changes in environmental factors.

Contributions of inundation and salinity. Deltaic marsh inundation was used as a key criterion for evaluating estuarine productivity in the previous Bays and Estuaries Program (TDWR 1982c). The main emphasis, however, was on transport of basic nutrients and organic matter to the rest of the estuary (i.e., export of deltaic materials). Correlation analysis was used to define the freshwater inflow needed in terms of river inflows at the most downstream stream gage in the river basins. Streamflows during the historical period 1941 to 1976 were analyzed and the annual frequencies of deltaic marsh inundation from riverine flooding were calculated. As shown in Table 5.2.6, two to three annual delta inundation events were computed to be necessary for sustaining productivity and maintaining suitable salinity regimes in each of the seven Texas estuaries. Table 5.2.6 also contains corresponding data on the amount of inflow to the estuaries required to produce the inundation and salinity regimes. It is interesting that, although the absolute amount of inflow to each estuary varied, the percent of total annual inflow required for delta inundation was between 56 and 69% for almost all the estuaries. However, cause and effect relationships between environmental factors and wetland vegetational communities were not addressed in these studies.

Comparison of the Henley and Rauschuber (1978) data in Table 5.2.5 shows that production of smooth cordgrass in the Nueces Delta was relatively low compared to the more salt-tolerant, high-marsh species such as saltwort, glasswort, sea ox-eye, and Gulf cordgrass (*Spartina spartinae*). This correlated with the wide range of salinity regimes approaching hypersaline levels and semiarid conditions occurring in this estuary. Henley and Rauschuber (1978) concluded that decreasing inflow from the Nueces River would increase the frequency of high salinity in Nueces Bay compared to the historical period 1941 to 1975. Salinities in the lower bay system, Corpus Christi Bay, were not predicted to increase proportionately due to the larger bay volume and mixing with Gulf of Mexico water. The higher salinities resulting from reduced freshwater inflows were predicted to affect the Nueces Delta marsh by shifting the species composition from less tolerant forms (smooth cordgrass and bulrush) to those more tolerant of salinity (saltwort and glasswort).

Table 5.2.6. Inflow statistics for delta and upper estuary of seven major Texas river drainages based on 1941 to 1976 historical records (TDWR 1982c).

Basin	Median annual delta inundations	Seasonal salinity regime in upper estuary (‰) Jan-Mar	Apr-Jun	Jul-Sept	Oct-Dec	Annual inflow (10^6 acre-ft) for delta inundation	Percent annual inflow needed for inundation
Sabine-Neches	3	10.0	3.5	12.0	9.3	5.68	—
Trinity-San Jacinto	3	10.0	3.0	11.5	8.8	3.17	61
Lavaca-Colorado	3	12.3	9.0	13.7	11.8	0.35	57
Guadalupe	3	18.3	13.3	18.3	17.6	1.24	69
Mission-Aransas		15.3	12.7	15.7	14.0	0.015	18
Nueces	2	26.0	16.0	16.0	19.3	0.356	62
Laguna Madre		40.0	34.0	35.0	34.7	0.177	56

Inundation frequency and water budget of the estuary. Relationships of physical factors (including climate and geomorphology) to estuarine vegetation distribution were analyzed by Deegan et al. (1986) for Gulf of Mexico (including Texas) estuaries. This study concluded that the types and areal extents of vegetated habitats were directly related to the intertidal area and the water budget of the estuary. Intertidal area was determined by oceanic tidal range and geomorphology of the shoreline area, while water budget was controlled by climate and freshwater input. The study distinguished between the effects of rainfall and river flow for providing freshwater input. For emergent marsh systems, Deegan et al. concluded that areal extent was directly dependent on rainfall, and not river discharge. Regardless of the freshwater source, however, freshwater input was a major prerequisite for marsh production in all estuaries.

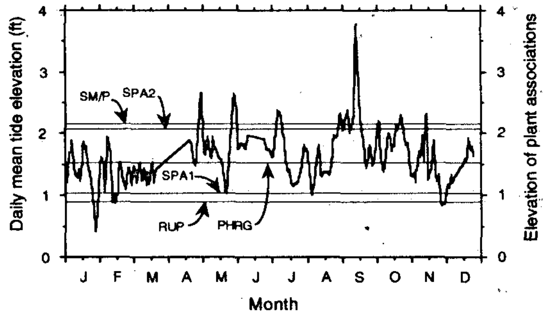


Figure 5.2.1. Water level record for upper San Antonio Bay near Seadrift during 1988. Elevations of plant associations at Guadalupe Delta study sites are superimposed on the daily mean tidal height. RUP = shallow *Ruppia*; SPA 1 = deepwater *Spartina alterniflora*; PHRGM = deepwater *Phragmites australis*; SPA 2 = upper *Sp. alterniflora*; SM/P = deepwater *Scirpus maritimus-Spartina patens*.

Pulich (1990) reviewed available data for San Antonio Bay and concluded that inundation frequency (i.e., hydroperiod) was the fundamental prerequisite for maintenance of aquatic plant communities. The exact species composition of the communities in turn varied with fluctuations in salinity and probably nutrient regimes of the flood waters. While saltwater-tolerant, euryhaline species such as smooth cordgrass or glasswort found in the middle and lower estuary received adequate inundation from daily tidal fluctuations, the brackish species in the upper bay and Guadalupe Delta areas (e.g., common reed, bulrush, and marsh-hay cordgrass) showed stringent requirements for inundation with low-salinity waters (Figure 5.2.1). The actual inundation duration experienced by the common reed, which occurs at the lowest elevation of 0.5 m (1.7 ft.), was 2 to 32% per year. The inundation duration for bulrush and marsh hay cordgrass, which occur at a higher elevation of 1.1 m (3.6 ft.), was only 1 to 10% per year (Figure 5.2.1). The appropriate water levels in these river delta communities were maintained primarily by seasonal inflow events in conjunction with spring and early fall high tidal stands. Unless suitable hydric soils were produced by riverine flooding at these critical times, moisture requirements of these brackish water plants would not be satisfied and the plants would eventually undergo desiccation.

Normally, salinity is at low levels (0 to 10‰) in waters surrounding the delta region when this seasonal flooding occurs (Pulich 1990). This is due to the prevailing pattern of high precipitation during spring and fall months in coastal Texas. If low inflows are occurring in the estuary due to drought, excessive upstream impoundment, or large withdrawals of river water, then salinity in the upper estuary may be inordinately high (greater than 10‰ to as high as 30‰). When inundation of the delta occurs at these times, these saline waters may spread into normally brackish or freshwater areas. As soils dehydrate, soil salinity will build up, subjecting the salt-sensitive marsh plants to osmotic stress. If flushing by rainfall runoff does not subsequently occur, the dry, saline soils will become unfavorable for brackish or freshwater marsh vegetation. Eventually, these drier areas may be colonized by high (distal) saltmarsh (e.g., Gulf cordgrass) or even upland, semi-terrestrial species such as groundsel bush (*Baccharis*) or salt cedar (*Tamarix*).

Marshes and sediment deposition. The dependence of estuarine marshes on continuous nourishment by riverborne sediments to maintain their elevation, and consequently their productivity, has been variously discussed by Shepard and Moore (1960), Redfield (1972), White and Morton (1987), and White and Calnan (1990a, 1990b). Generally, the concepts involved should be considered in a long-term historical time frame and from a geological perspective. The mechanisms and scenarios by which freshwater inflows contribute to sedimentary accretion in Texas estuaries, primarily in the river delta areas, are described at length in Section 4.4.

Sediment depositional processes appear especially critical to marsh areas experiencing erosion or relative rise in sea level from subsidence. Disturbance of these processes is suggested as the cause of current extensive loss of fresh and brackish marshes in Louisiana (Gagliano et al. 1981) and along the upper Texas coast in the Sabine-Neches and Trinity-San Jacinto estuaries (White and Calnan 1990b). In addition, White and Calnan (1990a) have discussed historical losses of emergent wetlands in the Nueces, Guadalupe, and Lavaca river deltas of the central coast in relation to supply of sediment from upstream sources and the possible contribution of upstream reservoirs to this problem.

While subsidence or erosion may be the ultimate causes of such wetland loss, the corresponding causes of subsidence and/or erosion are often difficult to establish and relate to freshwater inflows. High rates of subsidence in the Houston area during the 1960's and 1970's have been attributed primarily to extensive removal of ground fluids (viz., petroleum and groundwater) according to Swanson and Thurlow (1973) and Gabrysch (1984). Such factors that contribute to compactional subsidence in estuaries must be differentiated from loss of sediments due to decreased loads in freshwater inflows. Suffice it to say that, as subsidence and erosion processes increase, stress effects on wetland vegetation will intensify. The primary results will be exposure of fresh or brackish marshes to higher salinity waters as a result of saltwater intrusion, and submergence of marshes as they are exposed to consistently longer inundation periods or more frequent flooding.

Saltwater intrusion processes may also be aggravated by canals and channels constructed in estuarine areas. These structures allow for rapid exchange of saltwater into the interior of coastal, low-salinity marshes. Reduced freshwater inflow may reverse the hydraulic head of fresh water within delta areas, such that seawater flows faster into the freshwater interior via channels. Such channelization has exacerbated the loss of low-salinity marshes in Louisiana (Salinas et al. 1986).

Submerged Vascular Vegetation and Freshwater Inflows

Species of submerged vascular plants occupy different zones of Texas estuaries depending on salinity tolerance limits and light requirements. From Table 5.2.3, three distinct groups of these plants are evident (Pulich 1990). The first includes the marine seagrasses such as shoalgrass and turtlegrass (*Thalassia testudinum*), which are obligate halophytes and, with one exception, the most salt-tolerant submerged species. They are restricted to estuarine regions with salinity regimes of 20‰ or higher average annual salinity.

The second group consists of widgeongrass, a euryhaline species that is known to thrive in waters over a wide range of salinities, from low oligohaline to hypersaline conditions (Phillips 1960; Den Hartog 1970; Pulich 1980, 1985). Pulich (1985) documented the growth dynamics of this species in a monotypic grassbed in Laguna Madre from salinities between 31 and 48 ppt. Dunton (1990) also substantiated the lack of a direct salinity effect on leaf growth of widgeongrass from in situ studies in San Antonio and Corpus Christi Bays. Over the salinity range from 0 to 38‰, he observed no significant correlation ($P > 0.05$) between widgeongrass shoot production rates and bay water salinities or dissolved inorganic nitrogen. Den Hartog (1970) has concluded that widgeongrass is supplanted by other seagrasses under these high salinity regimes due mainly to its inability to compete with them. In a recent study, competitive interaction between widgeongrass and shoalgrass in moderate or higher salinity waters was attributed to sediment nutritional factors and seasonal competition (Pulich 1989). This widgeongrass generally persists as the dominant species in Texas seagrass beds only where water column salinity averages less than 25‰ during the growing season.

The true freshwater species form a third group that can tolerate only low-brackish salinity environments. Representative of such oligohaline species, the water nymph (*Najas guadalupensis*) sometimes dominates in the river delta

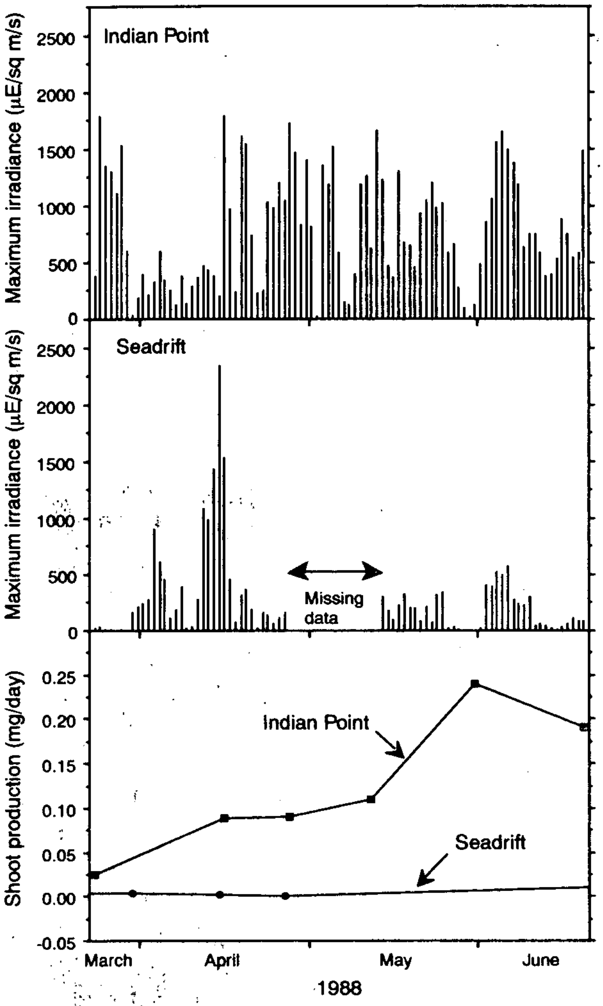


Figure 5.2.2. Relationship between shoot production rate of widgeongrass (*Ruppia maritima*) and corresponding underwater quantum irradiance regimes (in µEinsteins) found at study sites in two south Texas bays. Seadrift is in upper San Antonio Bay, while Indian Point divides Nueces and upper Corpus Christi bays. Data from Dunton (1989).

regions of Texas estuaries, if average salinities remain at less than 4‰ during the spring and summer.

Underwater light requirements. Seagrasses and oligohaline submerged vegetation, unlike emergent plants, must cope with another limiting factor, water clarity (Table 5.2.3). Because of their growth requirement for continuous submergence in estuarine waters (Figure 5.2.1), production of these rooted plants is critically dependent on the amount of available sunlight transmitted through the water. Thus, growth occurs only at those shallow depths, usually around the bay shoreline, where water clarity is sufficient for photosynthesis and other light-driven plant processes.

Sunlight is attenuated (i.e., decreased) through the water column due to absorption, reflection, and scattering from suspended sediments, plankton, microorganisms, and detritus particles. Taken together, these materials comprise turbidity, or the amount of suspended particulate matter. Where turbidity is high, light transmittance (i.e., water clarity) is low (Levinton 1982). Usually, turbidity is highest in an estuary where riverine discharges containing suspended particulate loads meet and mix with tidal currents (e.g., in the upper estuary) (Kennish 1986). Most Texas estuarine waters experience fairly turbid conditions due to the predominantly muddy bay sediments and prevailing strong southeasterly winds (Shepard and Moore 1960).

Studies indicate that light saturation of shoalgrass photosynthesis occurs at a fairly high level, around 10 to 15% of full summer sunlight (Beer and Waisel 1979; Williams and McRoy 1982). Shoalgrass light requirements are significantly higher than those of other aquatic plants including phytoplankton and benthic algae (0.5 to 3.0%) of terrestrial plants from shade environments (0.5 to 2.0%) (Stevenson 1988). These high light levels occur in shallow waters down to depths of only 0.3 to 0.6 m (1 to 2 ft) in most Texas bays (Pulich 1980; Dunton 1989, 1990). The light levels reflect the normally high turbidity which varies both seasonally and geographically throughout estuaries (Pulich and White 1989; Pulich 1990). Dunton (1989) demonstrated the overriding influence of underwater light regimes to the limitation of widgeongrass and shoalgrass production in both San Antonio and Nueces estuaries. Figure 5.2.2 compares the dramatic differences in water column irradiance between the two estuaries. Higher underwater light levels in Nueces Bay (Indian Point) compared to San Antonio Bay (Seadrift) directly correlate with the seasonal differences observed in productivity of widgeongrass. Regression analysis on Dunton's data gave a correlation coefficient (R value) of 0.686 (P = 0.01) between shoot growth rate and the mean, maximum daily underwater irradiance during the shoot growth period.

There is some indication that light requirements for seagrass leaf photosynthesis are in fact different from light requirements for sustained whole plant growth. Recently, Duarte (1991) compiled data on minimal light requirements of Gulf of Mexico seagrasses which were based on the maximum reported depth limits for their survival. Values ranged from 8.2% light (0.7 m depth) for widgeongrass, 15.3% light (7.5 m depth) for turtlegrass, 17.2% light (1.9 m depth) for shoalgrass, 19.2% light (16.5 m depth) for manateegrass, to 23.7% light (14.4 m depth) for clovergrass (*Halophila*). It is apparent that, although widgeongrass does not survive in waters as deep as shoalgrass can (0.7 m vs. 1.9 m depth), widgeongrass still does not have as large a minimal light requirement as the latter. Those relationships probably reflect differences between species in root tolerance to sediment factors or in respiratory capacity (Pulich 1989).

Substrate requirements. Water clarity often influences submerged vegetation production in combination with substrate stability. The herbaceous roots of submerged vegetation function in nutrient absorption and anchoring of plants on preferred muddy to fine sand sediments. This physical requirement is manifested by the occurrence of plants in shallow, relatively protected waters on muddy to sandy sediments (Pulich 1989, 1990). If wave energy or water currents are excessive, these soft bottom sediments can be eroded and resuspended. Resuspension leads to higher turbidity and physical uprooting or burial of plants. Since delta and upper estuary regions often have highly unconsolidated sediments around the bay margins (White and Morton 1987), submerged vegetation in these regions is particularly sensitive to high inflows and currents that produce turbidity and shifting bay-bottom topography. The bay margins in the middle and lower parts of the estuary, conversely, normally have sandier sediments with less mud, making them less susceptible to erosion or resuspension. This provides a stable substrate where submerged vegetation can establish permanent dense meadows.

Coastwide distribution of submerged vegetation. The environmental tolerance limits for submerged vegetation act in concert to control the coastwide distribution patterns, especially for shoalgrass and turtlegrass. The decreasing abundance of seagrasses going northward in Texas estuaries (Diener 1975) best reflects the combined lower salinity, higher turbidity, and slightly cooler temperature regimes existing on the upper coast compared to the lower coast. Monthly mean salinity records shown in Figure 4.1.17 indicate that Sabine, Trinity, East Bay, Upper Matagorda, and Lavaca bays have experienced monthly salinities consistently less than 20‰ during the 20-year period 1968 to 1987. These oligohaline to mesohaline salinities, superimposed on an overall cooler temperature regime, pose unfavorable conditions that prevent the subtropical seagrass species from becoming established in these areas. Conversely, the salinities are too high to support any fresh-to-brackish water species except widgeongrass. Thus, widgeongrass is the only species capable of surviving in these bay areas. Its distribution often appears limited by habitat requirements including sediment organic properties and hydrodynamic regimes (Pulich 1989; Dunton 1990).

Impacts in the Trinity-San Jacinto Estuary. The Trinity-San Jacinto Estuary presents an interesting case study of changes in submerged vegetative habitat. Pulich and White (1989) examined the chronology of submerged vegetation decline since 1958 for both Trinity and West bays and correlated this with corresponding impacts from physical processes. They concluded that water column salinity and turbidity changes probably had not directly caused widgeongrass in Trinity Bay and shoalgrass in West Bay to disappear between 1958 and the early 1980's. Rather, other factors unrelated to freshwater inflows were hypothesized to be responsible for the vegetation losses: human-induced land subsidence and relative sea-level rise; sediment erosion and wave energy during Hurricane Carla in 1961; and dredging, nutrient-loading, and toxin input from onshore developments. Thus, changes in estuarine wetlands must carefully distinguish between effects of altered freshwater inflows and effects of other coastal processes.

Epiphyte effects. Epiphyte populations and macroalgae (seaweeds) represent a major non-vascular plant component of submerged vegetative habitat. Epiphytic algae, which grows attached to submerged structures like seagrass leaves, is a highly nutritious and preferred food for herbivorous fish and crustaceans in the bays. Dunton (1989) showed that they contribute 30 to 80% of the total biomass of seagrass beds in San Antonio and Corpus Christi bays.

Although serving as quality food and habitat in estuarine systems, epiphytes themselves reduce the underwater light available for submerged vascular plants by covering and fouling the leaf surfaces. Water conditions that enhance production of epiphytes would concurrently inhibit submerged vegetation production through this shading effect (Sand-Jensen 1977; Pulich 1980). Results from Dunton (1990) suggest that the presence of possibly more noxious, green algal epiphyte populations may have contributed to less widgeongrass productivity in San Antonio Bay than in Corpus Christi Bay. Under the lower salinity and higher freshwater inflow regimes, diatoms and green algae *(Cadophora* and *Enteromorpha)* overgrew the widgeongrass leaves and formed floating mats in San Antonio Bay. In the higher salinity and lower inflows of Corpus Christi Bay, green algae were noticeably absent and epiphytes consisted mainly of red algae (*Polysiphonia* and *Gracilaria*) and diatoms. In this case, widgeongrass productivity was not as greatly affected by shading. Further analysis is necessary to clarify these relationships between freshwater inflow, nutrient loading, epiphyte abundance, and submerged vegetation production.

Conclusions

Texas estuaries have evolved characteristic vascular plant communities in accordance with the decreasing gradient in precipitation from north to south that controls freshwater inflows. The dominance of habitat types reflects the combined influence of basic physical and hydrological parameters, including coastline geomorphology, inundation and salinity regimes, and nutrient loading. Freshwater inflows operate through these different factors to affect plant production depending on the habitat type. Effects of altered freshwater inflow on estuarine plant communities reflect the growth requirements and environmental tolerance limits of the individual species present. In any particular bay, species composition of vascular plant communities changes in response to moderately long periods of altered freshwater inflow, normally a minimum growing season of several months. Thus, vegetation communities integrate salinity, nutrient, and sedimentation processes over time.

While many saltmarsh plants and true seagrasses have fairly high salinity tolerances (from less than 10 to 36‰), most brackish marsh and submerged vegetative communities in the upper estuary and delta regions show critical dependence on low-salinity (i.e., oligohaline to mesohaline) conditions. Consequently, the basic moisture requirements of these upper-bay aquatic plants must be satisfied by inundation with low-salinity waters (0 to 10‰) during freshwater inflow events. On the average, three to four delta inundations have occurred each year during the last 40-year period to provide these conditions in estuaries of the upper and central Texas coast. Middle- to lower-bay marshes containing smooth cordgrass receive sufficient saltwater from daily to seasonal tidal inundations. For all emergent marshes, these inundations also provide nutrients, organic matter (detritus), and sediments to stimulate the plant communities; however, little information is available documenting combined effects of these three factors on interspecific plant competition processes.

Submerged vegetation in most of the estuary except the delta is tolerant of a wide range of salinities, from about 5 to 36‰. This is exemplified by the dynamic interaction between widgeongrass and shoalgrass, whereby one species can replace the other depending on prolonged salinity regimes. The 20‰ isohaline seems to control distribution of the two species. Delta brackish submerged vegetation is effectively restricted to salinities of less than 4‰. The overriding factors that affect the density (as distinct from distribution) of submerged vegetation are turbidity and nutrient loading of freshwater inflow. These factors control underwater light availability and epiphyte populations. Submerged vegetation demonstrates minimum requirements for light levels generally found only in shallow zones of Texas bays. Freshwater inflow may exacerbate turbidity levels in the estuary by contributing discharges with high suspended particulate loads. Input of high dissolved nutrient loads may also lead to production of excessive phytoplankton blooms or algal epiphytes on submerged vegetation leaves, which decreases vascular plant productivity through leaf shading and fouling. However, additional information is necessary to predict submerged vegetation responses to combined effects of turbidity and nutrient components of freshwater inflows.

In addition to altering estuarine water budgets and salinity regimes, reservoirs and upstream diversions of fresh water reduce the loading of sediment and organic matter to estuaries. These reductions can lead to submergence and permanent loss of vegetative habitat. Without deposition of riverborne sediments and upland, detrital material, coastal wetlands may be deprived of the proper substrata and elevations with respect to sea level. While these factors normally affect long-term maintenance of the wetland communities, their deprivation may become noticeable over relatively short time periods in areas where land subsidence or coastal erosion also are occurring. Upper Texas coast brackish marshes in particular show serious stress and deterioration from deficiency of these components of freshwater inflows.

5.3 INFLUENCE OF FRESHWATER INFLOW ON ZOOPLANKTON

Introduction

Characteristics of zooplankton. Zooplankton are microscopic animals ranging in size from 0.02 to 2 mm (0.0008 to 0.08 inches). They are weak swimmers, largely transported by currents in their environment. Some zooplankton (holoplankton) spend their entire life cycle as members of the plankton, remaining within the size range described above. Other species are temporary residents (meroplankton), spending only part of their life cycle as eggs and larvae in the zooplankton, and then leaving to settle as benthic organisms or to develop and grow into much larger animals such as fish and shellfish.

Role of zooplankton in estuaries. In most aquatic systems, zooplankton are the principal consumers of carbon fixed in the aquatic environment by phytoplankton; they also feed on microorganisms, other zooplankton, and organic matter imported from adjacent terrestrial or aquatic habitats. As part of the food chain, they are preyed on by larger zooplankton as well as by larval and small fish and invertebrates (Govoni et al. 1983; Minello et al. 1987; Steele 1974). Zooplankton provide an essential link in the transfer of food energy from organic matter that is imported or produced in the estuary to sport and commercial species. Thus, they can be considered to be one form of estuarine life on which fish and shellfish are dependent.

Size groups. Zooplankton can be divided into two size groups: microzooplankton are 0.02 to 0.2 mm (0.0008 to 0.008 inches), and macrozooplankton are 0.2 to 2 mm (0.008 to 0.08 inches). While this division is somewhat arbitrary, determined by the mesh size of collecting nets, it

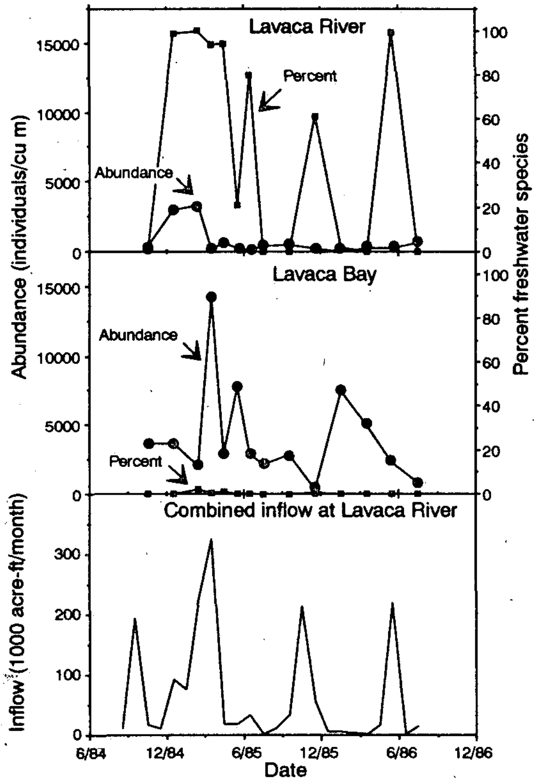


Figure 5.3.1. Displacement of zooplankton by river flows. Top panel shows abundance and percent freshwater species at station 45-1 in the Lavaca River (Jones et al. 1986). The middle panel shows abundance and percent freshwater species at station 85-1 in Lavaca Bay. The lower panel shows the monthly combined inflow from the Lavaca River and ungaged watershed through which the river flows. See Figure 5.3.4 for station locations.

is a convenient partition since some macrozooplankton may feed on microzooplankton.

Typical measurements of zooplankton. Traditionally, zooplankton have been laboriously identified and counted, and the standing crop reported as number of individuals per m^3. Sometimes the totals for macrozooplankton are divided into broad taxonomic groups such as copepods, barnacle larvae, and decapod larvae. This is useful in identifying major abundance changes that may be due to reproductive activities of a few species. A single species of copepod, *Acartia tonsa*, and barnacle larvae often comprise 70 to 85% of the macrozooplankton standing crop in Texas estuaries.

Biomass, the dry weight of zooplankton per m^3, is less frequently reported than standing crop. It is useful in evaluating food webs in estuaries since dry weight can be converted to a carbon basis to evaluate the flow of energy from plants and algae to higher trophic levels.

Day-night differences. While zooplankton are present throughout an estuary, the abundance of various species may differ from place to place in response to local environmental conditions and predation. Many macrozooplankton are strong diurnal migrators that move into the water column at night from the bottom sediments. Buskey (1989) found higher macrozooplankton abundances from night samples than from day samples in the Nueces Estuary. Using log-transformed abundance data, the geometric means of the night and day abundances were 9,720 and 4,047 individuals/m^3; the difference was statistically significant (three-factor ANOVA; $F = 7.21$; $df = 1, 44$; $P < 0.05$). Night biomass was 1.76 to 2.78 times the day biomass.

Buskey's (1989) data from two stations in the Guadalupe Estuary also showed night abundance was greater than day abundance (geometric means of 9,396 versus 2,239 individuals/m^3); the difference was statistically significant (three-factor ANOVA; $F = 9.39$; $df = 1, 20$; $P < 0.01$). Night biomass was 3.37 to 9.49 times the day biomass at the two stations where measurements were taken.

Wind-driven turbulence that resuspends the bottom sediment may reduce differences in abundance between day and night zooplankton samples. Night data are more representative of the total abundance of zooplankton, but most of the historical data were collected during the day. There is little information about mixing conditions that could be used to screen the data so that well-mixed daytime samples could be identified. In analyzing day abundance or biomass, the additional variation due to zooplankton diurnal migration increases the difficulty of drawing conclusions about small differences. Since the bulk of the data were collected during the day, it must be used for analysis even with the additional source of variation. Abundance data that are discussed in the rest of this section are based on day samples.

Generation time. One further complication to simple comparison of zooplankton data is the generation time of zooplankton. The life cycle of macrozooplankton is measured in weeks or months; the life cycle of microzooplankton is measured in hours or days. When the generation time scale is of the same order of magnitude as the duration of major freshwater inflow events, it can be difficult to assess the effect the inflow has on zooplankton populations since they may respond rapidly through reproduction to increased levels of food and decreased predation.

Inflow effects. Zooplankton can be affected by changes in the environment brought about by freshwater inflow variations. Direct effects include physiological adaptation to temperature and salinity alterations; most estuarine zooplankton are remarkably euryhaline. Because they are so small, zooplankton are swept with the currents and their distribution can be influenced by large freshwater inflow volumes. Indirectly, zooplankton may be affected by increases in phytoplankton due to nutrient input with large inflows, or decreases in phytoplankton due to turbidity associated with inflows.

Effects of High Inflows on Zooplankton

Reductions in zooplankton abundance that occur after large inflow events have been interpreted as resulting from physical displacement of the organisms by the large volume of inflowing water (Holland et al. 1975; Gilmore et al. 1976; Kalke 1981; Jones et al. 1986; Armstrong 1987; Buskey 1989). This process is recognized by zooplankton experts and is included in recent models of zooplankton population dynamics (Gaedke 1990). Estuarine zooplankton may be replaced by freshwater species that persist until the salinity of the water increases and the freshwater species die from osmotic effects or predation, or the estuarine zooplankton return in high numbers. Macrozooplankton population changes with time and inflow are shown for two locations in the Lavaca-Colorado Estuary (Figure 5.3.1). The Lavaca River station within the delta is located about 11 km (6.8 mi) from the mouth of the river into the bay and 8 km (5 mi) below Lake Texana (Jones et al. 1986). Large freshwater inflows occurred in October 1984, January through April 1985, November 1985, and June 1986. Except for April 1985, the salinity of the water at this station was less than 1‰ from January through July. Abundance dropped in April 1985 and then remained low throughout the rest of the study; the arithmetic mean abundance during the two-year period was 715 ± 1,051 individuals/m^3. During the January to July period and after other periods of high inflow, freshwater species constituted 60 to 99% of the macrozooplankton. When the inflow returned to lower levels, salinities ranged from 3.3 to 13.4‰ and the abundance of freshwater species decreased to zero. It is clear that the composition and abundance of zooplankton populations in the delta are strongly influenced by freshwater inflow.

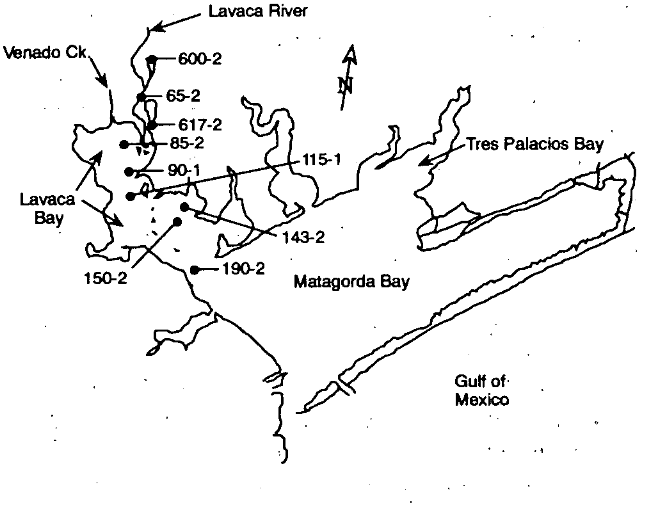


Figure 5.3.2. Zooplankton sample sites for the Lavaca River and Lavaca Bay from Gilmore et al. (1976).

A second station was located in mid-Lavaca Bay, within the influence of the river plume. The effect of large freshwater inflows on salinity at this station during late winter and early spring of 1985 was much smaller than at the fresher Lavaca River station. Large freshwater inflows reduced the salinity at this station to 1.8‰ in April 1985, but salinities during the other sampling periods of this study (November 1984 through August 1986) varied from 3.7 to 23.9‰, a range that is representative of Lavaca Bay conditions during the past 20 years. The abundance of freshwater species remained close to or at zero (Figure 5.3.1), while the arithmetic mean abundance of all macrozooplankton during the two-year study was 4128 ± 3612 individuals/m^3. Mean abundance at this estuarine location was more than five times higher than at the fresher Lavaca River station, and the abundance pattern did not vary consistently with inflow events. At the time of very high inflows in April 1985 (resulting in low macrozooplankton abundance in the Lavaca River), Lavaca Bay macrozooplankton populations increased to more than 14,000 individuals/m^3.

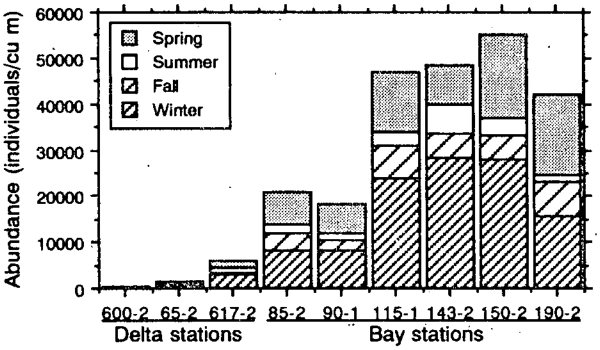


Figure 5.3.3. Seasonal average zooplankton abundance for delta and bay stations from Gilmore et al. (1976). The abundance values are geometric means of the groups.

Environmental Influences on Microzooplankton Abundance

Abundance and biomass. Buskey (1989) compared the abundance and biomass of microzooplankton at four stations along the lengths of the Nueces and Guadalupe estuaries. The stations were roughly comparable with respect to their distance from the river mouths. Inflow during the year of sampling in the Nueces Estuary was very low, but inflow during the Guadalupe sampling period was extremely high, especially in June. Average microzooplankton abundance and biomass at the station closest to the river mouth was slightly higher for the Nueces Estuary than for the Guadalupe Estuary, but abundance and biomass was higher in the Guadalupe Estuary than in the Nueces Estuary for all the other stations. None of the differences were statistically

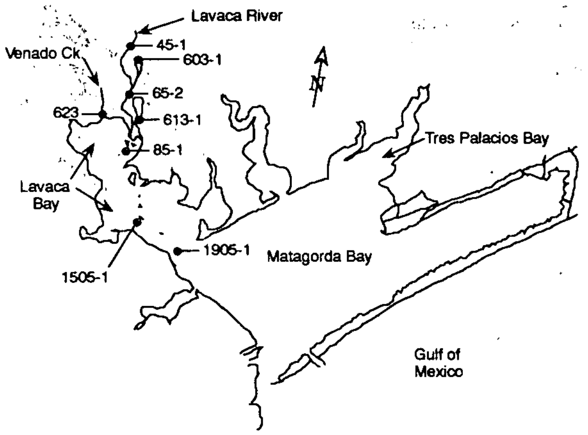


Figure 5.3.4. Zooplankton sample sites for the Lavaca River and Lavaca Bay from Jones et al. (1986).

significant, however. Average abundance ranged from 28.9 to 60.8 million individuals/m^3, and average biomass ranged from 24.6 to 207.2 mg/m^3.

Microzooplankton abundance peaks. Buskey (1989) sampled the microzooplankton in the Nueces Estuary about every two weeks and observed several abundance peaks in the range of 80 to 400 million individuals/m^3. These abundance peaks were unrelated to inflow since the estuary received very little fresh water during the study. In San Antonio Bay, Buskey took samples only six times during the year. The abundance of microzooplankton increased after several months of higher-than-average inflows in the spring. The highest population densities coincided with the period of extreme high inflow in June of 1987; at some stations, abundance was more than 100 million individuals/m^3. Buskey noted that single-celled tintinnids, which are ciliated protozoans, dominated the microzooplankton at this time. He favored the hypothesis that the high freshwater inflows stimulated tintinnid population growth. Generation time for tintinnids is measured in hours or days, so they may reproduce rapidly under favorable conditions.

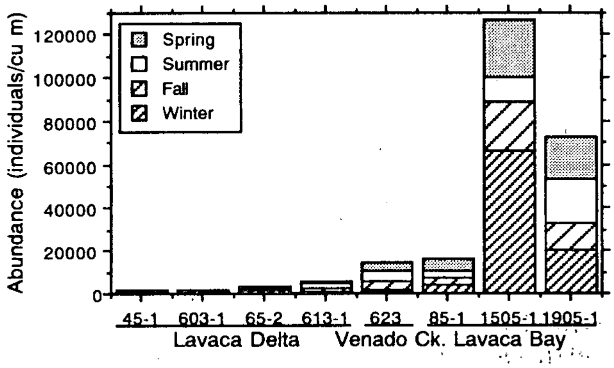


Figure 5.3.5. Seasonal average zooplankton abundance for delta and bay stations from Jones et al. (1986). The abundance values are geometric means of the groups.

Microzooplankton biomass compared to macrozooplankton. Buskey found that microzooplankton abundance in Texas estuaries (30 to 60 million/m^3) was 10 to 100 times as great as in several other shallow marine environments. A recent study of ciliates in Chesapeake Bay (Dolan and Coats 1990) found ciliate densities slightly lower than Buskey's study, 1.8 to 17.2 million individuals/m^3. The biomass of microzooplankton was about the same or slightly greater than the biomass of macrozooplankton in the Guadalupe and Nueces estuaries. This is consistent with the size fractions of phytoplankton measured by Stockwell (1989) in the Nueces Estuary. Microzooplankton graze on nanophytoplankton, which are less than 0.02 mm (0.0008 inches) in diameter. The latter provide the majority (85%) of the phytoplankton photosynthesis and standing crop in the Nueces Estuary (Stockwell 1989). No phytoplankton size measurements are available from other Texas estuaries. The nanophytoplankton-to-microzooplankton food web energy transfer may be the major pathway between photosynthesis and higher trophic levels. It is difficult to say what this means to productivity in Texas estuaries since the studies by Buskey and Stockwell are the first for Texas bays that have measured the different size fractions in the phytoplankton and zooplankton communities.

Environmental Influences on Macrozooplankton Abundance

Zooplankton data sources and analytical methods. Standing crop, the number of organisms present in a cubic meter of water, is the abundance measurement most often reported in zooplankton studies of estuaries. Holland et al. (1973, 1974, and 1975), Matthews et al. (1975), and Gilmore et al. (1976) collected macrozooplankton in the 1970's in the Nueces and Guadalupe estuaries, and in Lavaca Bay and the Lavaca Delta. Armstrong (1987) reviewed these and a few other macrozooplankton studies in a profile of the ecology of open-bay bottoms in Texas. The studies covered periods of time ranging from one to nearly three years. More recent studies by Jones et al. (1986) and Buskey (1989) resampled the Lavaca Bay system (Lavaca, Keller, and Chocolate bays), the Guadalupe Estuary, and the Nueces Estuary a decade later, under different inflow regimes. Study duration, station location, and sampling frequency differed substantially in each of the six studies. The collecting gear was similar (plankton nets with 153 µm mesh) for all studies, so statistical analyses and comparisons of the abundance data were possible.

For each estuary, station location and sampling frequency were different for each pair of studies. By using data from selected stations in comparable areas of the estuary and grouping the data by season, abundance among seasons and station locations could be compared. Since some data sets covered multiple-year periods, it was possible to select an annual data set collected under relatively high inflow conditions and another annual data set collected under relatively low inflow conditions for each estuary. This allowed a comparison of abundance based on annual inflow regime. In all analyses, zooplankton abundance was transformed to $\log_{10}$(abundance + 1), the usual method of transforming count data when means and variances are correlated. The data were analyzed by analysis of variance (ANOVA) for factorial measures with unequal cell frequencies. Due to the transformation, individual values given below are geometric rather than arithmetic means, unless otherwise indicated.

Lavaca Delta and Lavaca Bay, 1973 to 1975. Gilmore et al. (1976) sampled zooplankton at nine sites in the Lavaca Delta and in Lavaca Bay (Figure 5.3.2). Sampling began in September 1973 and continued through June of 1975. The sampling period was long enough to include two different fall, winter, and spring seasons, but only one summer season, in 1974. When all of the Gilmore et al. abundance data were analyzed, it was clear that the effects of station location were statistically significant (two-factor ANOVA; F = 51.2; df = 8,324; $P < 0.001$). Differences among stations were tested and the stations fell into two groups, delta stations (600-2, 65-2, and 617-2 in Figure 5.3.6) and bay stations (85-2, 90-1, 115-1, 143-2, 150-2, and 190-2). Zooplankton abundance at the delta stations was significantly lower than abundance in Lavaca Bay (Scheffe F-test for multiple comparisons; df = 359; $P < 0.05$). Further analyses treated the delta and bay stations separately.

For the three delta stations, zooplankton abundance was significantly different from site to site. Abundance was lowest at the most inland site (600-2) and increased the closer the sites were to Lavaca Bay (Scheffe F-test; df = 115; $P < 0.05$). Stations located in Lavaca Bay, however, did not have significantly different abundances.

At both the delta and bay sites, abundance was significantly related to season. In Lavaca Bay, winter and spring abundances were similar, but were significantly higher than summer and fall abundances (Scheffe F-test; df = 243; $P < 0.05$). The seasonal pattern was essentially the same for the delta stations.

The mean zooplankton abundance in the delta was 340 individuals/m^3; in the bay, mean abundance was 7,948 individuals/m^3. Figure 5.3.3 shows the average abundance of zooplankton by station and season using all of the Gilmore et al. (1976) data.

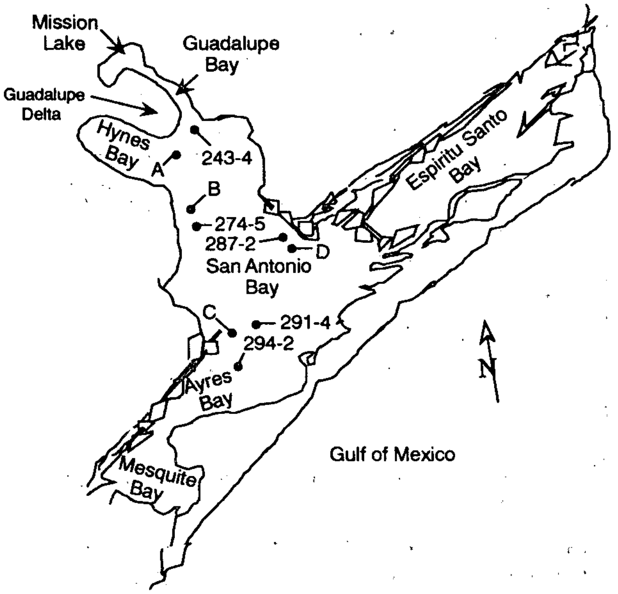


Figure 5.3.6. Zooplankton sample sites for the Guadalupe Estuary from Matthews et al. (1975) and Buskey (1989). Stations 243-4, 274-5, 287-2, 291-4, and 294-2 are from Matthews et al., while A, B, C, and D are from the Buskey study.

Lavaca Delta and Lavaca Bay, 1984 to 1986. In a two-year study, Jones et al. (1986) sampled zooplankton 14 times in the Lavaca Delta and Lavaca Bay. Four stations in the Lavaca Delta, one station at the mouth of Venado Creek, and one station in upper Lavaca Bay were sampled during the entire period (Figure 5.3.4). Two other stations in Lavaca Bay (1505 and 1905) were sampled at least once each season during the second year of the study.

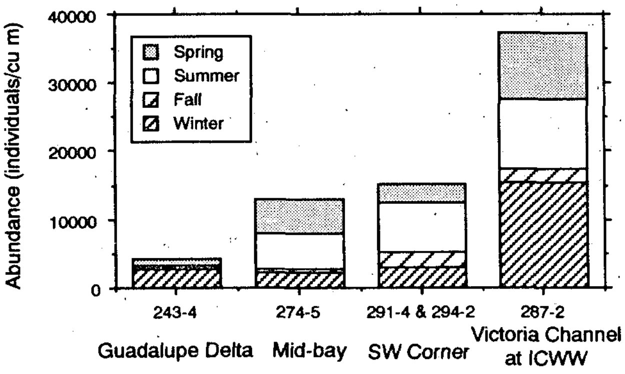


Figure 5.3.7. Seasonal average zooplankton abundance for the Guadalupe Estuary from Matthews et al. (1975). The abundance values are geometric means of the groups.

Analysis of this data showed a statistically significant relationship between station location and abundance (two-factor ANOVA; F = 17.7; df = 7, 64; $P < 0.001$). Like the earlier Gilmore et al. (1976) study, the sites within the Lavaca Delta (45-1, 603-1, 65-2, and 613-1) had significantly lower abundances than the sites in Lavaca Bay (85-1, 1505-1, and 1905-1) or at Venado Creek (623) (Scheffe F-test; df = 95; $P < 0.05$). Due to this difference, further analyses were done separately on delta and bay stations.

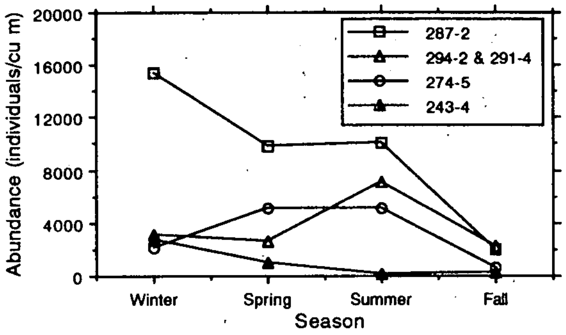


Figure 5.3.8. Station-by-station average zooplankton abundance for the Guadalupe Estuary for each season, from Matthews et al. (1975). The values are geometric means of the groups. The difference in abundance patterns for stations 287-2 and 243-4 versus 294-2 and 291-4 and 274-5 shows the effect of a statistically significant ($P < 0.05$) season-station interaction.

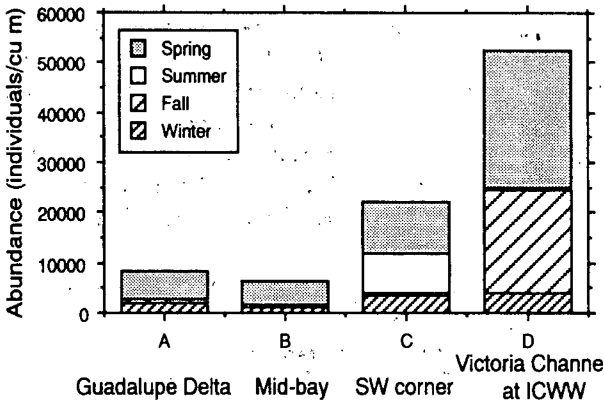


Figure 5.3.9. Seasonal average zooplankton abundance for the Guadalupe Estuary from Buskey (1989). The abundance values are geometric means of the groups.

There were significant differences in zooplankton abundance within the delta according to sample site location (two-factor ANOVA; $F = 3.63$; df = 3, 40; $P < 0.05$). Significantly fewer zooplankton were found at station 603-1 in Redfish Lake, a relatively inland site, than at station 613-1 at the mouth of Swan Lake, the delta station closest to Lavaca Bay (Scheffe F-test; df = 55; $P < 0.05$). Abundance at the two river stations (45-1 and 65-2) was not significantly different from abundance in the brackish lakes, however.

In Lavaca Bay, there were significant differences among the sampling sites (two-factor ANOVA; $F = 15.2$; df = 2, 6; $P < 0.01$). Abundance at station 85-1 near the mouth of the river was significantly lower than abundance at the other stations, 1505-1 and 1905-1 (Scheffe F-test; df = 17; $P < 0.05$).

For both delta and bay stations, abundance did not vary according to season for this study. Seasonal mean abundances were similar throughout the year for both areas. Overall, mean abundances for the delta and bay stations were 482 and 8,609 individuals/m^3. Figure 5.3.5 shows the mean abundance of zooplankton by station and season using the Jones et al. (1986) data.

Comparison of the Lavaca Bay zooplankton studies. Both studies, separated in time by ten years, showed that zooplankton abundance at delta stations was much lower than at bay stations. For the delta stations, the studies indicated that abundance at the station closest to the bay was significantly higher than at the most inland delta site. In the delta, it appears that sites with the highest abundance are those influenced the most by bay waters, while the stations with the lowest abundance are those with the least bay water influence.

At bay stations, the Gilmore (1976) study did not show significant differences among bay sites. The Jones et al. (1986) study showed that abundance at the bay station nearest the river (85-1) was significantly lower than at the two other bay sites.

The studies showed a major difference regarding the effect of seasons. In the Gilmore et al. (1976) study, winter and spring abundances were the highest but were not significantly different from each other; summer and fall abundances were the lowest and were not significantly different; but, the high winter-spring abundances were significantly greater than the low summer-fall abundances. The Jones et al. (1986) study, however, showed no significant abundance differences among the seasons, and abundance means for all four seasons were very similar.

Gilmore et al. (1976) noted that zooplankton abundance in Lavaca Bay was directly related to salinity and inversely related to temperature. Through regression analysis, the Jones et al. (1986) data confirmed a statistically significant direct relationship between abundance and salinity (multiple regression; df = 53; $P < 0.001$), and a significant inverse relationship with temperature (multiple regression; df = 53; $P < 0.01$).

Inflow and abundance in Lavaca Bay. Inflow conditions differed between the two studies, allowing a comparison of abundance based on freshwater inflow. Since abundance was often related to season and station location in the two studies, zooplankton data sets were chosen to include all four seasons and comparable station sites. A three-factor ANOVA with inflow, station, and season as independent variables was used to analyze changes in zooplankton abundance. Data for three delta areas (600-2, 65-2, and 617-2 from Gilmore et al. (1976); 603-1, 65-2, and 613-1 from Jones et al. (1986)) and three bay areas (85-2, 150-2, and 190-2 from Gilmore et al. (1976); 85-1, 1505-1, and 1905-1 from Jones et al. (1986)) were analyzed separately.

The inflow interval, which was matched with zooplankton abundance in the ANOVA, was defined to include the inflow in the month previous to the period of interest and to run for 12 months. For example, the inflow period for zooplankton samples taken from January 1974 through December 1974 began in December 1973 and ran through November 1974. This ensured that the actual inflow had occurred at the time zooplankton samples were taken. There was a large variation in the inflow during the period spanned by the studies. A 12-month period of high inflow (1,872,742 acre-ft) began with zooplankton samples taken in October 1973, and a 12-month period of low inflow (719,592 acre-ft) began with samples collected in October 1985. Using the 47-year period of record for comparison, 12-month inflows greater than the low-flow amount have occurred historically about 71% of the time, while inflows greater than the high-flow quantity have occurred only about 19% of the time.

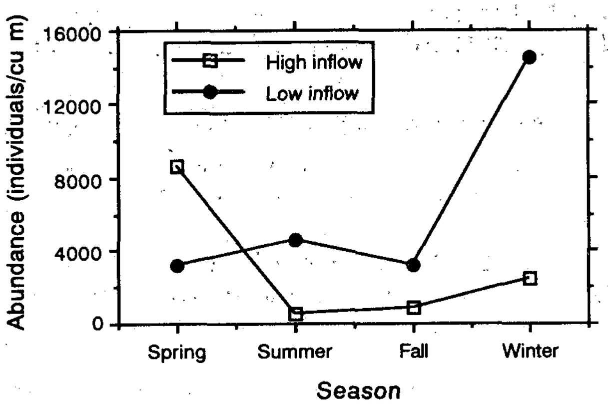


Figure 5.3.10. Inflow-related average zooplankton abundance for the Guadalupe Estuary for each season, from Matthews et al. (1975) and Buskey (1989). The abundance values are geometric means of the groups. The difference in the abundance patterns between the spring and winter for high- and low-flow conditions is the result of a statistically significant (P < 0.01) inflow-season interaction.

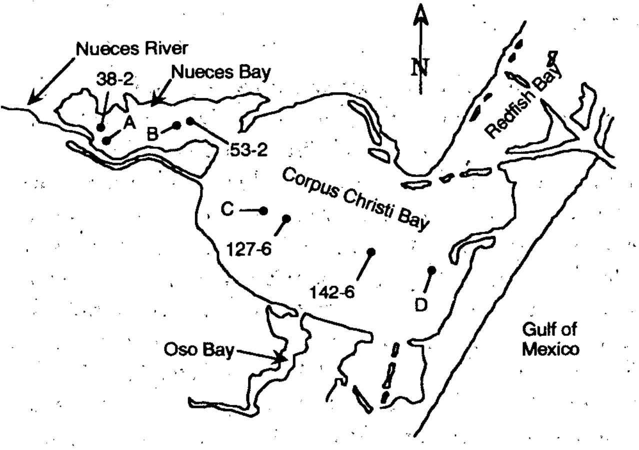


Figure 5.3.11. Sample sites for Nueces Estuary zooplankton from Holland et al. (1973, 1974, 1975) and Buskey (1989). Stations 38-2, 53-2, 127-6, and 142-6 are from the Holland et al. studies, while A, B, C, and D are from the Buskey study.

For the delta stations, inflow did not have a significant effect on delta zooplankton abundance. Station location and season were both statistically significant (three-factor ANOVA; F = 8.4; df = 2, 66; P < 0.001; and F = 3.5; df = 3, 66; P < 0.05) as expected from the analyses of the separate data sets.

For the bay data, there was a significant difference in abundance based on inflow (three-factor ANOVA; F = 7.09; df = 1, 65; P < 0.01). Annual abundance under high inflow (5,684 individuals/m^3) was significantly lower than annual abundance with low inflow (12,262 individuals/m^3). As expected, station and season were both statistically significant (three-factor ANOVA; F = 5.21; df = 2, 65; P < 0.01; and F = 2.93; df = 3, 66; P < 0.05) in the combined analysis.

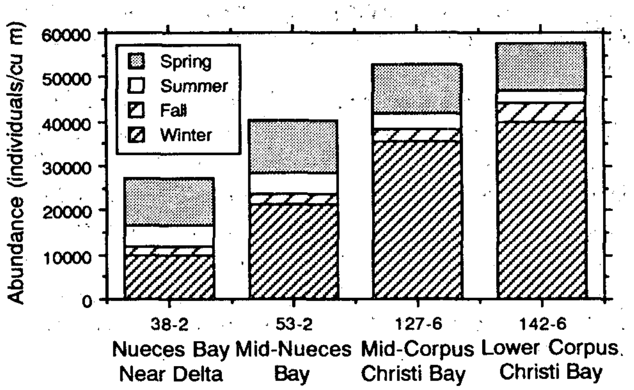


Figure 5.3.12. Seasonal average zooplankton abundance for the Nueces Estuary from Holland et al. (1973, 1974, 1975). The abundance values are geometric means of the groups.

Guadalupe Estuary, 1972 to 1974. Matthews et al. (1975) sampled zooplankton at 15 stations in the Guadalupe estuary during the period March 1972 through July 1974. Stations were sampled monthly or bimonthly. Data from five stations were selected for analysis since these stations were very close to the sampling sites of a later study by Buskey (1989); data from two of the stations (291-4 and 294-2) were combined because of their proximity to each other and to the location of one station from the Buskey study. Figure 5.3.6 shows the location of the sampling sites of the Matthews et al. (1975) and Buskey (1989) studies.

Zooplankton abundance was significantly related to season (two-factor ANOVA; F = 5.62; df = 3, 122; P < 0.01), station location (two-factor ANOVA; F = 14.3; df = 3, 122; P < 0.001), and the interaction of season and station (two-factor ANOVA; F = 2.43; df = 9, 122; P < 0.05). The presence of an interaction between season and station complicates describing the pattern of zooplankton abundance response to either factor by itself; the response with respect to one factor changes depending on the value

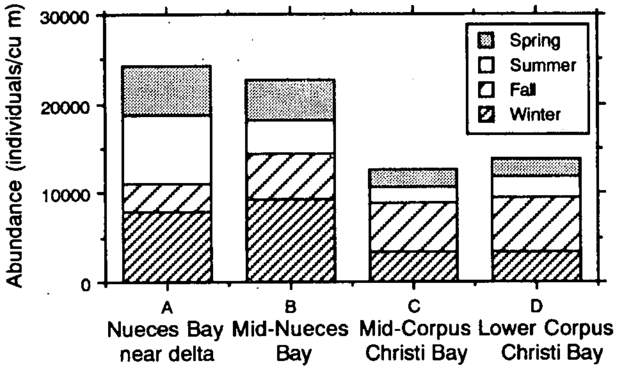


Figure 5.3.13. Seasonal average zooplankton abundance for the Nueces Estuary from Buskey (1989). The abundance values are geometric means of the groups.

of the other factor. Zooplankton abundance at stations 243-4 near the Guadalupe Delta and 287-2 at the intersection of the Victoria Channel and ICWW generally displayed the pattern most frequently seen in Texas estuaries: highest abundance in winter, slightly lower abundance in spring, and lowest abundance in summer and fall (Figure 5.3.7). The other two stations showed a different pattern: abundance was unusually low during the winter and was higher in summer than winter. The interaction effect identified by statistical analysis was the result of there being more than one pattern of abundance response to season, depending on which station is considered.

Station 243-4 is the only station for which a clear comparative statement about abundance can be made. This station had a significantly lower mean abundance than the other sites (Scheffe F-test; df = 137; $P < 0.05$), which do not differ significantly from each other.

Due to the interaction effect, the statistically significant relationship of abundance with season is somewhat overstated. Average winter abundance is significantly higher than the average fall abundance (Scheffe F-test; df = 137; $P < 0.05$). Figure 5.3.8, however, shows that the statistically significant result is almost entirely due to the high level of abundance at station 287-2. The mean abundance for samples in this analysis was 2,505 individuals/m^3.

Guadalupe Estuary, 1986 to 1987. Zooplankton populations were sampled at four stations (Figure 5.3.6) on six occasions during the period October 1986 through July 1987 (Buskey 1989). From the statistical analysis, zooplankton abundance was significantly related to station location (two-factor ANOVA; F = 3.31; df = 3, 20; $P < 0.05$). The Scheffe F-test did not reveal significant differences among stations. A less conservative multiple comparison test, Fisher's Protected Least Significant Difference, did show that zooplankton abundance at station D (in the lower portion of the bay near the intersection of the Victoria Channel and the ICWW) was significantly higher than at stations A and B (Fisher's PLSD; df = 35; $P < 0.05$).

Zooplankton abundance was also significantly related to season (two-factor ANOVA; F = 5.93; df = 3, 20; $P < 0.01$). Abundance during the spring was significantly higher than during the summer and fall (Scheffe F-test; df = 35; $P < 0.05$). Winter abundance, however, was not significantly different from any other season, although it was generally higher than in summer and fall. Mean abundance of zooplankton during the study was 2,408 individuals/m^3. Figure 5.3.9 illustrates the abundance by station and season for the study period.

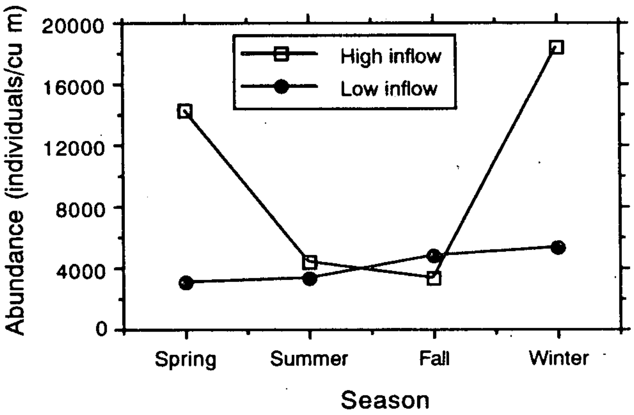


Figure 5.3.14. Inflow-related average zooplankton abundance for each season in the Nueces Estuary with data from Holland et al. (1973, 1974, 1975) and Buskey (1989). The abundance values are geometric means of the groups. The unusually low abundances during the winter and spring low-flow conditions are the result of a significant ($P < 0.01$) season-inflow interaction.

Comparison of the Guadalupe Estuary zooplankton studies. The two studies showed similar abundance patterns in corresponding areas of the estuary. Zooplankton abundance in the upper bay area tended to be low, while it was high in the lower bay region. Abundance during the summer and fall was low in both studies. Abundance in the winter and spring was not significantly different, but was significantly higher than in the summer and fall.

Armstrong (1987) stated that zooplankton abundance in the Guadalupe Estuary varied directly with salinity and inversely with temperature. His statement was based largely on observations from the Matthews et al. (1975) study. Data from Buskey et al. (1989) were analyzed for these relationships, using night samples alone, day samples alone, and day and night samples combined. In no case was there a statistically significant relationship between abundance and temperature or salinity. The unusually high inflow during the Buskey et al. study probably confounded the analysis,

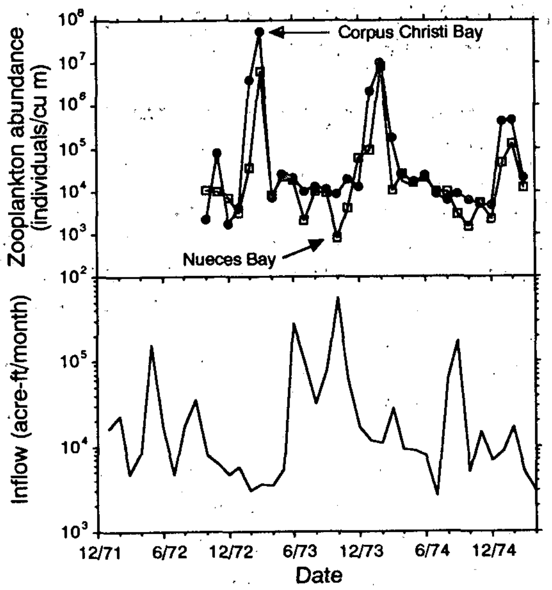


Figure 5.3.15. Average monthly zooplankton abundance and monthly combined inflow for the Nueces Estuary. In the upper panel, the Nueces Bay abundance data (open squares) are the arithmetic average of five stations, and the Corpus Christi Bay data are the arithmetic average of 16 stations. The lower panel shows monthly combined inflow to the estuary. The data come from Holland et al. (1973, 1974, 1975).

although the lack of a temperature-abundance relationship was surprising since there was a significant seasonal relationship.

Inflow and abundance in the Guadalupe Estuary. Calendar year 1987 had the highest inflow to the Guadalupe Estuary in the 47 years of inflow records. Inflow during the month of June alone (2,457,912 acre-ft) was greater than the 47-year annual average. Although the highest flows occurred toward the end of the period of the study, the 12-month period was wetter than usual throughout the entire year. To test for a relationship between inflow and abundance, inflows were matched to zooplankton abundance using an annual inflow interval, defined to include the flows that occurred from September 1986 through August 1987. This ensured that the actual inflow had occurred at the time zooplankton samples were taken. Inflow for the period of zooplankton collection by Buskey et al. (1989), which began in October 1986, totaled 5,682,025 acre-ft. This was higher than 97% of all annual inflows to the estuary.

Combined inflows for 12-month intervals that occurred during the Matthews et al. (1975) study were also calculated. The 12-month period beginning with July 1972 had the lowest inflow of all of the time spans for which zooplankton samples were collected, 2,167,570 acre-ft. Inflow for the 12-month period beginning in July is exceeded by about 52% of historical annual inflows. This period of inflow is low only in comparison to the extreme high inflows that occurred during the Buskey et al. (1989) study. Abundance from these two periods was analyzed by a three-factor ANOVA comparing the effects of inflow, season, and sample site on zooplankton abundance.

From the combined data sets, abundance was significantly related to inflow (three-factor ANOVA; F = 9.37; df = 1, 48; P < 0.01). Station and season were also statistically significant (three-factor ANOVA; F = 10.7; df = 3, 48; P < 0.001; and F =5.67; df = 3, 48; P < 0.01), and there was a significant interaction between inflow and season (three-factor ANOVA; F = 6.04; df = 3, 48; P < 0.01). Overall, zooplankton abundance was lower in the high-flow year (2,408 individuals/m^3) than during the low-flow year (5,382 individuals/m^3). The significant interaction did not substantially complicate the interpretation of the effect of inflow on abundance; Figure 5.3.10 shows that average zooplankton abundance was lower for the high-flow year in all seasons except the spring.

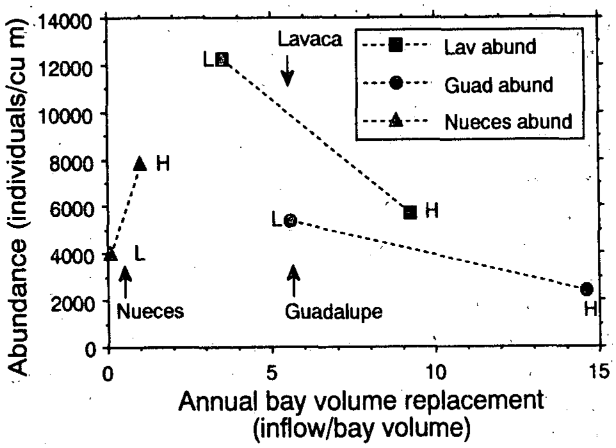


Figure 5.3.16. Annual average zooplankton abundance for high- and low-flow years versus the number of inflowing bay volumes of fresh water for the Lavaca Bay system, Guadalupe Estuary, and Nueces Estuary. The abundance values are geometric means of the groups. The arrows show the median annual inflows during the 47-year period of record. The dashed lines show only that the end points on the lines are from the same estuary; no sequence of events is implied. H = high flow; L = low flow.

Nueces Estuary, 1972 to 1975. Holland et al. (1973, 1974, 1975) collected zooplankton samples at 20 sites in Nueces and Corpus Christi bays. The samples were collected monthly; at some sites, there were 2.5 years of data for examination. A later study by Buskey (1989) concentrated on four stations, two in Nueces Bay and two in Corpus Christi Bay. Data from four stations in corresponding locations were drawn from the Holland et al. study to provide comparable samples. Figure 5.3.11 shows the location of the Holland et al. and Buskey sampling sites.

Table 5.3.1. Median combined inflow, bay volume, median inflow to bay volume ratio, and high and low inflow to bay volume ratios for Lavaca Bay, the Guadalupe Estuary, and the Nueces Estuary. Bay volume information from Diener (1975); inflow information from records at the TWDB. Volumes are in acre-ft and inflows are in acre-ft/yr.

Estuary	Median inflow	Bay volume	Median ratio	High inflow ratio	Low inflow ratio
Lavaca system [a]	1,120,440	203,506.7	5.51	9.20	3.54
Guadalupe Estuary [b]	2,208,939	388,693.8	5.68	14.62	5.58
Nueces Estuary [c]	414,337	865,513.1	0.48	1.01	0.08

[a] The Lavaca system volume is the sum of the volumes of Lavaca, Keller, and Chocolate bays.

[b] The Guadalupe Estuary volume is the sum of the volumes of San Antonio, Guadalupe, Hynes, and Mission bays.

[c] The Nueces Estuary volume is the sum of the volumes of Nueces, Corpus Christi, and Oso bays.

Zooplankton abundance from the Holland et al. data was not significantly related to station location. Season, however, significantly influenced abundance (three-factor ANOVA; F = 13.2; df = 3, 64; $P < 0.001$). Winter and spring abundance were both significantly greater than fall abundance, and winter abundance was significantly greater than summer abundance (Scheffe F-test; df = 95; $P < 0.05$). All other comparisons were not statistically significant. Figure 5.3.12 shows the mean seasonal abundances for the four stations. The dominance of winter abundance over other seasons is clear, especially at the two stations in Corpus Christi Bay (127-6 and 142-6). The average abundance for the samples used in the analysis was 7,257 individuals/m^3.

Nueces Estuary, 1987 to 1988. Buskey (1989) collected day samples at four stations in the Nueces Estuary approximately every other week for a year (Figure 5.3.11). The study took place during an extremely dry period with very low freshwater inflows.

An analysis of the data shows that zooplankton abundance was not significantly related to season. Station location, however, was a statistically significant factor (two-factor ANOVA; F =3.14; df = 3, 76; $P < 0.05$). While the stations in Nueces Bay had higher abundances than the Corpus Christi Bay sites, the differences were not significant according to the Scheffe F-test. Fisher's Protected Least Significant Difference, a less conservative multiple comparison test, indicated that abundance at station A near the Nueces Delta was significantly higher than both Corpus Christi Bay sites, and abundance at station B (mid-Nueces Bay) was significantly higher than station C in mid-Corpus Christi Bay (Fisher's PLSD; df = 91; $P < 0.05$). The low levels of inflow resulted in little month-to-month variation in salinity at any of the stations. Figure 5.3.13 shows the average abundance of zooplankton for seasons and stations from this study. The mean zooplankton abundance for the Buskey et al. study was 3,997 individuals/m^3.

Comparison of the Nueces Estuary zooplankton studies. There were not a lot of similarities between the two Nueces studies. Abundance patterns from the subset of stations selected from the Holland et al. (1973, 1974, 1975) data agree with the patterns noted by Armstrong (1987) for the entire set of data: maximum abundance occurred in the winter, and the summer and fall had the lowest levels of abundance. The Buskey et al. (1989) study data showed that the winter and fall had the highest average abundances, while the spring and summer were generally lower. Neither data set made a clear case for spatial differences in abundance in the Nueces Estuary. The Buskey et al. (1989) data showed significant station differences only when a less conservative test was used.

Holland et al. (1975) stated that temperature and salinity were the two most important factors regulating species composition, seasonal occurrence, and distribution of zooplankton. They concluded that low salinities resulted in low abundances and high salinities in higher abundance. They also noted that various zooplankton species were more numerous under warmer conditions, while others were most numerous when temperatures were the lowest. The effects of salinity and temperature on abundance from the Buskey et al. (1989) data were analyzed by regression, using the data collected during the day. There was a statistically significant inverse relationship between abundance and temperature (multiple regression; df = 99; $P < 0.01$). There was no significant relationship between abundance and salinity, although the range of salinities in the Nueces Estuary during the Buskey et al. (1989) study was limited (33.6 ± 4.9‰).

Inflow and abundance in the Nueces Estuary. Inflow during the Buskey et al. (1989) study was very low, 67,029 acre-ft, from September 1987 through August 1988. More than 95% of the annual inflows to the Nueces Estuary during the 47-year period of record exceeded the inflow that occurred during this year of zooplankton sampling. During the Holland et al. (1973, 1974, 1975) study, the inflow for the 12-month period beginning in September 1973 and ending in August 1974 was high, 871,596 acre-ft. Only about 24% of the annual inflows to the estuary were greater than this amount. These two substantially different inflow periods allowed comparison of zooplankton abundance under high- and low-flow conditions. A three-factor ANOVA was used, comparing the effects of inflow, season, and sample site on zooplankton abundance. The Holland et al. (1973, 1974, 1975) zooplankton abundance values used were from samples taken during the 12-month span from October through September to ensure that the actual inflow had occurred at the time the zooplankton samples were taken.

There was a statistically significant relationship between abundance and inflow in the combined data set (three-factor ANOVA; $F = 10.4$; $df = 1, 108$; $P < 0.01$); overall, zooplankton abundances were higher during the high inflow year. Station location was not a significant factor. Season was statistically significant (three-factor ANOVA; $F = 4.60$; $df = 3,108$; $P < 0.01$) and there was also a significant interaction between inflow and season (three-factor ANOVA; $F = 4.61$; $df = 3, 108$; $P < 0.01$). Figure 5.3.14 shows the seasonal mean abundances for the high- and low-inflow conditions. The pattern of abundance is different between the high- and low-flow years, thus causing the significant interaction. In the high-flow year, abundance is high during the winter, a little lower during the spring, and much lower in the summer and fall; this seasonal pattern is consistent with many of the other Texas zooplankton studies. During the low-inflow year, abundance is highest during the winter and fall and slightly lower for the spring and summer. Actually, average abundance did not vary much in the low-inflow year. The interaction effect involves the difference in abundance in the spring and winter in the high- and low-flow years. It is interesting to note that the summer and fall abundances during the high- and low-flow years are very similar. The low inflow appears to affect mainly spring and winter abundance levels. The average abundance is 7,868 individuals/m^3 under high-inflow conditions, and 3,994 individuals/m^3 under low inflows.

The Holland et al. (1973, 1974, 1975) data set is a particularly long record spanning 30 months. Figure 5.3.15 shows the pattern for the arithmetic mean zooplankton abundance for five stations in Nueces Bay and 16 stations in Corpus Christi Bay; the plot also shows monthly combined inflow. It appears that the average abundance peaks in the winter occur four to six months after the inflow peaks. The regularity of abundance peaks and the timing with respect to large monthly inflows (greater than 100,000 acre-ft) suggest that abundance is linked to inflow. A much longer record of zooplankton abundance would be needed to adequately test this timing hypothesis, however.

Comparing zooplankton abundance among estuaries. The major effect of station location from these analyses is that lower bay sites have higher zooplankton abundances than locations in or very close to the deltas. The studies on Lavaca Bay and the Guadalupe Estuary gave the clearest indication of this contrast. The Nueces studies do not fit into this pattern and suggest that during periods of extremely low inflow, zooplankton abundance may be higher near the delta than in other bay areas.

Season was generally a significant factor affecting zooplankton abundance. In four of the six studies, highest abundances in the bay occurred in the winter. In the other two studies (Lavaca Bay by Jones et al. (1986) and the Guadalupe Estuary by Buskey et al. (1989)), abundance was highest in the spring and slightly lower in the winter, but the two seasons did not differ significantly.

The effect of inflow on zooplankton abundance is not straightforward. For both Lavaca Bay and the Guadalupe Estuary, abundance was significantly lower ($P < 0.01$) during the high-inflow year than during the low-inflow year. In the Nueces Estuary, higher inflows were associated with significantly higher zooplankton abundances ($P < 0.01$). These estuaries receive different amounts of water relative to their volume, so it is useful to compare abundance with inflow amount.

Figure 5.3.16 shows the overall average zooplankton abundance from each study compared to the number of bay volumes of water that were received during the study (i.e., the number of times the bay could have been completely filled with inflowing freshwater). The dashed lines connecting point pairs are meant to help the reader determine which points are associated with each estuary, not to imply any sequence of change. The vertical arrows indicate the number of inflowing bay volumes represented by the 47-year median inflows. Median rather than mean inflow is used here because it is more representative of the central tendency of the data in frequency distributions that may be skewed by a few extreme measurements. See Table 5.3.1 for bay volume and inflow information.

The Lavaca Bay system and the Guadalupe Estuary have similar median bay volume replacements, 5.5 and 5.7 bay volumes per year, respectively. The Nueces Estuary, however, has a much lower median bay volume replacement, 0.5 volumes per year, less than 10% of the inflow of the other two estuaries on a bay-volume basis. Compared to the Lavaca Bay system and the Guadalupe Estuary, the Nueces Estuary is starved for fresh water and the materials carried in by inflow.

For the inflow-abundance test for Lavaca Bay, high inflow was 167% of the median, while low inflow was 65% of the median. For the Guadalupe inflow-abundance test, high inflow was 257% of the median and low inflow was 98% of the median (not a true low-flow situation); high flow was about 2.6 times the low flow for both systems. High flows were large enough to displace zooplankton from the bay, keep salinity levels low, and retard macrozooplankton population recovery, so high-flow abundance was lower than low-flow abundance. High-flow abundance in the Guadalupe Estuary was lower than high-flow abundance in Lavaca Bay. This is not surprising since the 15 replacement volumes of the Guadalupe system during the high-flow year were greater than the nine replacement volumes of Lavaca Bay, perhaps causing a greater displacement of zooplankton out of the bay or a greater effect of very low salinities on zooplankton species.

The median inflow state for the Nueces Estuary is greatly reduced in terms of inflow volumes compared to the other two estuaries. Certainly the lower input of fresh water and materials from reduced river flow and ungaged runoff is closer to limiting production processes in the Nueces Estuary than in the Lavaca or Guadalupe systems. The low-flow year had freshwater input equivalent to only 8% of the estuary's volume, only one-sixth of the median inflow. It is not surprising, therefore, that the annual average zooplankton abundance for the Nueces low-flow comparison shown in Figure 5.3.16 was lower than the low-flow abundances of the other two estuaries. The high-flow year had an inflow slightly greater than one estuary volume, 13 times the inflow of the low-flow year. This extra input of fresh water and materials (twice the median inflow) probably provided the right mix of conditions to allow increased secondary productivity. But it is not clear whether the higher productivity was due to a stimulation of primary production from macro- or micronutrient increase, stimulation of reproduction of zooplankton, stimulation of resting stages of zooplankton in the benthic sediments, control of predators, or other factors. The high inflow volume in the Nueces Estuary was still much less on a relative basis than high inflows in the Lavaca system and Guadalupe Estuary. Therefore, displacement of zooplankton by freshwater inflows was not a factor in zooplankton abundance.

From this limited analysis, it appears that freshwater inflow has a strong effect on macrozooplankton abundance. The response of zooplankton to inflow is different depending on the level of inflow. Freshwater inflow probably exerts its greatest influence on zooplankton production in systems receiving very little inflow in terms of bay volumes. Inflow increases result in greater zooplankton abundance. For systems normally receiving much more inflow (six bay volumes and more), the effects of increasing inflow decreases the standing crop due to displacement or other effects of low salinities. Figure 5.3.16 suggests that on an annual basis, there may be an optimum bay volume replacement for zooplankton abundance. Replacement rates of between one and six bay volumes per year may provide conditions that maximize mean annual zooplankton abundance.

Inflow Effects on *Acartia tonsa*

The copepod *Acartia tonsa* comprised 40 to 60% of the individual macrozooplankton in the Lavaca-Colorado, Guadalupe, Mission-Aransas, and Nueces estuaries (Holland et al. 1975; Jones et al. 1986; Buskey 1989). Armstrong (1987) noted that *Acartia* contributed to 85% of the standing crop in Sabine Lake, dominated the zooplankton in the upper and lower Laguna Madre, and with barnacle nauplii larvae constituted more than 70% of the standing crop in Trinity Bay. Since *Acartia* often represents a large fraction of the zooplankton, several studies have attempted to relate its abundance with environmental parameters including season, temperature, salinity, and inflow.

Secondary production. Buskey (1989) prepared a direct measure of secondary production of the abundant copepod *Acartia tonsa*. Potential secondary production, an indication of the rate of energy storage in biomass of the organisms, can be calculated from a life table using the age distribution of different developmental stages; the method is sensitive to short-term increases in population growth. In the Nueces Estuary in 1987, potential zooplankton production averaged about 1.9 mg C m^{-3} day^{-1} and ranged from 0.05 to 56 mg C m^{-3} day $^{-1}$, based on samples taken approximately every two weeks at four stations. A few high productivity measurements were observed but did not coincide with freshwater inflow events since inflows were very low throughout the study period.

Samples were taken six times at four stations in San Antonio Bay, including June 1987, the largest inflow event in the past 47 years. Excluding the June measurements, potential production of *Acartia* also averaged about 1.9 mg C m^{-3} day $^{-1}$, with a range of 0 to 9 mg C m^{-3} day^{-1}. During the June flood, average potential production was 72 mg C m^{-3} day^{-1}, with a range of 11 to 152 mg C m^{-3} day^{-1}. In the absence of flood conditions, average potential secondary production of *Acartia* was similar for the Nueces and Guadalupe estuaries. There appeared to be substantial stimulation of production during the high-inflow period, however. Duration of the increased production was not long-lived; a sampling of the stations six weeks later showed low production levels and very low zooplankton abundance, possibly due to displacement by fresh water.

By comparison, Heinle (1966) estimated maximum potential production of *Acartia* during summer months to be 29.6 mg C m^{-3} day^{-1} in the Patuxent River Estuary that opens into Chesapeake Bay. Durbin and Durbin (1981) determined mean potential production of *Acartia* at to be 19.0 and 22.9 mg C m^{-3} day^{-1} at two stations in Narragansett Bay; productivity estimates ranged as high as 140 mg C m^{-3} day^{-1}. Both sets of measurements were made during the summer to early fall period, when *Acartia* production is at its highest levels. Estimates of production from the Nueces Estuary during the corresponding time of the year ranged from 0.6 to 56 mg C m^{-3} day^{-1}, with a mean value of 9.0 mg C m^{-3} day^{-1}. Production in the Guadalupe Estuary ranged from 0 to 152 mg C m^{-3} day^{-1} with a mean of 25 mg C m^{-3} day^{-1}. A few extremely high readings after the June 1987 flood may bias the Guadalupe Estuary summer productivity estimates. Nevertheless, it appears that secondary production of *Acartia* in Texas estuaries is roughly comparable to production of the same species in East Coast estuaries.

Salinity. Holland et al. (1975) and Jones et al. (1986) associated low salinities with low densities of *Acartia*. They also noted that low salinities limit predators of *Acartia*, allowing rapid population growth to occur after high inflows. Jones et al. identified *Mnemiopsis mccradyi* (a jellyfish-like invertebrate called a comb jelly) as a probable predator that may control *Acartia* abundance. Buskey (1989) examined *Acartia* abundance with respect to salinity in the Guadalupe and Nueces estuaries. Only 5 to 6% of the variation in *Acartia* abundance could be statistically attributed to salinity; about 16% of the variability in the proportion of *Acartia* in the total zooplankton population could be attributed to salinity. Buskey's results were hampered by the bimonthly sampling regime in the year-long study of the Guadalupe Estuary and the lack of inflow and salinity differences in the one-year Nueces Estuary study.

Lee et al. (1987) assessed the relationship between *Acartia tonsa* and selected environmental factors using data from several different studies (Holland et al. 1975; Matthews et al. 1975; Gilmore et al. 1976; Wiersma et al. 1976; and unpublished data from Espey, Huston & Associates, Inc. cited in TDWR 1981). Combining data from six bays, Lee et al. showed that abundance is more highly correlated with salinity than temperature. When the bays were evaluated separately, *Acartia* abundance was more dependent on salinity than temperature in Nueces, Matagorda, and San Antonio bays, but was more dependent on temperature than salinity in Sabine Lake, Trinity Bay, and Corpus Christi Bay. Examining the independent variables used by Lee et al., mean temperatures varied over approximately the same range in all six estuaries for the data sets. Mean salinity variation had a range of 11 to 12‰ for Trinity Bay and Sabine Lake, and only 4‰ for Corpus Christi Bay; the range of mean salinity variation was 22 to 29‰ for the other bays. It is possible that the narrower range of salinity conditions in Trinity Bay, Sabine Lake, and Corpus Christi Bay contributed to the diminished importance of salinity in the analysis of these bays when compared to the nearly constant range of temperature variation present among all six estuaries.

Conclusion

Inflow displaces zooplankton, replacing estuarine or marine forms with freshwater species. Areas within and near river deltas have lower zooplankton abundance than the open bays, and under usual inflow conditions, winter and spring abundances are higher than in the summer or fall.

In the short-term, fresh water seems to stimulate micro- and macrozooplankton population growth, although the point at which displacement overbalances population increase is not clear. Inflow appears to influence zooplankton abundance on an annual basis, but the degree of influence seems more related to the relative amount of water entering the estuary than the absolute amount. In terms of the number of bay volumes of inflow received each year, estuaries that normally receive high inflows probably do not experience much increase in zooplankton abundance from higher inflows before the inflows physically displace the zooplankton due to the water movement. Estuaries that normally receive low inflows, in terms of bay volumes, seem to have significant abundance increases due to high inflows.

Salinity, which is affected by freshwater inflow, influences the abundance of single species such as *Acartia tonsa* and abundance of the entire zooplankton community in estuaries that are not experiencing periods of extreme high or low inflow. Unfortunately, as Armstrong (1987) noted, we are a long way from having the necessary information to adequately assess zooplankton dynamics; as yet, we cannot quantitatively relate inflow and zooplankton production or abundance except to say that there is a relationship.

5.4 EFFECTS OF INFLOW ON BENTHIC ORGANISMS

Introduction

Types of benthic organisms. Benthic organisms live in or on the bottom sediments of bays and are collectively called the benthos. Infauna are benthic animals that burrow into the sediments. As adults, many of the infauna are permanent residents of the bay bottom and are not particularly mobile; some are able to move only a few meters in any horizontal direction in their lifetimes. Examples of infauna include polychaete worms, clams, and lugworms. Most of the organisms living on the surface of the sediment (epibenthos) are mobile; crabs and snails belong to this group. Some of the animals that swim freely in the water column (nekton) are occasional visitors, seeking the bottom to feed on benthic species.

In addition to the animals described above, many small single-celled organisms live in the sediment including bacteria, fungi, protozoans, and microalgae. The latter constitute the microphytobenthos, algae that are mixed into the uppermost layer of sediment.

Benthic organisms are often grouped into three size categories, based on the mesh openings of the sieves used to separate them. Macrobenthos are the largest benthic organisms and are traditionally considered to be those visible to the unaided eye. Some older studies regarded macrobenthos to be organisms that were greater than 1 mm (0.039 inches) in any dimension. More recent works use 0.5 mm (0.0197 inches), the mesh opening size of a U.S. Standard No. 35 sieve, as the dimension limit (Levinton 1982). Examples of macrobenthos are adult stages of clams, polychaete worms, and crabs.

Meiobenthos are metazoans (multi-celled animals), ranging from 0.063 mm (4×10^{-5} inches) to 0.5 mm (0.0197 inches) in length; the former is the mesh opening size of a U.S. Standard No. 230 sieve. Some older studies considered meiobenthos to range from 0.1 to 1 mm (0.004 to 0.04 inch) in size. Meiobenthic species are further divided into the permanent meiobenthos that never become larger than 0.5 mm and the temporary meiobenthos that grow large enough to be classed as macrobenthos. Nematodes, harpacticoid copepods, gastrotrichs, and kinorhynchs are examples of the permanent meiobenthos; temporary meiobenthos include juvenile stages of clams, snails, polychaete worms, and amphipods.

Microbenthos is the smallest size class and include protozoans, microphytobenthos, and other microorganisms; any benthic organism smaller than 0.063 mm (4×10^{-5} inches) is considered part of the microbenthos.

Feeding habits of benthic animals. Macrobenthic organisms have a wide range of feeding habits in the benthic environment. Carnivorous species select and prey on benthic organisms. The lightning whelk (*Busycon contrarium*) and moon snail (*Polinices duplicatus*) are examples of predatory mollusks that feed on other mollusks. Many macrobenthic animals are scavengers, feeding opportunistically on anything they find including macroalgae, animal remains, detritus, and living animals; the blue crab (*Callinectes sapidus*) and striped-leg hermit crab (*Clibanarius vittatus)* are scavengers in Texas bays (Armstrong 1987). Filter or suspension feeders have sticky mucus-covered appendages or bristles that entrap plankton or organic material from water that flows by or is pumped by the animal. Mollusks such as the dwarf surf clam (*Mulinia lateralis*), the jackknife clam (*Ensis minor*), and the southern quahog (*Mercenaria campechiensis*) are primarily filter feeders (Fotheringham and Brunenmeister 1975; Armstrong 1987). Deposit feeders either select specific particles of food from sediment or ingest sediment directly and non-discriminantly consume organic detritus, algae, small animals, or bacteria contained therein. Say's Tellin clam (*Tellina texana*), the hemichordate acorn worm *Balanoglossus* spp., and the polychaete worm *Mediomastus californiensis* are abundant deposit feeders in Texas bays.

Most meiobenthic organisms in Texas estuaries are grazers and select single-celled microbenthos for food (Montagna and Yoon 1991); some, such as nematodes, are deposit-feeders. The feeding habits of the microbenthos are very diverse. Some species with chromoplasts are holophytic, satisfying their nutritional requirements photosynthetically. Saprozoic microbenthos obtain nourishment by absorbing dissolved organic matter from decaying organic matter. Holozoic microbenthos ingest or engulf organic particles or other microbenthos as their food source. Many of the microbenthos are able to use multiple modes of nutrition depending on the conditions encountered.

The sedimentary environment. The characteristics of the overlying water including salinity are important in determining the type and abundance of benthos, but sediment characteristics such as texture and organic content significantly influence the benthic community through size, mobility, and feeding style of the bottom-dwellers. For example, sandy sediments have interstitial spaces that favor very small meiobenthic species; bay bottom sediments consisting of flocculated clays that are frequently resuspended by turbulent flow may inhibit colonization by burrowing benthic species (Fleeger et al. 1983); and, sediments with high organic content may provide favored habitats for deposit-feeders that extract organic matter and associated bacteria.

Bottom sediments vary in texture depending on the proximity of rivers, gulf, and bay shoreline. Close to river mouths, the surface sediment is sandy silt grading into silty clay and clay-sized particles farther away from the river (see Table 4.4.1 for sediment size definitions). Clay-sized particles dominate the mid-bay areas away from the bay shores, while the areas closer to the bay shore tend to be sandier than mid-bay. Near the barrier islands, the clay-sized particles are mixed with fine sand that has been transported bayward by the wind or redistributed from washover channels and flood tidal deltas built during storms. The amount of organic carbon in the sediments also varies spatially and is high in the vicinity of river mouths and the deeper, central portions of bays.

The function of benthos. Benthic organisms have two functions in estuarine ecosystems: they are important components of the food web for higher trophic-level animals, and they process organic matter and release inorganic nutrients to the water column (Flint et al. 1982). Microbenthos are used as food by the meiobenthic organisms. Meiobenthos are preyed on by other meiobenthic, epibenthic, and macrobenthic animals, as well as nekton species such as brown shrimp, grass shrimp, and juvenile mullet (Anderson 1985; Collins 1985; Lassuy 1985a); many macrobenthos species are consumed by sport and commercial fish and shellfish.

Benthic organisms also process organic matter that settles to the bottom or is in suspension close to the bottom. All size classes of the benthos participate in this process, breaking detrital material and dead organisms into smaller particles and digesting complex organic molecules (such as cellulose from terrestrial plants) into simpler compounds. The micro- and meiobenthos are largely responsible for the release of inorganic forms of nutrients from organic compounds. These regenerated nutrients are then returned to the water column where they can be reused by the primary producers.

Microbenthos

There is relatively little information on the abundance and distribution of microbenthos in Texas bays. While there have been a few studies to identify and enumerate species of particular microbenthic groups (Wood 1963), most studies have been process-oriented, characterizing the rates of decomposition of organic carbon (Volkman and Oppenheimer 1962), benthic production (MacIntyre 1988), or regeneration of nutrients (Flint 1985) in which microbenthos is one of several components.

Bacterial density. Bacteria constitute a significant portion of the microbenthos. Montagna and Yoon (1991) made bacterial counts at four stations in the Guadalupe Estuary in January, April, and July of 1987, a year with very high inflows. Average bacterial densities for these months are shown in Table 5.4.1. There were only minor differences among stations and among months. For all three sampling periods, the inflows of the previous month were higher than average. The July sampling followed the highest monthly inflow to the estuary in 47 years.

Montagna and Kalke (1989a) compared bacterial cell density from sediments in the Lavaca-Colorado, Guadalupe, and Nueces estuaries in April of 1988. Following the example of the Guadalupe Estuary, each estuary had two upper bay stations and two lower bay stations. The average cell densities for these estuaries are also shown in Table 5.4.1. Cell densities were highest in the Guadalupe Estuary, lower in the Nueces Estuary, and lowest in the Lavaca-Colorado Estuary. Densities varied among stations within an estuary, and the Lavaca-Colorado cell densities increased from the most river-influenced station to the most marine-influenced station.

The year 1988 was relatively dry along the Texas coast compared to 1987; February, March, and April inflows to the Guadalupe Estuary were below average. The average bacterial density in April 1988 was less than half that measured in 1987. Densities in the other estuaries, which also had low inflows in 1988, were lower than in the Guadalupe Estuary. This comparison suggests that high average bacterial densities may be associated with high inflows; however, the data are very limited and do not provide conclusive confirmation of the pattern.

Microphytobenthos. Microphytobenthos are microscopic algae found in the upper few centimeters of benthic sediment. Light for photosynthesis at the sediment surface, measured as photosynthetically active radiation, is typically only a fraction of that found in the upper water column. Yet, there is a significant biomass of photosynthetically active chlorophyll *a* (2 to 40 mg/m^2) in the sedi-

Table 5.4.1. Average density of bacteria (cells/cm^3) in three Texas estuaries; data from Montagna and Kalke (1989) and Montagna and Yoon (1991).

Month	Guadalupe Estuary 1987	Guadalupe Estuary 1988	Nueces Estuary 1988	Lavaca-Colorado Estuary 1988
January	1.84×10^9			
April	1.45×10^9	5.02×10^8	9.39×10^7	1.61×10^7
July	2.03×10^9			

ment, and the benthic algae containing the chlorophyll contribute to community primary production at many sites (95 to 480 mg C m^{-2} day^{-1} from MacIntyre 1988; MacIntyre and Cullen 1988; Blanchard and Montagna 1992). At one site, MacIntyre (1988) measured microphytobenthic production to be 37% of the net areal production of 256 mg C m^{-2} day^{-1}.

The top centimeter of estuarine benthic sediment is frequently resuspended by turbulent mixing. Microphytobenthos primary production is generally light limited and extremely variable, largely as the result of fluctuations in irradiance in the water column during wind-induced sediment resuspension (Blanchard and Montagna 1988). While sediment resuspension decreases light levels, it also resuspends the microphytobenthic algae into the water column where more light is available for photosynthesis than at the sediment surface. MacIntyre (1988) noted that because of these two opposing processes, predicting whether net primary production will increase or decrease as the result of wind-induced resuspension is not easily accomplished. The likely significance of microphytobenthos to estuarine ecosystems is that it constitutes a large reservoir of photosynthetically capable organisms that, under the right conditions, can be resuspended into the water column and rapidly contribute to the system's primary production.

Meiobenthos

Influence of freshwater inflow on meiobenthic density and community structure. A little more information is available about meiobenthos than microbenthos. Rogers (1976) sampled meiobenthos in San Antonio Bay from May 1972 to January 1973 and found month-to-month variation in the density of nematodes, ostracods, copepods, and miscellaneous species (including kinorhynchs, polychaete worms, gastropods, pelecypods, and pycnogonids). The lowest densities occurred in the spring, with the highest densities in the fall or winter. Figure 5.4.1 shows the monthly variation in density and the inflow pattern. Monthly inflow in late 1971 and early 1972 was slightly greater than the 47-year average (195,000 acre-ft/month), with a much larger than usual inflow during May of 1972; thereafter, inflows remained near the median monthly value (119,000 acre-ft/month) until the end of the study.

Montagna and Kalke (1989b, 1992) compared the effects of freshwater inflow on meiobenthic populations in the Guadalupe and Nueces estuaries. The Guadalupe study took place in 1987; four stations were sampled in January, April, and July, two in the upper portion of the estuary, and two in the lower portion, east of the Intracoastal Waterway. In January, meiobenthos was about eight times as abundant at the lower bay stations as at the upper-bay stations (about 250,000 individuals/m^2). The density of upper bay meiofauna in April remained about the same, but the lower bay density decreased to about four times the upper bay density. Upper bay meiobenthic density remained unchanged in the July sampling, while the density of lower bay meiofauna declined to upper bay levels after a record inflow in June.

For the entire year, the average density of meiobenthos in the Guadalupe Estuary was four times as great at the lower bay stations as at the upper bay stations, 700,000 individuals/m^2 for the whole estuary (Montagna and Kalke 1989b). This is substantially lower than the average density of 5,800,000 individuals/m^2 measured by Rogers (1976). Figure 5.4.1 shows the average meiofaunal density for the entire estuary, and the inflow pattern for the six months before the sampling period. 1986 was slightly wetter than average, with an unusual inflow of more than 500,000 acre-ft occurring in December; January, February, and March of 1987 also had atypically high inflows. The largest monthly inflow on record, 2,457,912 acre-ft, occurred in June 1987, and July and August had unusually high inflows as well. Salinities at the upper bay stations averaged 1.4‰ during the six-month sampling interval; lower bay stations ranged from 4.1 to 13.2‰ in winter and spring, and were about 1‰ in July.

The lower bay stations of the Nueces Estuary in 1988 had higher meiobenthic densities than the upper bay sites, and the densities at most marine stations were 2.5 to 7 times higher than the upper bay sites (Montagna and Kalke 1992); the average meiobenthic density was about 1,300,000 individuals/m^2. Salinities during the Nueces benthic sampling in 1988 ranged from 27 to 45‰, much higher than the salinities in the Guadalupe Estuary the previous year. Figure 5.4.1 shows the inflow that occurred in the Nueces Estuary from June 1987 through August 1988. Inflow in June 1987 was 535,991 acre-ft, more than 66% of annual inflow for the year. After this flood, inflows remained extremely low through the end of the sampling.

Taxonomic groups of organisms that make up the meiobenthic community varied, depending on the freshwater inflow regime and the salinity characteristics of the bay system (Montagna and Kalke 1992). In the Guadalupe Estuary in 1987, nematodes constituted about 60% of the individuals of the lower bay stations, but only about 35% of the individuals of the upper bay stations; the high levels of inflow and low salinities had a greater effect on nematodes than other meiobenthic groups. There was also a surge of recruitment of juvenile mollusks at the upper bay stations associated with the inflow pulse. In the Nueces Estuary in 1988, nematodes constituted about 50% of the individuals at upper bay stations and one of the lower bay stations. The station with the greatest marine influence had nematodes

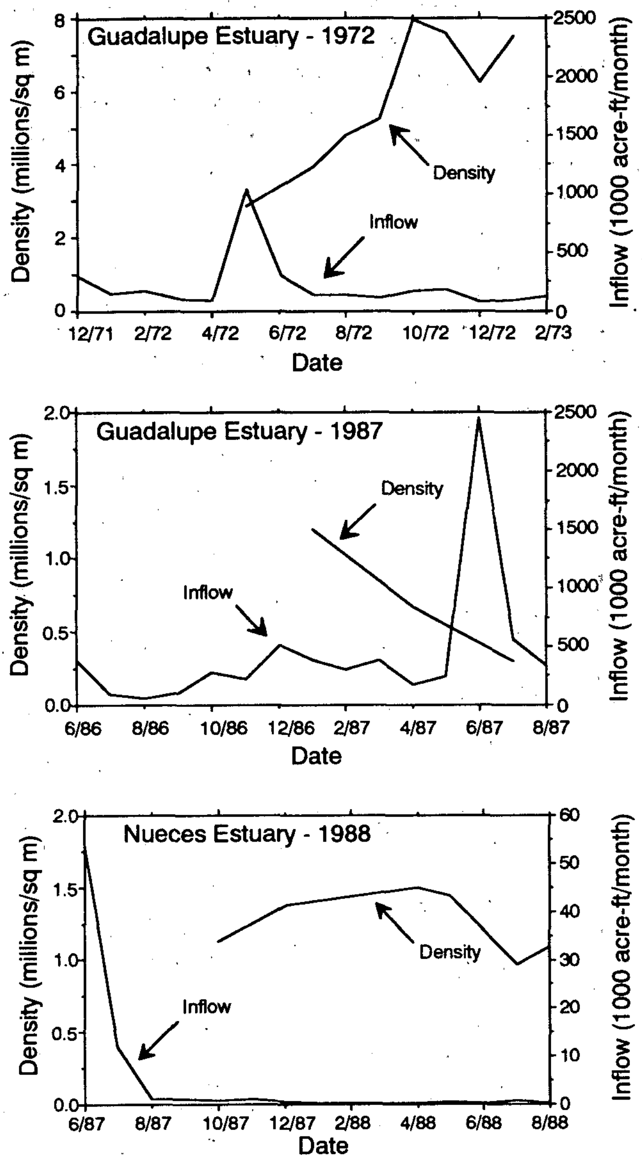


Figure 5.4.1. Density of meiobenthic species and monthly combined inflow for the Guadalupe Estuary in 1972 and 1987, and the Nueces Estuary in 1988. Guadalupe Estuary data are from Rogers (1976) and Montagna and Kalke (1989a); Nueces Estuary data are from Montagna and Kalke (1989b).

comprising more than 80% of the individuals.

A comparison of the patterns of meiobenthic density in these three studies does not suggest that season is a controlling factor. The reduction in meiobenthic density and changes in community structure in the 1987 Guadalupe study appear to be related to increased inflows; the decline in density with large inflows is probably the result of low salinity levels that are not tolerated by various species. Montagna and Kalke (1992) concluded that meiobenthos was typically a marine community and declined in density and diversity with increasing freshwater inflow.

The 1972 Guadalupe Estuary study (Rogers 1976) and the 1988 Nueces Estuary study (Montagna and Kalke 1992) show increases in meiobenthic density five to eight months after large inflows occur. Integrating the observations for all three studies suggests an overall cycle for meiobenthos. Very high inflows or long periods of moderately high inflow may produce low salinity conditions for several months that reduce the density of meiobenthos by eliminating individuals not adapted to prolonged periods of low salinity. Conditions become favorable for meiofaunal production, so their density increases and remains high for several months. At some point, meiobenthic density declines to lower levels and remains low until after the next major inflow event occurs. Perhaps the large input of organic and inorganic matter accompanying high inflows and the recycling of these materials over several months provides the energy source for meiobenthic density increase. Unfortunately, there is no continuous meiobenthic record at any site to substantiate this proposed pattern.

Effect of freshwater inflow on consumption of microbenthos. Montagna and Yoon (1991) studied the production of bacteria and microbenthic algae and their consumption by meiofauna in 1987 in the Guadalupe Estuary. They used radioisotopes to measure the production of bacteria and algae, and traced the radioactive label to six groups of meiobenthos: juvenile mollusks, juvenile polychaetes, and juvenile amphipods, which are all temporary meiobenthos; and harpacticoid copepods, nematodes, and others (largely ostracods, kinorhynchs, and turbellarians), which are all permanent meiobenthos. Overall, the grazing rate by meiobenthos on microalgae was four times as great as on bacteria. Juvenile mollusks consumed 39% of the microalgae and 68% of the bacteria; the "other" meiobenthos consumed 33% of the microalgae.

Grazing rates of the meiobenthic community on bacteria were 3.5 times as high at the freshwater-influenced stations as at the marine stations. This was largely due to juvenile mollusks whose rate of bacterial consumption increased by about 30 times at freshwater-influenced stations.

Meiobenthic grazing rates on microalgae were 2.5 times as high at freshwater-influenced stations as at marine stations, due to juvenile mollusks, "other" meiobenthos, and, to a lesser extent, harpacticoid copepods; the grazing rate of juvenile mollusks was more than 40 times the rate at more marine stations. According to Montagna and Yoon (1991), the nutrients associated with large inflows of fresh water stimulate microalgal production, and the microalgae are transported down the bay by the prevailing water currents. The meiobenthos respond to this abundance of microalgal food materials by increasing their grazing rates.

Montagna and Yoon (1991) noted that the grazing rates of microalgae from this study were several times greater than other meiofauna grazing rate measurements from the east and west coasts, but bacterial grazing rates were lower. They attributed the high level of microalgal grazing in the freshwater-influenced areas to the high production and advection of microalgae that occur during high freshwater inflows. As discussed in Section 5.1, there does appear to be a relationship between production and freshwater inflow up to moderate inflows. Beyond that point, the relationship is complicated by turbidity, displacement of algae by flushing, and other factors.

Response of Macrobenthos to Salinity

Background. Benthic animals have limited mobility; therefore, they usually serve well as indicators of environmental conditions. In addition, many benthic species are small, with high potential population growth rates and with mechanisms for larval dispersal and colonization. These features mean that the benthos are capable of a dynamic response to environmental change within a time scale of weeks or months. Benthic organisms are important also in the food chain of the estuary. For these reasons, monitoring the benthos in Texas estuaries has been an important feature of investigations.

The difficulties encountered in collecting and processing sediment samples pose the main challenge to the use of benthic organisms as indicators of environmental change. In the context of the analyses discussed below, the various methods that have been used in surveys of benthos in Texas bays limits the precision of comparisons among these data sets. Even so, valuable qualitative and quantitative comparisons can be drawn from these data concerning benthic populations over a complete salinity gradient.

A number of studies on Texas estuaries provide data on the distribution and abundance of benthic organisms and include analyses to identify the environmental factors that determine these distributions. Thorough surveys of Texas estuaries and the coastal shelf by the University of Texas Bureau of Economic Geology (cf. White et al. 1983, 1985) provide an excellent baseline comparison of benthic communities among the bays of the Texas coast. Their data demonstrate that substrate type is very important to the distribution and abundance of benthic species. Results of many other investigations into the benthos of Texas estuaries are presented in Armstrong (1987) and by Kalke and Montagna (1989). This section summarizes specific data from these and other studies that pertain to the influence of freshwater inflows on benthic organisms.

Most of the studies cited here deal with the benthos exclusive of oysters and oyster reef communities, as these organisms have received much attention for their economic importance (see Section 6.7). It should be noted, however, that oyster reefs and the shell bottoms associated with oyster reefs may enhance local populations of other benthic animals (Holland et al. 1973; White et al. 1985).

General patterns of distribution and diversity. There is a group of benthic species that are prevalent in Texas estuaries. Within this group, some are better adapted to high salinities, and others to low salinities. But in general, the range of salinities they can tolerate is broad. These species thrive in estuaries because they are tolerant of a wide range of conditions or because they have adaptations to avoid mortality from fluctuations in salinity. Table 5.4.2 (from Table 8 in Armstrong 1987 and other sources) lists some common species that often dominate benthic samples. Figure 5.4.2 shows the typical range of salinity conditions for many bays of the Texas coast (average ± one standard deviation in each bay) and indicates the distribution of common benthic species across this series of bays. The wide range of occurrence of some species, such as the polychaete worm, *Mediomastus californiensis*, and the dwarf surf clam (*Mulinia lateralis*) is striking in Figure 5.4.2, with regard to the range of salinities they inhabit. In contrast, the bivalve, *Rangia cuneata*, and the polychaete, *Hobsonia florida* (not in figure), represent species with distributions limited to fresher bays. There is a general trend toward increased importance of marine polychaetes in South Texas bays, at least in terms of numbers of species. These are represented by *Paraprionospio* and *Glycinde* in Table 5.4.2. Filter feeders,

Table 5.4.2. Common species of benthic macroinvertebrates in Texas estuaries.

Scientific name	Taxonomic group
Cossura delta	polychaete
Glycinde solitaria	polychaete
Littoridina sphinctostoma	gastropod
Lyonsia hyalina floridana	bivalve
Macoma mitchelli	bivalve
Mediomastus californiensis	polychaete
Mulinia lateralis	bivalve
Nereis succinea	polychaete
Parandalia fauveli	polychaete
Paraprionospio pinnata	polychaete
Rangia cuneata	bivalve
Streblospio benedicti	polychaete

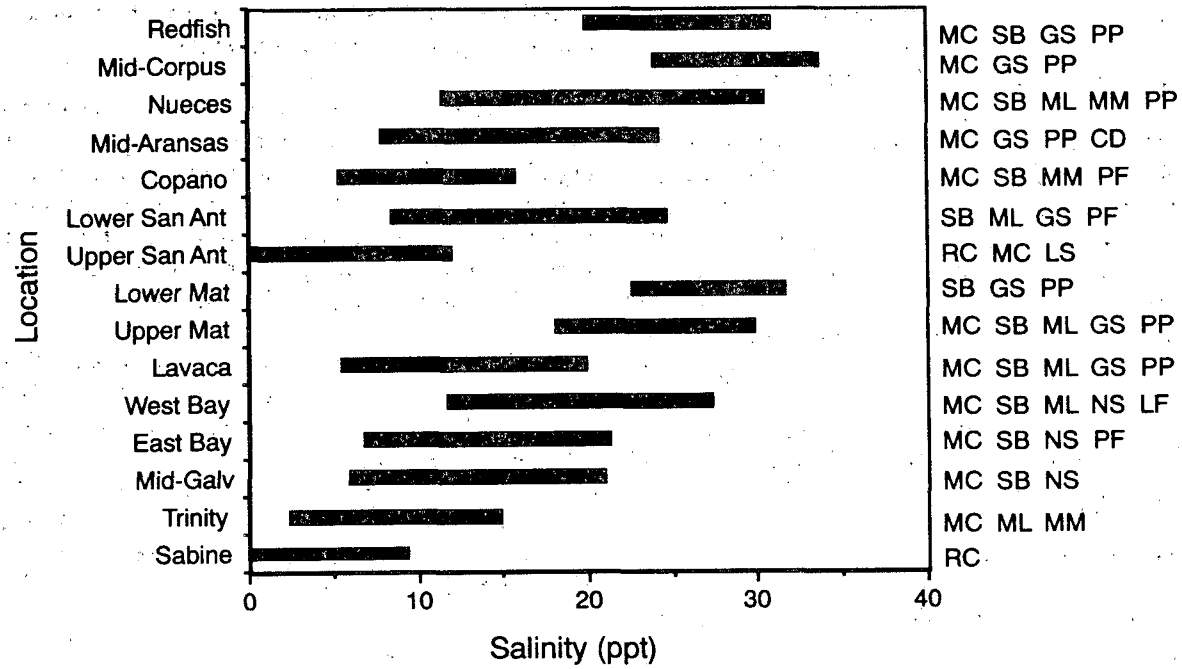


Figure 5.4.2. Salinity ranges, mean salinities (± one standard deviation) for representative bay areas, and benthic species commonly dominant in each bay, showing the range of salinities over which these species are commonly found. CD = *Cossura delta*; GS = *Glycinde solitaria*; LS = *Littoridina sphinctostoma*; LF = *Lyonsia hyalina floridana*; MM = *Macoma mitchelli*; MC = *Mediomastus californiensis*; ML = *Mulinia lateralis*; NS = *Nereis succinea*; PF = *Parandalia fauveli*; PP = *Paraprionospio pinnata*; RC = *Rangia cuneata*; SB = *Streblospio benedicti*.

e.g., bivalve mollusks, are more dominant in northern Texas bays where freshwater inflow is higher (Paul Montagna, University of Texas Marine Science Institute, personal communication).

Numerical abundance of a species does not necessarily equate to the importance of a species to the structure or productivity of the benthos. For example, one individual of *Rangia cuneata* might equal the biomass of 100 *Mediomastus californiensis* polychaetes. Colonization of the mid-Corpus Christi Bay sediments by the acorn worm (*Schizocardium*) during the spring of 1982 enhanced the biomass and abundance of the entire benthos through its bioturbating activities (Flint et al. 1983).

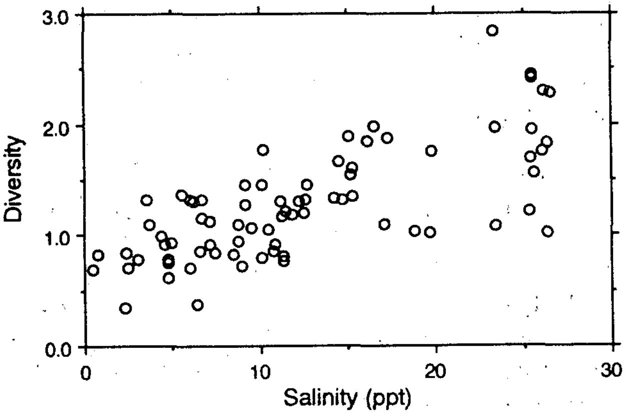


Figure 5.4.3. Average benthic species diversity (H′) versus average salinity for each collection site in several Texas bays; data from Gilmore et al. (1976), Mathews et al. (1975), Wiersma et al. (1976), Holland et al. (1975), and TDWR (1981).

To some extent, species are replaced by others with complementary salinity tolerances and performing similar roles over the salinity gradient within an estuary. There is also a change in the numbers of species present. In waters approaching marine salinities, there are increased opportunities for colonization by numerous marine species. In contrast, the possible pool of freshwater species to colonize portions of the bays that are periodically fresh is more limited. The result is a definite relationship between benthic diversity and salinity. Diversities were calculated using the Shannon-Wiener information measure (Wilson and Bossert 1971) for each benthic sample collected by a number of studies (Matthews et al. 1974; Holland et al. 1975; Gilmore et al. 1976; Wiersma et al. 1976; Copeland and Fruh 1970). Figure 5.4.3 presents long-term average salinity and benthic diversity for these collection sites, which span the Texas coast from Sabine Lake to Corpus Christi Bay. The positive relationship between benthic diversity and salinity shown in this plot is also noted by Kalke and Montagna (1989).

The trend of increasing benthic diversity with increasing salinity does not hold as salinities rise above marine levels to hypersaline conditions. Establishment of hypersaline

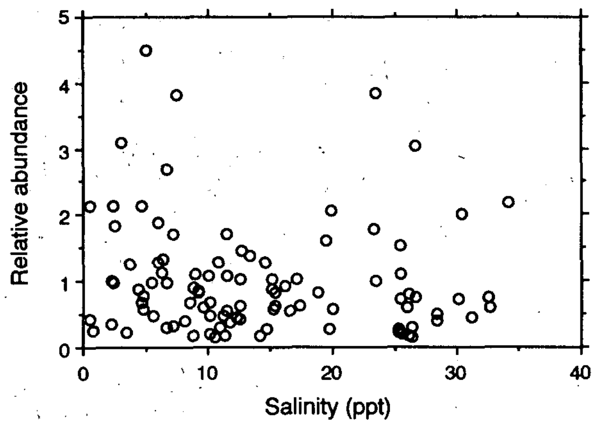


Figure 5.4.4. Relative average abundance of benthos at each collection site versus average salinity at that site; data from Gilmore et al. (1976), Mathews et al. (1975), Wiersma et al. (1976), Holland et al. (1975), TDWR (1981), and Montagna and Kalke (1989).

salinities are shown in Figure 5.4.4. From these relative numbers, trends of benthic abundances along salinity gradients across many bays can be compared. The figure does not show consistent trends. In some estuaries, abundances are higher at low salinity sites, while in other estuaries, abundances are higher at high salinity sites.

A graphical comparison of average benthic abundances among Texas estuaries is presented in Figure 5.4.5. Counts were tabulated from all sites and placed into several salinity categories. This summary was compiled from data in Matthews et al. (1974), Gilmore et al. (1976), Holland et al. (1975), Wiersma et al. (1976), TDWR (1981d), and Cornelius (1984). Over the period of collection in Alazan Bay, Cornelius (1984) found a full range of salinity conditions. Counts were highest in the range of marine salinities, but moderate at brackish salinities. A similar pattern of high conditions in Nueces Bay and Baffin Bay during periods of very low inflow has been observed to result in loss of benthic diversity (Paul Montagna, University of Texas Marine Science Institute, personal communication).

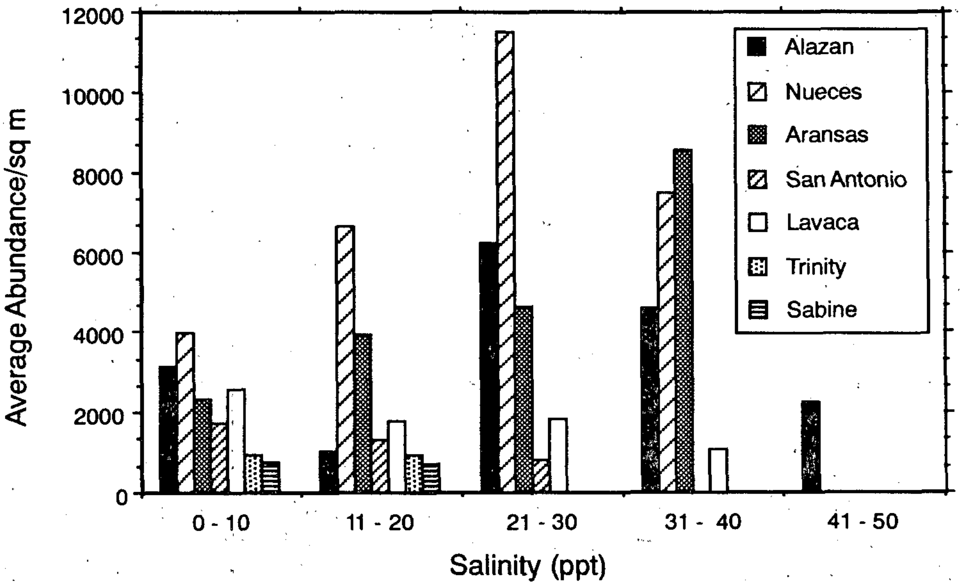


Figure 5.4.5. Average benthic abundances for collections tabulated by salinity categories; references cited in text.

General patterns of abundance. The abundance of benthic animals as a group is not a simple function of salinity. The general relationship between benthic abundance and salinity was explored through comparisons among data collected in a number of investigations. Abundance data from Texas bays was compared after transforming the data from each investigation into relative values. Relative abundance for each site within an estuary was calculated as the ratio of the average benthic abundance at that site to the grand average abundance calculated over all sites, specific to a given investigation. Relative abundances and average site

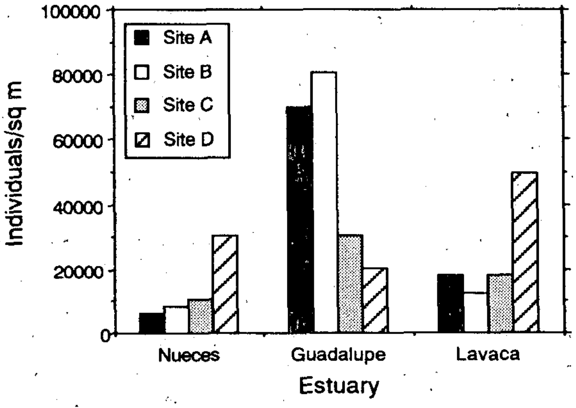


Figure 5.4.6. Average benthic abundances in synoptic sampling of three Texas estuaries, from data in Montagna and Kalke (1989); in each estuary, sites A and B represent the upper estuary; sites C and D represent the lower estuary.

abundance at near-marine salinities was seen in the Nueces Estuary. Ranges of salinities encountered in other bays during periods surveyed were less complete. It appears, however, that in San Antonio Bay, abundances are consistently highest at brackish salinities (0 to 10‰). Abundances of benthic macroinvertebrates in brackish Sabine Lake and Trinity Bay have been reported as lower than average numbers found in more saline bays (Copeland and Fruh 1970; Armstrong 1987), but this could be an artifact of sampling technique, as few collections have been made in these bays.

Comparisons of benthic populations were made among Nueces, Guadalupe, and Lavaca-Colorado estuaries by Montagna and Kalke (1989a, 1989b), on the basis of synoptic data collection in these systems. Results are illustrated in Figure 5.4.6. In each estuary, sites A and B represent the upper, fresher portion, while sites C and D represent the mid- to lower estuary. Similarities were found between the Nueces and Lavaca-Colorado estuaries in trends of increasing benthic abundance and biomass toward the Gulf-influenced part of the estuaries. In contrast, Guadalupe Estuary stations showed just the opposite trend.

Considering all the stations within each estuary, benthic abundance was highest in the Guadalupe Estuary, followed by the Lavaca-Colorado and Nueces estuaries. Two factors are cited by Montagna and Kalke (1989a) as a likely explanation for the difference between the Guadalupe Estuary and the other estuaries. First, the Guadalupe Estuary has high inflows in proportion to its volume (or a low residence time, see Armstrong 1982). This could be translated into a higher delivery of nutrients to this system (see Section 4.3). Second, the Guadalupe Estuary does not have the direct connection with the Gulf that the other bays have. This may dampen the swing toward higher salinity during periods of low inflow, swings which would destabilize the brackish fauna of this bay.

Benthic response to inflow events. Several benthic collection efforts have spanned periods of high and low inflow and documented effects on benthic populations. The effects varied, depending on the severity of salinity changes and the season. The acclimation of local populations was also probably a factor.

Heavy rains in the central Texas coast during September 1979 caused a salinity decrease in Alazan Bay of 33‰ (Cornelius 1984). Benthos abundances in Alazan Bay declined by 40 to 89% compared to the August levels, almost entirely due to mass mortality of the dominant dwarf surf clam. During the remainder of 1979, benthic population sizes remained stable, and diversity actually increased somewhat. In contrast, inflow to the Nueces Estuary resulting from this same rain system produced dramatic increases in benthic abundances in lower Corpus Christi Bay (Flint et al. 1982). The species that contributed most to this increase were mollusks which typically reach peak densities during the cool season. From data collected during the seven years preceding this event, Flint et al. found other periods during which benthic populations increased slightly in response to freshening of Corpus Christi Bay. No other episodes produced such a high increase, however.

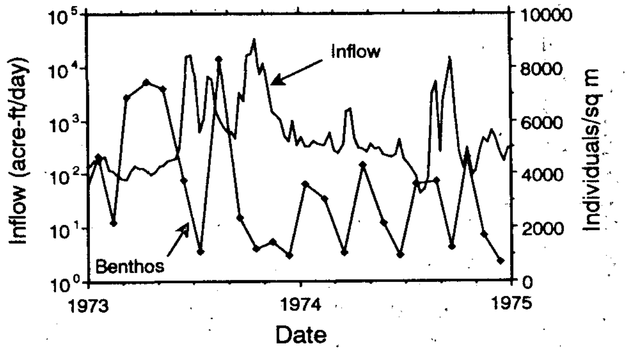


Figure 5.4.7. Benthos abundance (number/m^2) at mid-Corpus Christi Bay station 147-3 (Holland et al. 1975) and freshwater inflow to the Nueces Estuary.

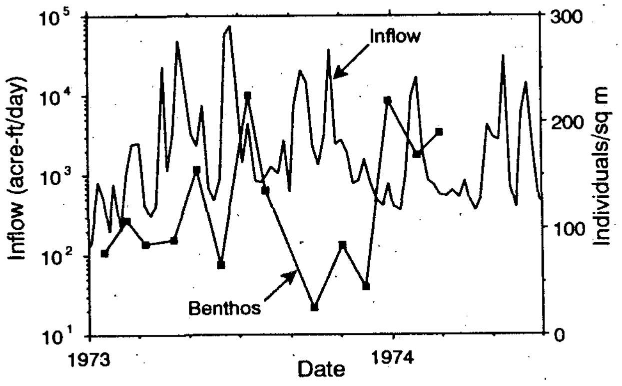


Figure 5.4.8. Abundance of benthos (number/m^2) in Lavaca Bay, station 90-2 (Gilmore et al. 1976) and freshwater inflow to the Lavaca-Colorado Estuary.

Figure 5.4.7 shows the variation over time of benthic abundance in mid-Corpus Christi Bay and daily inflows (benthic data from Holland et al. 1975, site 122-6). During 1973, the records seem to show that flood flows depress benthic populations. The population surge in August could be a lagged positive response to nutrients brought into the estuary by the earlier high flows. During 1974, the variability of benthic population sizes does not indicate a response to inflow variation.

Benthic populations in mid-Lavaca Bay seem to be enhanced by freshwater inflows to the bay during the first half of 1973 (Figure 5.4.8, data from Gilmore et al. 1976,

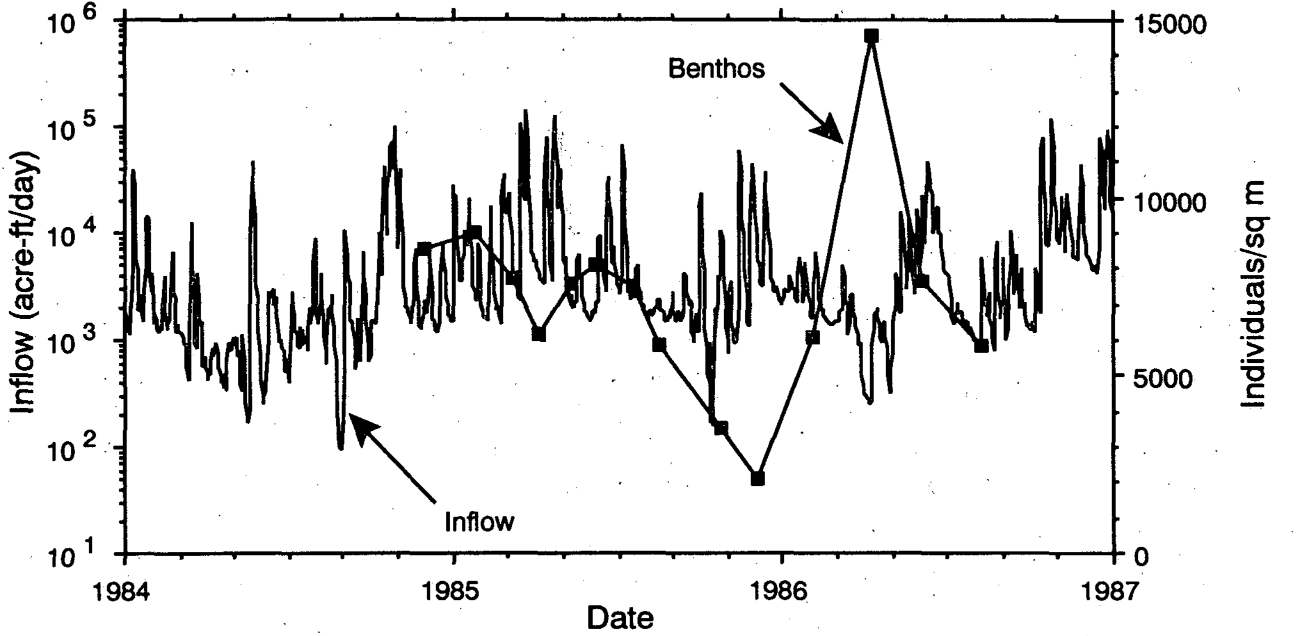


Figure 5.4.9. Abundance of benthos (number/m^2) in Lavaca Bay, station 85 (Jones et al. 1986) and freshwater inflow to the Lavaca-Colorado Estuary.

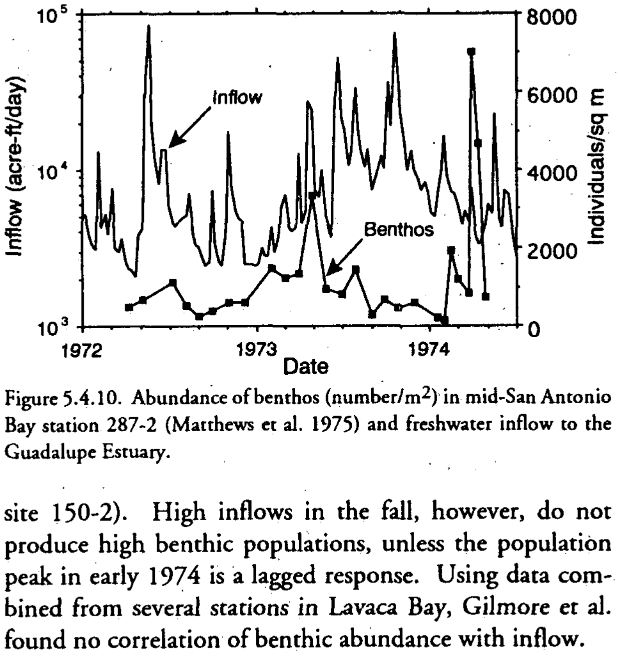


Figure 5.4.10. Abundance of benthos (number/m^2) in mid-San Antonio Bay station 287-2 (Matthews et al. 1975) and freshwater inflow to the Guadalupe Estuary.

site 150-2). High inflows in the fall, however, do not produce high benthic populations, unless the population peak in early 1974 is a lagged response. Using data combined from several stations in Lavaca Bay, Gilmore et al. found no correlation of benthic abundance with inflow.

Kalke and Montagna (1991) followed benthos in Lavaca Bay through periods of high and low freshwater inflow. Their data from upper Lavaca Bay (station 85) may show lagged positive responses of benthos numbers to inflow peaks (Figure 5.4.9) during November 1984 and June 1985. However, the benthic response is most easily interpreted as a population growth sequence during moderate and low freshwater inflow periods, with dramatic decreases during times of very high inflows.

Figure 5.4.10, from collections in mid-San Antonio Bay (station 287-2) by Matthews et al. (1975), shows instances during late 1972 and the early summer of 1973 in which the abundance increases during or immediately after an inflow peak. However, this association is not consistent; in the fall of 1973, high inflows were associated with a drop in benthic abundance.

That the previous figures do not show a consistent response of benthos to freshwater inflow may be due, in part, to the reproductive biology of important benthic species. Benthic invertebrates typically have planktonic larvae which settle on the sediment to develop. Good population recruitment depends on the environmental conditions encountered by these larvae. Calabrese (1969) found that although the adult surf clam has a wide salinity tolerance, successful development of embryos required salinities between 15 and 37.5‰. Therefore, the salinity regime and available nutrients following times of spawning by surf clam cohorts may have greater impact on population sizes in coming months than do salinity variations at other times.

Discussion. In some estuaries or sections of estuaries, the benthos responds to increased freshwater inflows by increases in populations. At sites with benthic fauna acclimated or adapted to more marine salinities, high inflows produced population declines. The high diversity of benthic species at near-marine salinities indicates that low salinities are stressful to many potential colonists of the estuary from gulf benthic communities. But some species do tolerate those conditions and flourish. The benefits to benthic

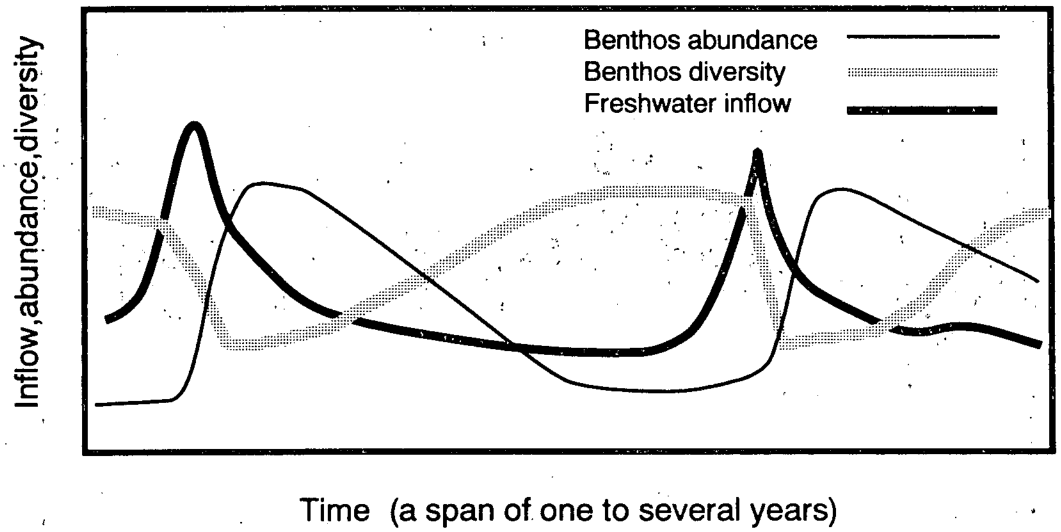


Figure 5.4.11. The Kalke-Montagna conceptual model of macrobenthos dynamics in Texas estuaries (adapted from Kalke and Montagna 1989).

species capable of surviving in the low salinity waters of upper estuaries are probably the result of nutrient material brought into the estuary by inflows. The benthos of the lower estuary also derive benefits from these nutrient inputs, but these effects are lagged or indirect (Flint 1984).

Kalke and Montagna (1989) present a conceptual model of the trade-off between nutrient richness and habitat stability for estuarine benthos. Figure 5.4.11 presents their idea diagrammatically. Typically, the hydrology of Texas estuaries consists of pulses of inflows, followed by periods of low inflow (see Section 4.1). During or immediately after high flows, a larger quantity of nutrient materials are delivered to the sediments. Benthic abundances skyrocket as the low-salinity species multiply and grow. At the same time, other species suffer mortality from low-salinity stress, and diversity decreases. As inflows decrease and salinity increases, more species can take advantage of the available nutrients. This leads to increased species diversity. As the benthos uses up available nutrients, numbers decline—perhaps unable to match increased predation rates—to sizes supported by inputs from the plankton and other sources. This conceptual model may help explain the differences in average benthic abundance and diversity among Texas estuaries.

Benthic Regeneration

The process of regeneration. Benthic regeneration starts with particulate organic matter (POM) that settles to the bay bottom, mixing with the fine surface sediments. POM includes: dead estuarine organisms; organic matter imported from rivers, adjacent bays, or the Gulf; material excreted by living estuarine organisms; waste materials from human activities; and organic aggregates produced by physical or bacterial action on dissolved organic material (Baylor and Sutcliffe 1963; Mann 1988). As the POM settles to the bottom, bacteria, fungi, and protozoans inhabit the surface of the decaying matter, slowly digesting it. Larger scavengers, such as crabs, and deposit feeders, including polychaete worms, mechanically and chemically break the particles into smaller pieces, digesting some of the organic material, and providing the reprocessed material as feces or pseudofeces for further microbenthic processing.

Regeneration involves the splitting of large organic macromolecules by hydrolysis into small organic molecules, and the remineralization of some of this organic material into inorganic compounds. The regenerated materials include phosphate, sulfate, silicate, carbon dioxide, ammonium, nitrite, nitrate, free amino acids, and other dissolved organic forms. Regenerated inorganic materials such as ammonium, nitrate, and phosphate are used as nutrients by estuarine plants to support their growth and photosynthesis. Some microorganisms directly take up amino acids, carbohydrates, and inorganic forms for growth (Klump and Martens 1983); a few chemotrophic bacteria use regenerated ammonium as an energy source and oxidize it to nitrite or nitrate (Sprent 1987).

Phosphate and nitrogen compounds are usually of greatest interest in regeneration since their availability is known to limit plant and bacterial productivity in aquatic environments. The regeneration of phosphorus is relatively straightforward; phosphate is taken up by microorganisms and autotrophs, incorporated into their tissues, and regenerated as phosphate and organic phosphorus compounds when they die. In general, the oxidation state of phosphorus does not change throughout the uptake-regeneration cycle.

Nitrogen is considered to be the nutrient in shortest supply in coastal waters (Ryther and Dunstan 1971); consequently, it has received the most study. Hydrolysis of proteins into amino acids and nitrogenous bases, and their deamination (release of ammonium) under aerobic conditions produces much of the regenerated nitrogen, but other biochemical reactions involving microorganisms in the aerobic and anaerobic layers of the sediments complicate nitrogen recycling. Nitrogen may exist in at least seven different oxidation states in estuarine water and sediments, ranging from its most reduced form, ammonium, to its most oxidized form, nitrate (Webb 1981). Regenerated ammonium may be oxidized to nitrite or nitrate, which can be used by phytoplankton and bacteria; regenerated ammonium and free amino acids may be adsorbed onto sediment particles and released to interstitial waters at a later time, or remain fixed and buried with the sediment (Rosenfeld 1979). A portion of the regenerated nitrate may be reduced by bacteria back to ammonium and then be taken up by other microorganisms to support their growth; some of the nitrate and nitrite may be denitrified by bacteria to form nitrogen gas or nitrous oxide which eventually escapes to the atmosphere. Factors such as temperature, salinity, reduction-oxidation potential of the sediment, water depth, rate of sediment and organic material deposition, nutrient loading of inflowing water, and benthic community composition all influence the many different pathways of oxidation and reduction of nitrogen compounds, and complicate attempts to analyze and explain observed rates of regeneration.

Importance of regeneration to phytoplankton production. Nixon (1981) attributed the relative shortage of nitrogen (compared to phosphorus) in coastal ecosystems to losses from denitrification that occur in the anaerobic benthic sediments. Other losses of nitrogen from the system include export to the sea or adjacent bays, burial, and harvest (sections 5.5 and 7.3 deal with these processes in more detail). The effect of these losses on phytoplankton and bacterial production is diminished due to the regeneration of nitrogen from particulate organic matter.

Regeneration takes place in the water column as well as in the bay sediments. Nixon (1981) pointed out that a major difference between coastal marine systems and open sea systems is the influence the benthos has on regeneration. Benner and Yoon (1989) measured water column and benthic regeneration at several stations in the Nueces and Guadalupe estuaries. The contributions by each of these processes to regeneration were about equal and within the range of regeneration rates for other estuaries. Their measurements were consistent with Nixon's (1981) observation that one quarter to one half of all the organic production and loading in shallow coastal marine systems was mineralized by the benthos.

Table 5.4.3. Percent of phytoplankton demand for nitrogen that could potentially be satisfied by benthic regeneration.

Bay or estuary	% demand		Reference
	mean	range	
Lavaca Bay	78	76 to 80	Jones et al. 1986 [a]
Guadalupe Estuary	44	29 to 72	Benner and Yoon 1989 [b]
Nueces Estuary	81		Flint and Kalke 1985 [c]
	9		Flint et al. 1986 [c]
	38	4 to 78	Benner and Yoon 1989 [d]

[a] One station, April and June 1986.

[b] Three stations, summer 1988.

[c] Three stations, ten or more samples each, May 1981 to October 1983.

[d] Four stations, summer 1988.

It is possible to evaluate the relative importance of regeneration by comparing the demand for nitrogen from phytoplankton with the regeneration rate. Nitrogen demand (gm N m^{-2} day^{-1}) can be estimated by multiplying net phytoplankton primary productivity (gm C m^{-2} day^{-1}) by the ratio of carbon to nitrogen found in healthy populations of phytoplankton (Redfield et al. 1963). Regeneration could potentially satisfy about 40% (range, 0 to 200%) of the phytoplankton demand for nitrogen in 13 coastal systems reported by Nixon (1981). Table 5.4.3 shows how much of the demand could be satisfied by benthic regeneration for studies in three Texas estuaries. The study by Benner and Yoon (1989) corrected for regeneration that occurs in the water column in the metabolism chamber; as a result, their figures are more indicative of benthic regeneration alone than the other two Nueces Estuary studies cited in the table. Benner and Yoon's estimates of 38 and 44% are very similar to the 40% average calculated from Nixon (1981).

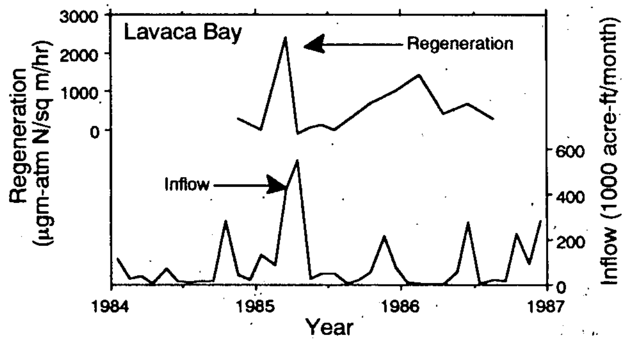


Figure 5.4.12. Regeneration rate in Lavaca Bay at station 85 (Jones et al. 1986) and monthly freshwater inflow.

Effects of inflow. Measurements of regeneration over a range of inflow conditions have been made in several Texas bays and estuaries. Jones et al. (1986) measured regeneration in 1985 and 1986 in Lavaca Bay, near the mouth of the Lavaca River. The inflow of 1.71 million acre-ft in 1985 was greater than average (exceedance probability 31%) while the inflow of 1.01 million acre-ft in 1986 was less than average (exceedance probability 64%). Figure 5.4.12 shows the regeneration rate measurements compared with monthly inflows. The early peak in regeneration in March of 1985 occurred during a high inflow period, but the lowest regeneration rate occurred the following month, when monthly inflow was even higher. There was only a modest decrease in benthic standing crop (abundance) and benthic biomass accompanying the sharp decline in regeneration, but the abundance of a few species such as the polychaete, *Streblospio benedicti*, and the pelecypod, *Macoma mitchelli*, did decrease by more than 50% (Jones et al. 1986). Regeneration remained low through the summer of 1985 and began to rise to a peak in February of 1986, after which it declined.

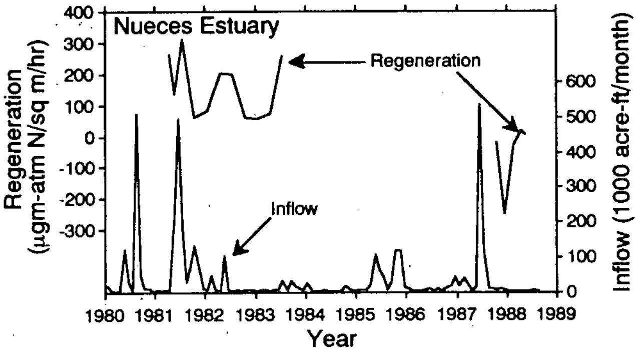


Figure 5.4.13. Average regeneration rate in the Nueces Estuary (Flint et al. 1983; Flint and Kalke 1985; Flint et al. 1986; and Montagna and Yoon 1989) and monthly freshwater inflow.

Regeneration in the Nueces Estuary from April 1981 through October 1983 was reported by Flint, Kalke, and McCoid (1983), Flint and Kalke (1985), and Flint et al. (1986). The average regeneration at two bay stations from these studies is compared to the inflow pattern in Figure 5.4.13. Inflow in 1981 was six to eight times the inflow in 1982 and 1983 (1.2 million acre-ft versus 215,000 and 150,000 acre-ft) and was substantially greater than the median inflow of 414,000 acre-ft. Flint and Kalke (1985) pointed out that the peaks in regeneration occurred in July of each year and were associated with freshwater inflow pulses, although Figure 5.4.13 shows that not every inflow pulse was accompanied by elevated regeneration rates. With the limited amount of regeneration information available, it is hard to say if the peaks are related to season or inflow. There was not a strong correlation between the number and biomass of benthic species and regeneration rates at the Nueces and Corpus Christi bay stations.

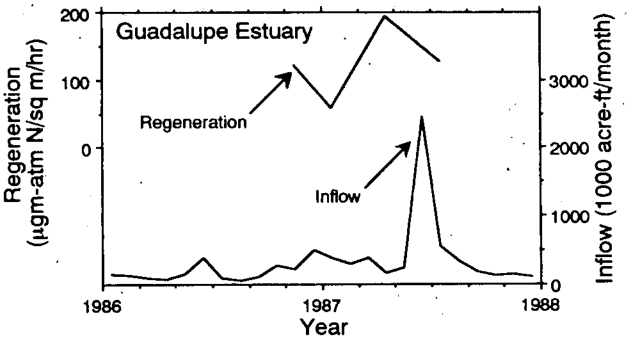


Figure 5.4.14. Average regeneration rate in the Guadalupe Estuary at stations A and C (Montagna and Yoon 1989) and monthly freshwater inflow.

Montagna and Yoon (1989) measured regeneration five times in the Nueces Estuary, from November 1987 through May 1988 (Figure 5.4.13); this was a period of very low inflows to the estuary. Regeneration was negative for the first three sampling periods—sediments took up more ammonium from the water column than they regenerated. While there was a net ammonium flux from the sediments in April and May, the regeneration rate was quite low compared to other measurements.

Montagna and Yoon (1989) also measured regeneration in the Guadalupe Estuary in November 1986 and January, April, and July 1987 (Figure 5.4.14). Inflows during that period were very high. Except for May of 1987, all other months had inflows higher than average, and June had the highest monthly inflow measured in the 47-year inflow record. It is interesting to note that the decrease in regeneration rates in January occurred after a several-month period of high inflows, and the decrease in July occurred after a month of extremely high inflows.

These four studies were not entirely comparable and were different with respect to duration, spatial coverage, measurement method, and length of time between samples. However, when considered together, they suggest a possible pattern of regeneration with respect to inflows. Periods of three or more months with low inflows result in low regeneration rates, sometimes even uptake of ammonium by the sediments. If a period of low inflow and low regeneration is followed by a moderately large freshwater inflow pulse, regeneration increases substantially. If the inflow pulse is very large or is followed by other large pulses in succeeding months, regeneration declines. The pattern suggests that freshwater inflow pulses may turn on or turn off higher levels of regeneration, depending on the timing and quantity of the inflows. The data do not indicate how long regeneration may continue after a single pulse, but this may be irrelevant since the usual pattern of inflows is an irregularly spaced sequence of pulses. While it seems possible that there is a relationship between inflow and regeneration due to the coincidence of flow and regeneration peaks, it is not a clean correspondence; the effects of season may also confound the relationship. While the pattern described here is plausible, the duration of studies and frequency of sampling is such that the data do not provide definitive confirmation. Moreover, regeneration is not well correlated with the abundance and biomass patterns of macrofauna, and meio- and microbenthos population data are inadequate to support such a hypothesis. Billen's (1978) study of recycling in North Sea sediments showed that about 80% of benthic recycling was due to meio- and microbenthic organisms. If the same relationship held for the benthos of Texas estuaries, large changes in regeneration rates would have parallel changes in meio- and microbenthic populations.

Conclusions

Benthic populations and benthic processes appear to be affected by freshwater inflow. There is some evidence to suggest that microbenthic bacterial density varies with freshwater inflow, although further study will be needed for confirmation. Meiobenthic density and community structure is definitely influenced by freshwater inflow; high inflows result in overall decreases in meiobenthic density and diversity, decreased density of nematodes, but increased density of juvenile mollusks. Recovery and expansion of meiobenthic populations may occur some months after inflow pulses, although this will require more extensive observation for confirmation. The rate of consumption of bacteria and microalgae by meiobenthos increases dramatically after inflow pulses, stimulated by high levels of bacterial and microalgal production.

Macrobenthos species have tolerances to broad ranges of salinity, although some species are adapted to high salinities and others to low. Generally, benthic diversity increases with salinity; for estuaries with free access to the Gulf, abundance and biomass tends to increase along the axis toward the Gulf. The opposite trend was found in the Guadalupe Estuary, which has indirect connection to the Gulf through other bays. There are examples of dramatic increases and decreases in macrobenthic abundance with inflows; this is most likely related to the type of benthic community in place. High diversity macrobenthic communities adapted to near-marine conditions are severely stressed when salinities fall to low levels during large freshwater inflow surges. But communities adapted to brackish conditions or species that can tolerate low salinities may flourish due to the increased nutrient load carried to the estuary by the inflow.

Regeneration appears to be influenced by freshwater inflow: in some cases, high regeneration rates occur during and immediately after high inflows; however, prolonged or unusually high inflows may be accompanied by very low regeneration rates. Integrating benthic abundance, community structure, regeneration, and other processes will require a study over several years of these components, with frequent sampling that includes a range of inflow regimes.

5.5 INFLUENCE OF FRESHWATER INFLOWS ON NUTRIENT CYCLES

Introduction

Carbon, nitrogen, phosphorus, and other nutrients undergo cycles in estuarine ecosystems in which the elements exchange many times between trophic levels (i.e., different members of the food chain) before being lost from the system. From analyses comparing phytoplankton nutrient demands with nutrient supply, it is clear that in many estuaries much of the nitrogen which is incorporated into phytoplankton biomass has been recycled rather than newly brought into the estuary (cf. Flint 1984; Benner and Yoon 1989). In systems that are relatively stable or that exhibit predictable patterns of physical and chemical variation, nutrient cycling accounts for an efficient utilization of available nutrients. In estuaries under physical or chemical stress or experiencing a major salinity transition, the pathways of nutrient cycling may be disrupted and the system may not be as efficient in using nutrient inputs (Edwards 1981). Texas estuaries experience both perturbation and stability in nutrient cycling, depending on the area and time. Fluctuation in freshwater inflows is one of the major perturbing influences. This section discusses the relationships of freshwater inflows to estuarine cycles of carbon, phosphorus, and nitrogen, with most emphasis on the nitrogen cycle.

The Carbon Cycle

Dissolved carbonates in seawater and carbon dioxide in the atmosphere serve as major sources and sinks for available carbon in estuaries. Studies of carbon cycling deal mainly with organic carbon fixed by photosynthesis, which serves as the basic fuel for secondary productivity in estuarine ecosystems.

Differences among Texas estuaries in aspects of their carbon cycles could develop from differences in their predominant sources of fixed carbon. As discussed in Section 5.6, terrestrially derived carbon may be more important in estuaries which receive high inflows, such as the Guadalupe Estuary, than in low inflow estuaries such as the Nueces Estuary. Figure 5.5.1, modified from a figure presented by

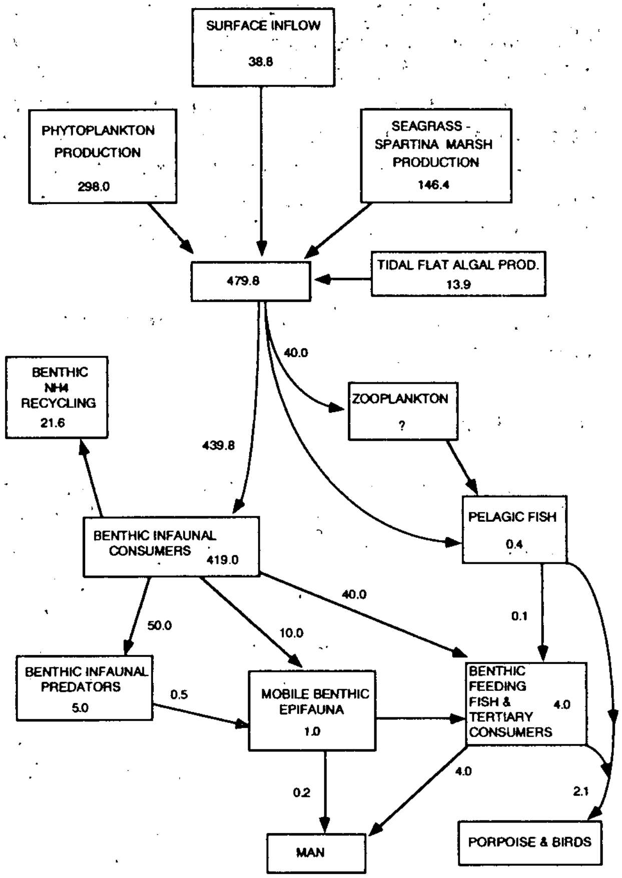


Figure 5.5.1. Organic carbon flow in the Nueces Estuary. Masses are in gm C/m^2; rate of production and flux are in units gm C m^{-2} yr^{-1}. Figure modified from Flint et al. (1983).

Flint et al. (1983), displays the movement of carbon among components of the Nueces Estuary. In estuaries of the upper Texas coast, the quantities associated with seagrass would be much smaller, while materials derived from phytoplankton would be greater (Armstrong 1987). In all Texas estuaries, periodic floods provide large quantities of carbon associated with terrestrial detritus.

The Phosphorus Cycle

Phosphorus is very important in the control of productivity in freshwater ecosystems. In estuaries, however, it is not as important over the long term as nitrogen. Adsorption-desorption reactions of phosphorus with suspended clays and organic particulates play an important role in keeping phosphorus available in the estuary. Phosphorus follows basically the same path as carbon in the estuarine system, except that the sediment serves as both a sink and a source, depending on fluctuations of phosphorus concentrations in the water column and rates of sediment resuspension (Pomeroy et al. 1965; Montagna et al. 1989). Montagna et al. (1989) found no significant association between inflow rate and phosphorus flux from the sediment in Nueces and Guadalupe estuaries.

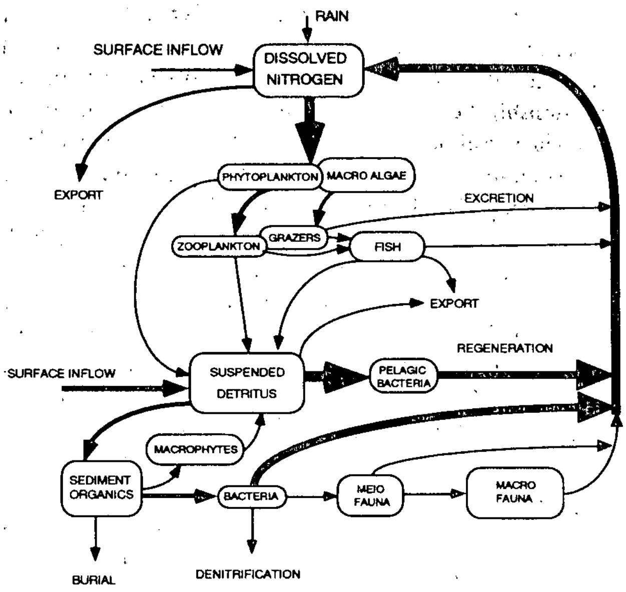


Figure 5.5.2. General diagram of compartments and flows of the nitrogen cycle in an estuary. Sizes of boxes and arrows reflect typical relative magnitudes of rates and masses. Figure based on Nixon (1981).

The Nitrogen Cycle

General features. Major components of the nitrogen cycle in estuaries are diagrammed in Figure 5.5.2, based on the conceptual form presented by Nixon (1981). The size of boxes and arrows illustrate the relative importance of the parts based on recent research in Texas estuaries. Phytoplankton uptake of dissolved nitrogen is the major process bringing dissolved inorganic nitrogen into the food chain in most bays, although seagrasses and macroalgae contribute greatly in some bays (Armstrong and Gordon 1979; Dunton 1989). Bacteria, fungi, and protozoa help make nitrogen from terrestrial, marsh, and seagrass detritus available to phytoplankton. Losses of nitrogen from the system include burial in the sediments, export and advection out of the estuary, and loss to the atmosphere through denitrification.

Nitrogen loss through denitrification. Denitrification is the biochemical reduction of available nitrate and nitrite to biologically unavailable nitrogen gas by bacteria. Bacteria use these oxidized nitrogen compounds as electron receptors in their respiration. Denitrification occurs in anoxic sediments that have nitrate in the sediment porewater (Seitzinger 1988). Nitrate and nitrite are present in the porewater as the result of a second biochemical process in the sediments, nitrification. In the latter process, ammonium (derived from the catabolism or breakdown of dead organic material in the sediment) is oxidized to form nitrate and nitrite.

The nitrogen gas that is formed as the result of denitrification is dissolved in the estuarine water and diffuses to the atmosphere where it is lost from the estuarine nitrogen cycle. Denitrification can result in significant nitrogen losses from estuarine systems. For example, Smith et al. (1985) estimated that 50% of riverine nitrogen entering Four League Bay in Louisiana was lost to denitrification.

Feedback loop. The standing stock of phytoplankton and zooplankton at any one time is small relative to other components of estuarine systems. Figure 5.5.2, however, shows a major loop of nitrogen through other constituents of the system, feeding back to the planktonic producers as dissolved nitrogen. This feedback loop is the essential element of recycling and maintains high production rates in the plankton. Within the plankton, rates of nutrient uptake, growth, herbivority, and carnivority are high. Thus, the planktonic processes are analogous to small, quickly turning gears driving the larger wheels of a complex machine.

Exchange between the planktonic and benthic parts of the estuarine system (lower row in Figure 5.5.2) are very important in Texas estuaries. Unfortunately, it is difficult to measure some basic processes such as sedimentation, and it is difficult to monitor the movement of nitrogen between the functional compartments of the sediment. We do know that the processing of nitrogen in the sediment can be greatly influenced by the activities of benthic clams and worms that mix the upper level of the sediment (Blackburn and Henricksen 1983; Flint et al. 1983; and Section 5.4).

In many estuaries, deltaic marshes and salt marshes that fringe the bays play a long-term role in nutrient cycling. During some seasons and tidal conditions they convert dissolved nutrients into biomass; under other conditions, they affect a net release of nutrients to the bays through tidal exchange. Studies have been done to quantify rates of nutrient processing by salt marshes (cf. Armstrong et al. 1975). However, with the relatively small tidal amplitude along the Texas coast, the tidal exchange of materials between marsh and bay is not as important to the entire system as is the case in some other coastal environments.

Recent studies of nitrogen processes in Texas estuaries have greatly increased the understanding of rates of material exchange among nitrogen cycle compartments. In the following discussion, the results of these studies are presented in the context of the influence of freshwater inflows on the nitrogen cycle.

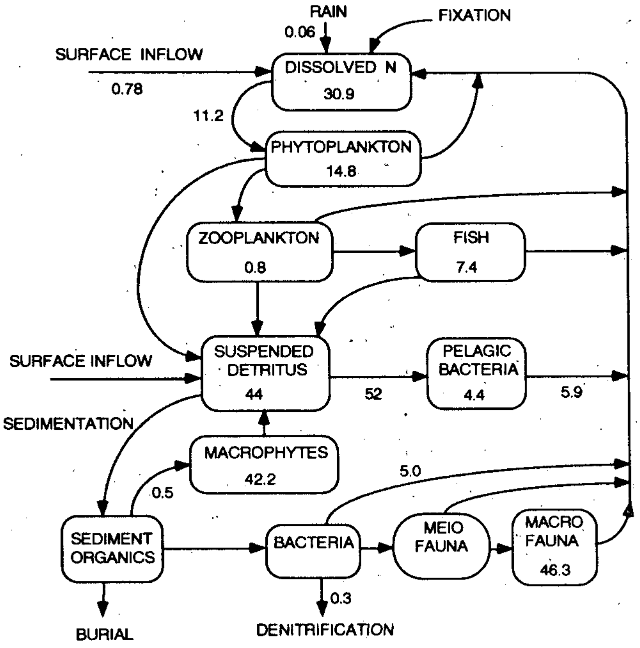


Figure 5.5.3. Nitrogen cycle components in the Guadalupe Estuary during average conditions. Units are μg-at/l for masses and μg-at l^{-1} day^{-1} for rates.

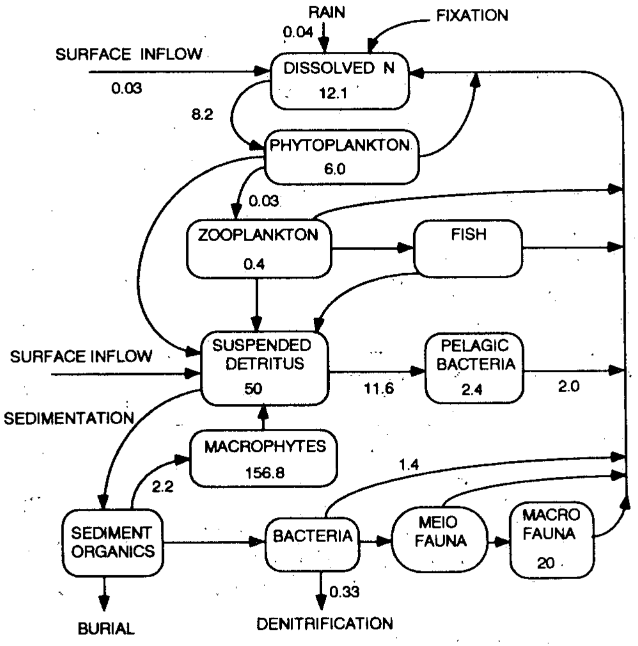


Figure 5.5.4. Nitrogen cycle components in the Nueces Estuary during average conditions. Units are μg-at/l for masses and μg-at l^{-1} day^{-1} for rates.

Comparison of Guadalupe and Nueces estuaries. Elements of the nitrogen cycle in the Guadalupe and Nueces estuaries have been synthesized by Whitledge (1989). The

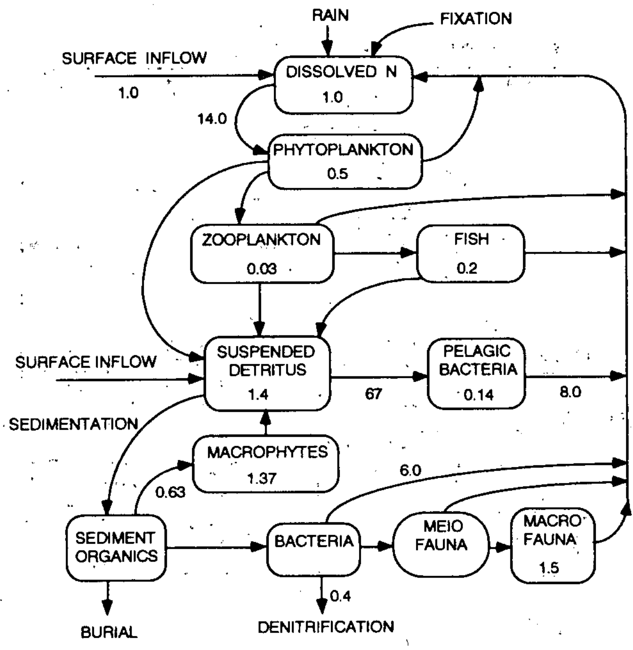


Figure 5.5.5. Guadalupe Estuary nitrogen cycle, average conditions, with flows normalized to the rate of N input from terrestrial sources, and masses normalized to the mass of N dissolved in the water column.

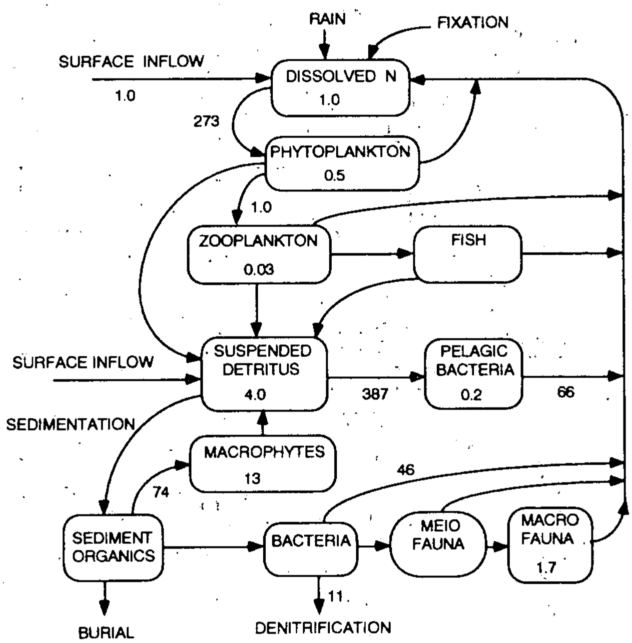


Figure 5.5.6. Nueces Estuary nitrogen cycle, average conditions, with flows normalized to the rate of N input from terrestrial sources, and masses normalized to the mass of N dissolved in the water column.

contrast in typical freshwater inflow volumes to these two estuaries is great. Consequently, one way to assess the influence of freshwater inflows on estuarine nutrient cycling is to compare the differences in aspects of cycling between them. Figures 5.5.3 and 5.5.4 are diagrams of the nitrogen cycle during average inflow conditions in the Guadalupe and Nueces estuaries, respectively, adapted from figures in Whitledge (1989). As in most estuarine studies, more is known about processes and quantities of the dissolved and planktonic parts of the cycle. The Guadalupe Estuary (during the 1986 to 1987 study period) received almost ten times the nitrogen input as the Nueces Estuary (1987 to 1988 period), due to a much larger riverine inflow. This difference in nitrogen loading, however, does not simply translate into equivalently larger rates and quantities within the entire cycle, as shown in the following analysis.

In figures 5.5.5 (Guadalupe Estuary) and 5.5.6 (Nueces Estuary), nitrogen masses and exchange rates (from Whitledge 1989) have been normalized to facilitate comparison. Exchange rates were normalized to the rate of surface water input and are expressed as the ratio of measured daily process rate to the rate of the nitrogen input from combined freshwater sources. Masses were normalized to the mass in the dissolved nitrogen pool. Therefore, storage and flow quantities in the figure are dimensionless ratios, not actual loading values. But this procedure enables us to clearly see differences between the systems. Notice that, with the exception of macrophytes, the normalized nitrogen masses in the biological compartments are similar in the two estuaries. In contrast, the flux rates are generally much higher in the Nueces Estuary than in the Guadalupe Estuary. Both the flux between dissolved nitrogen and phytoplankton and the flux between pelagic bacteria and dissolved nitrogen are two orders of magnitude higher in the Nueces than in the Guadalupe Estuary. This indicates much higher rates of use of regenerated nitrogen in the Nueces Estuary. Relative to the rates of input from outside the system, rates of remineralization and denitrification are higher in the Nueces Estuary, even though the absolute mass of nitrogenous materials processed is greater in the Guadalupe Estuary. Two reasons may account for this: in the Guadalupe Estuary, biological processes may be overwhelmed by physical flushing and sedimentation rates; or, in the Nueces Estuary, conditions may be more stable, allowing the development of an efficient chain of nutrient processes.

Unfortunately for this comparison, the results of these studies may not represent an average, long-term difference between these estuaries. The Guadalupe Estuary was studied during a period of higher-than-normal inflow (1987), whereas the Nueces Estuary was studied during low inflow conditions (1988). Further complicating the comparison, the Nueces estuary had received flood-flow inputs during the year previous to the measurements. Nueces data may therefore be more indicative of the long-lasting influence of an input event rather than of normal nitrogen processing.

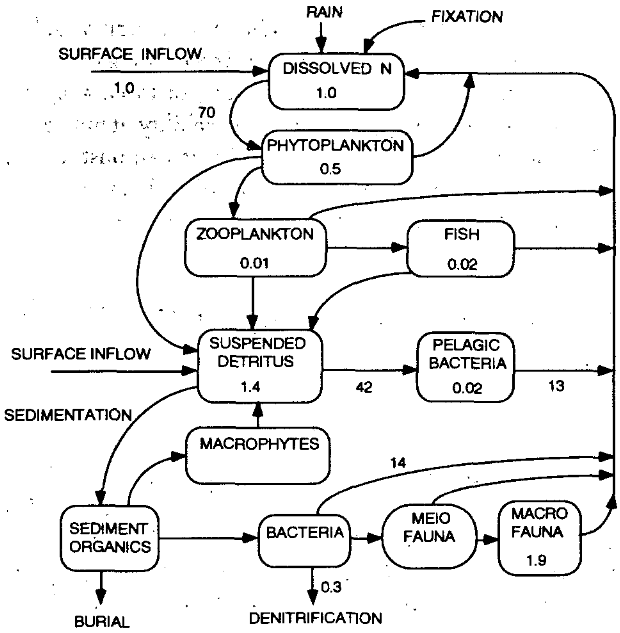


Figure 5.5.7. Guadalupe Estuary nitrogen cycle, normal freshwater inflow conditions, with flows normalized to the rate of N input from terrestrial sources, and masses normalized to the mass of N dissolved in the water column.

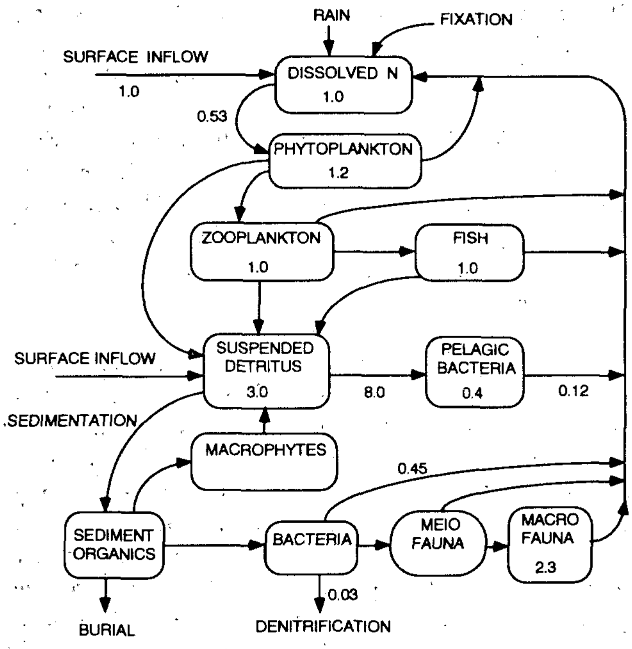


Figure 5.5.8. Guadalupe Estuary nitrogen cycle, high freshwater inflow conditions, with flows normalized to the rate of N input from terrestrial sources, and masses normalized to the mass of N dissolved in the water column.

Contrast between high and low inflow conditions. During the period of the Guadalupe Estuary study (fall 1986 through summer 1987), the inflows to the estuary shifted from normal to very high. Data collected during this period provide a contrast between nitrogen processing in an estuary during seasons of very different nitrogen input rates. Figures 5.5.7 (normal inflow) and 5.5.8 (high inflow) display differences in the nitrogen cycle in the Guadalupe Estuary during these two periods. Data are from figures in Whitledge (1989), normalized to input rates and water column dissolved nitrogen mass.

Nitrogen masses within compartments during the high inflow period are higher than during the period of lower input rates. This could be in part a function of seasonal differences in biological population growth rates since the normal inflow occurred in November, while the high inflow condition occurred in July. Rates of nitrogen processing appear to be substantially higher during the normal inflow period than during the period of high freshwater inflow. Since the high inflow period represents a major flood, physical removal of organisms or low salinity stress on the estuarine biota may have reduced the efficiency of the cycling interactions during this period.

Inflow Effects on Nitrogen Processes

Other sections in chapters 5 and 6 discuss the influence of salinity variations and inflow fluctuations on plankton, benthos, shellfish, and finfish. Nitrogen cycling is a system activity, involving all trophic levels; the effects of inflow variation on any part of this ecosystem could affect processing of this nutrient. Since microbial activity is so important in nitrogen dynamics, the following discussion of the influence of inflows and salinity variation emphasizes effects on microbial activities.

Denitrification. The strongest determinant of denitrification rates reported by Benner and Yoon (1989) was temperature; rates were higher during the summer. They reported no correlation between denitrification rate and salinity at the sampling date and location. In addition, denitrification rates determined for sites in the Guadalupe and Nueces estuaries were very similar (Benner and Yoon 1989). Since these estuaries display different average salinity regimes, this also suggests that denitrification rates in Texas bays may not be influenced by salinity. The upper estuary sites were found to have higher denitrification rates than sites farther downstream, but this probably reflects a greater availability of nitrogenous substrates in the upper estuary, as opposed to more favorable salinities.

Nitrification. For the Potomac Estuary, Elkins et al. (1981) showed evidence that at high inflow rates, the second part of the nitrification process—oxidation of nitrite to nitrate—was inhibited. Because nitrite is more reactive than nitrate, they suggested that the inhibition could be important to the estuarine nitrogen cycle, possibly permitting higher rates of sedimentary denitrification. Limited measurements of nitrification in San Antonio Bay did not show inhibition of nitrification following floodwater inflows (Benner and Yoon 1989).

Water column nitrogen regeneration. The processes of nitrogen uptake by phytoplankton and nitrogen regeneration by zooplankton and pelagic bacteria appear so closely coupled that measurable concentrations of dissolved nitrogen may remain low in bay waters even when phytoplankton show strong growth (Nixon and Pilson 1983). Rates of water column regeneration were similar in Nueces and Guadalupe estuaries, and within the range of rates reported for many other estuaries (Benner and Yoon 1989). At sites farther from the river mouth, nitrogen regeneration in the water column contributes as much or more available nitrogen than benthic regeneration (Benner and Yoon 1989).

Benthic ammonium flux. Montagna et al. (1989) reported no difference in benthic ammonium flux between dates and stations in the Guadalupe Estuary. There was no correlation between rates and salinity. Benner and Yoon (1989) found no large differences in benthic flux between the Guadalupe and Nueces estuaries. However, Montagna et al. (1989) stated that the Guadalupe Estuary had a higher potential for response to nitrogen inputs than the Nueces Estuary, since the former had higher benthic populations. Therefore, the major influence of freshwater inflow on exchange of ammonium from sediments to the water column may be indirect, as the inflow affects populations of benthic organisms.

Conclusion

Recent studies have quantified many aspects of carbon and nitrogen cycling in Texas estuaries. Major influences on these cycles include the external supply of detritus, and the exchange between benthos and planktonic communities. Rates of processes important to nitrogen cycling were observed to differ in the Guadalupe Estuary between high-flow and low-flow periods. In general, nitrogen flux rates between compartments were higher per unit of nitrogen input during normal inflow than during high-inflow periods. In contrast, nitrogen masses in system compartments were greater per unit of dissolved nitrogen in the water during high-inflow periods than during low-inflow periods.

Differences in process rates were also observed between the fresher Guadalupe Estuary and saltier Nueces Estuary. Fluxes between estuarine system components per unit of nitrogen input were higher in the Nueces Estuary than in the Guadalupe Estuary, although nitrogen masses per unit of dissolved nitrogen were about the same for the various system components (except macrophytes). It is difficult to determine from current information whether these differences can be attributed to a controlling influence of fresh water on the biological system, or to an adaptive response of the biotic system to the availability of nutrients. The features of the nutrient cycles that demonstrate an estuary's response to changing volumes of fresh water may be the most characteristic features in Texas estuaries.

5.6 INFLUENCE OF INFLOW ON CARBON INPUT FROM DELTAS AND UPLAND AREAS

Introduction

Previous studies. In some previous studies of Texas bays, efforts were made to measure the movement of dissolved and particulate organic and inorganic material between delta wetlands and the bay waters. Laboratory studies were undertaken to measure the exchange between delta marsh plants and soils and the floodwater draining from the marsh surface (Armstrong et al. 1975; Dawson and Armstrong 1975; Armstrong and Brown 1976; Armstrong and Gordon 1977a, 1977b; Armstrong et al. 1977). In addition, field studies of exchange were done in the Trinity River Delta (Belaire and Price 1977) and in the Lavaca, Guadalupe, and Nueces river deltas (Wiersma et al. 1977). No consistent pattern of movement of materials between the marsh and the bay waters was seen. Some materials were imported into various marshes but exported from others; some materials were imported during one or more seasons of the year but exported during other seasons. It is possible that most material movement is episodic, occurring during major floods or storms, but no direct measurements are available to substantiate this possibility.

The lack of a consistent pattern of material movement is not unique to the delta marshes of Texas bays. In a review of 20 years of research on the role of marshes in estuarine productivity and water chemistry, Nixon (1980) noted that there was no consistent evidence that marshes were strong sources or sinks for nutrients in coastal nutrient cycles. The flux of materials in areas with strong tidal influence was not as large as originally thought, so the lack of clear exchange patterns in Texas bay delta marshes that have low tidal fluxes is not surprising. Most studies have measured very small changes in the concentration of materials moving into and out of marshes from one tidal cycle to the next, and have estimated flood and discharge volumes to calculate mass exchange. These estimates may be subject to large errors (Nixon 1980). In addition, the small calculated exchange quantities can be overwhelmed by large flows that occur during storms and floods when it is very difficult to assess transport.

Most studies have focused on the role that adjacent wetland areas may play in supplying carbon to the bays. Only a few studies have considered the effect that terrestrial carbon from upland sources may have in bay ecosystem operation.

Stable isotope methods. To avoid some of the short-term measurement problems, stable isotopes have been used to measure material movement over a longer term. Carbon has two stable forms in nature, ^{12}C with an atomic weight of 12 and ^{13}C with an atomic weight of 13; about 99% of the world's carbon is ^{12}C. It is possible to accurately measure the ratio of these two isotopes in any organic material with specialized analytical equipment. Measurements are usually presented as $\partial^{13}C$ units, the difference in parts per thousand between the isotope ratio of a sample and the ratio of a particular limestone used as a standard by most laboratories. Negative values of $\partial^{13}C$ mean that the sample contains less ^{13}C than the standard. The more negative the value, the less ^{13}C it contains; the more positive the value, the greater ^{13}C it has.

Plant species have distinctive $\partial^{13}C$ values that are related to each species' physiology and habitat. Many terrestrial species have $\partial^{13}C$ values of -30 to -25, phytoplankton have values of -22 to -18, benthic algae have values of -20 to -13, many wetland species have values of -15 to -11, and seagrasses have values of -12 to -6 (Fry and Scherr 1984). Because of the differences in isotope ratios of plant groups, it is theoretically possible to use the ratios to trace the movement and distribution of plant organic material in the bays. For example, if a bay has two major sources of carbon, terrestrial carbon (-30) that is carried into the bay with river flow and phytoplankton produced in the bay (-20), the organic carbon in the sediments will show a gradient of $\partial^{13}C$ values from the river into the bay between these two extremes.

While this tracing method appears simple in concept, interpreting the results can be complicated. If carbon sources in a bay have three or more distinct isotope ratios (-30, -20, and -10, for example), it may be impossible to distinguish a mixture of carbon from the two extreme sources (-30 and -10) with carbon from the third source (-20) without additional information. The isotope ratios of the source materials also may change over time. For example, the isotope ratios of the major chemical components of smooth cordgrass (*Spartina alterniflora*) detritus are not uniform (cellulose = -11.8, hemicellulose = -11.6, and lignin = -17.4; Benner et al. 1987). Since lignin decays more slowly than the other components, the isotope ratio of cordgrass detritus tends to become lighter and more like phytoplankton carbon through time (Benner et al. 1987; Fogel et al. 1989). Even with these complications, the possibility of tracing the sources of carbon in the bay is an attractive tool and might provide important insights into bay functioning.

Carbon Sources and Use in Lavaca, San Antonio, Nueces, and Corpus Christi Bays

Carbon isotopes of bay-bottom sediment, suspended particulate organic matter (POM), fish, and shellfish were measured in Lavaca Bay by Jones et al. (1986), and in San Antonio, Nueces, and Corpus Christi bays by Parker et al. (1989). Because sediment accumulates through time, the $\partial^{13}C$ of bay-bottom sediment is a long-term indicator of input of various carbon sources into the bay. The $\partial^{13}C$ of POM is a short-term measure of the bay carbon sources; the carbon may come from recent phytoplankton production or be imported organic matter from the river, marshes, adjacent waters, or a combination of these sources. Organisms generally reflect the $\partial^{13}C$ of their foods, although biochemical processes during assimilation and metabolism result in slightly more negative $\partial^{13}C$ values for organisms than their foods. Since fish and some shellfish are mobile and usually selective of the materials they ingest, their $\partial^{13}C$ values indicate their food source or the habitat in which they live.

Table 5.6.1. Areas and combined plant community $\partial^{13}C$ values for organic material available for export from the Lavaca, Guadalupe, and Nueces deltas. Based on area, quantity of exportable dry organic matter, and species composition information from Adams (1977) and Wiersma et al. (1977); $\partial^{13}C$ values for individual species from Haines (1977), Fry and Sherr (1984), Jones et al. (1986), Pulich and Scalan (1987), Parker et al. (1989), and W.M. Pulich (Texas Parks and Wildlife Department, Austin, Texas; pers. com.).

Delta name	Delta wetland area (ha)	Percent of marsh with C_4 plants	Combined $\partial^{13}C$
Lavaca Delta	4,524.82	73.9	-14.8
Guadalupe Delta	3,321.75	72.6	-16.3
Nueces Delta	5,014.08	70.2	-13.9

Long-term input to the sediment. The Lavaca, Guadalupe, and Nueces river deltas have marshes ranging in size from 3,322 to 5,014 ha (8,208 to 12,390 acres, Table 5.6.1). Rivers generally transport carbon from terrestrial sources with $\partial^{13}C$ values ranging from -30 to -25. Marsh plants, however, do not have uniform $\partial^{13}C$ values; the ratios of some plants are in the -27 to -24.5 range, while others are heavier, in the -15 to -11 range. More than 70% of the area and exportable plant biomass in the three deltas is from wetland plants with $\partial^{13}C$ values in the -15 to -11 range. When information derived by Adams (1977) and Wiersma

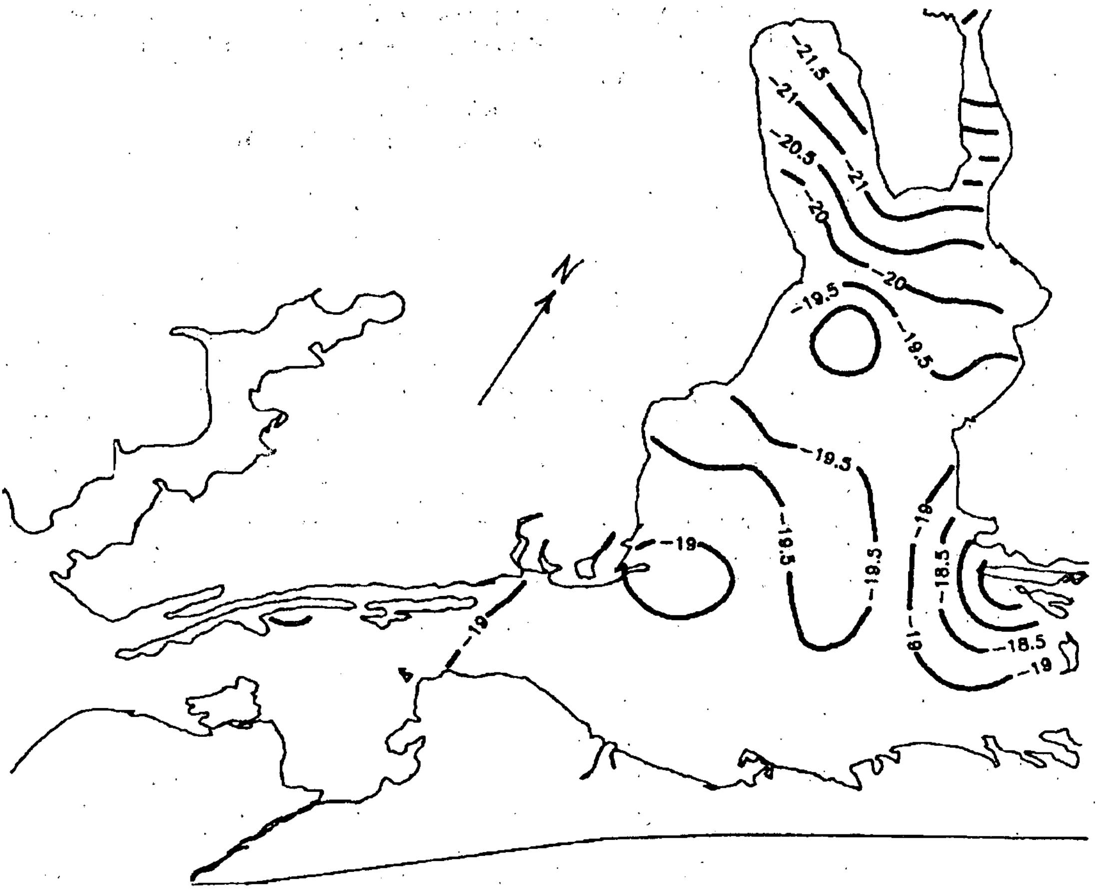


Figure 5.6.1. San Antonio Bay sediment $\partial^{13}C$ values (from Parker et al. 1989).

et al. (1977) on area, quantity of exportable dry organic matter, and species composition is analyzed, the $\partial^{13}C$ values of the exportable carbon of the combined plant communities show a range from -16.3 to -13.9 (Table 5.6.1). If a river transports any of this heavier, marsh-produced carbon, $\partial^{13}C$ values of sediment carbon in the river or close to the river mouth in the bay will be less negative than terrestrial carbon alone. The $\partial^{13}C$ values of samples from the Lavaca River to just below the Lavaca Delta ranged from -22.3 to -18.7 (Jones et al. 1986); in San Antonio Bay, the sediment $\partial^{13}C$ value at the river mouth was -23.6 (Parker et al. 1989). These values are consistent with the transport of some marsh plant material along with terrestrial carbon to the bay over the long term. However, phytoplankton production in the rivers, creeks, and lakes in the deltas could also provide carbon with isotope ratios in the same range.

While the evidence about river transport of delta marsh carbon to the bays is equivocal, it is clear that the Guadalupe River carries organic matter from terrestrial sources into San Antonio Bay. Figure 5.6.1 (from Parker et al. 1989) is a map showing contours of sediment $\partial^{13}C$. The gradient of $\partial^{13}C$ from river influence is clear in Guadalupe Bay and around the delta, but diminishes in the upper portion of San Antonio Bay. The range of river influence on sediment carbon is not as clear for Lavaca Bay and appears to be very minor for Nueces Bay (Parker et al. 1989). The lack of a clear river signal in Nueces Bay is consistent with the usual low gaged and ungaged flow to this bay, while the intermediate signal for Lavaca Bay correlates with the moderate inflows of the Lavaca River.

Short-term distribution of carbon. POM, the organic matter suspended in the water column, consists of terrestrial or marsh detritus, plankton, seagrass and benthic algal detritus, and resuspended organic matter from the bottom sediment. The distribution of $\partial^{13}C$ values for POM in Lavaca, San Antonio, Nueces, and Corpus Christi bays is shown in Figure 5.6.2 (after Parker et al. 1989). More than half of the POM from the river samples in the Lavaca Estuary has $\partial^{13}C$ in the -27 to -23 range. Since terrestrial organic matter is the only large carbon source with values more negative than phytoplankton carbon (-22 to -18), any $\partial^{13}C$ values more negative than -22 are definite indicators of the input of carbon from terrestrial sources. About 25% of the POM samples from Lavaca Bay waters were in the -25 to -23 range, which indicates a terrestrial origin for some of these materials.

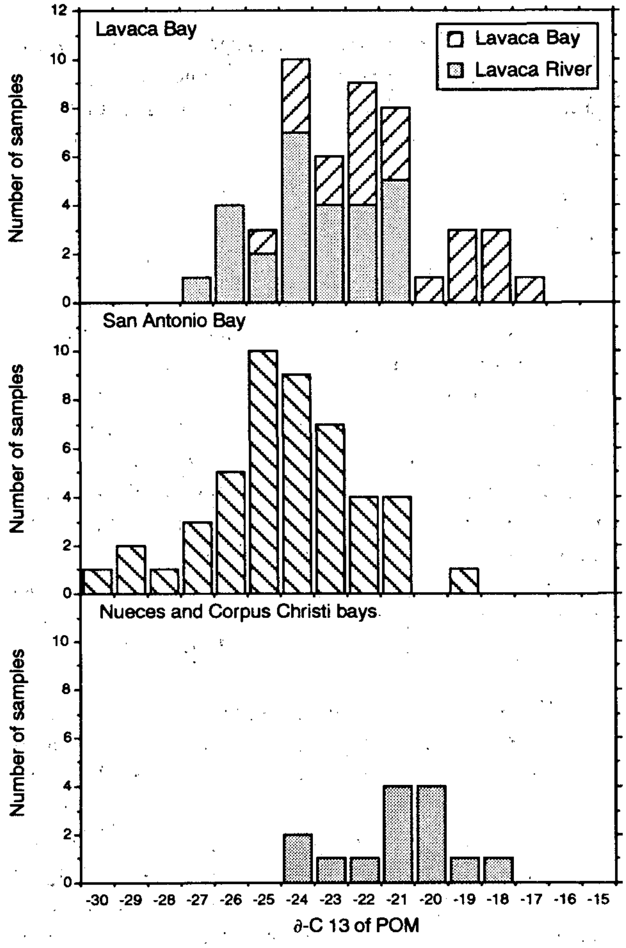


Figure 5.6.2. Particulate organic matter (POM) $\partial^{13}C$ values for Lavaca, San Antonio, Nueces, and Corpus Christi bays (after Parker et al. 1989).

More than 80% of the POM samples in San Antonio Bay had $\partial^{13}C$ values in the -30 to -23 range (Parker et al. 1989). The mean value of samples from the inner bay (close to the river mouth) had nearly identical $\partial^{13}C$ values compared to outer bay stations; and POM samples collected near the Intracoastal Waterway and Matagorda Island, 16 to 24 km (10 to 15 mi) from the river, had unmistakable terrestrial carbon signatures.

Fewer POM samples were collected in Nueces and Corpus Christi bays and the concentration of seagrasses (-12 to -6) makes it difficult to detect terrestrial carbon input. However, a few samples were collected in the -24 to -23 range, which indicates a terrestrial carbon source. The stations where these samples were collected were in Nueces Bay and in Corpus Christi Bay near the Nueces Bay Causeway.

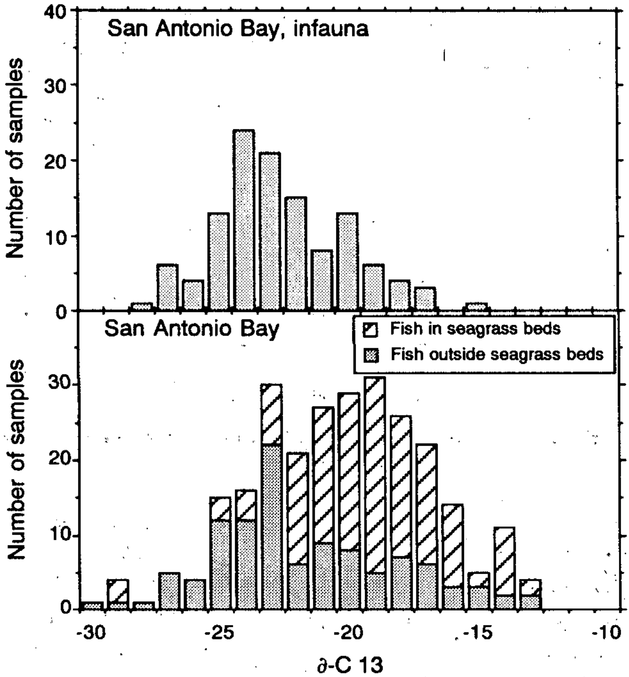


Figure 5.6.3. San Antonio Bay $\partial^{13}C$ values for infauna and fish (after Parker et al. 1989).

The 1987 gaged inflows in the Guadalupe basin were the highest in the past 47 years. The terrestrial carbon detected throughout San Antonio Bay was undoubtedly transported by the large gaged and ungaged inflows and shows that terrestrial carbon (and possibly delta marsh carbon) can be spread throughout the bay. The Lavaca and Nueces data also confirm that this same type of terrestrial carbon enters the bay. While terrestrial carbon is occasionally spread throughout the system, its signature in the sediment carbon is not always obvious.

Use of imported organic matter by biota. The isotopic composition of biota reflects the materials they ingest and assimilate. Since the life cycle of most larger organisms is measured in terms of months or years, the isotopic signature of the carbon in their tissues is an integrated measure of the carbon sources used during their life span. Benthic infauna are generally small invertebrates that live in the top few centimeters of the sediment. Many of the animals in the infauna ingest organic material suspended in the bottom waters. Infauna collected between October 1986 and July 1987 in San Antonio Bay had $\partial^{13}C$ values ranging from -28 to -15 (Figure 5.6.3, after Parker et al. 1989). More than half of the samples collected had values between -28 and -23, which indicates that a significant proportion of their carbon came from terrestrial sources. Since POM values in San Antonio Bay water ranged from -30 to -19 during that period, it is not surprising that the terrestrial carbon signal is evident in benthic organisms that directly filter particulate matter. Note that the distribution of $\partial^{13}C$ values for infauna is less negative than the POM distribution (Figure 5.6.2). The infauna are probably using some of the less negative sediment carbon in addition to the POM.

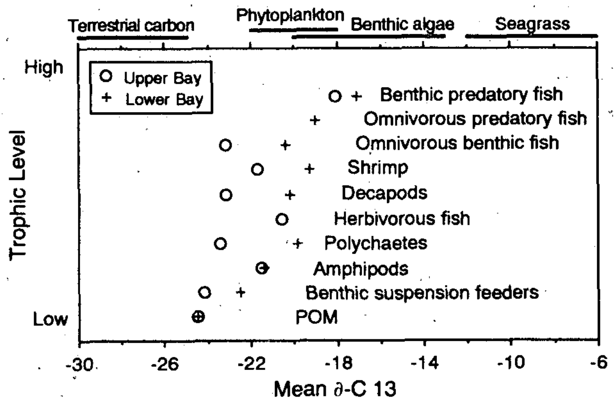


Figure 5.6.4. Mean ∂^{13}C values of trophic groups in the inner and outer portions of San Antonio Bay (after Parker et al. 1989).

Assessing the use of carbon by higher-trophic-level organisms can be complicated because the animals are mobile and have access to carbon in their foods with a wide range of ∂^{13}C values. Physiology can also affect ∂^{13}C values because metabolic processes may favor the use of one isotope over another. In general, the farther removed an animal is in the food chain from plant carbon sources, the more positive its ∂^{13}C values will be compared to the carbon sources (Fry and Sherr 1984). Fish samples collected in San Antonio Bay are presented in Figure 5.6.3 (from Parker et al. 1989) for comparison with the infauna. Overall, fish display a very wide range of values of assimilated carbon (-30 to -13). About one-third of the fish captured in seagrass beds have values greater than -18, the upper limit of phytoplankton carbon. A substantial portion of their carbon must come from seagrasses, benthic algae, or animals that feed on these plant materials. Fish captured out of seagrass beds show the widest range of ∂^{13}C values, but more than 50% of the samples collected had values in the -30 to -23 range, indicating that a portion of their foods came directly or indirectly from terrestrial carbon. Parker et al. (1989) showed that a terrestrial carbon signal is detectable in Lavaca Bay fish during years of smaller inflows but is absent from the Nueces and Corpus Christi bay fish, which are strongly influenced by seagrass or benthic algal carbon.

The influence of terrestrial carbon may be best illustrated by examining the ∂^{13}C values for a variety of organisms from different trophic levels in San Antonio Bay. The mean values of upper bay benthic suspension feeders, polychaetes, decapods, shrimp, and omniverous benthic fish show definite signs of terrestrial carbon influence with mean values equal to or less than -23 (Figure 5.6.4, after Parker et al. 1989). They also show a shift to enriched ^{13}C with higher trophic levels. Even though most of the organisms in the lower bay trophic levels do not show the clear-cut terrestrial signal of the upper bay groups, their mean values could be interpreted as a mixture of terrestrial and phytoplankton carbon with a shift due to trophic fractionation. Except for predatory fish, it appears that the influence of seagrass or benthic algal carbon is less than terrestrial or phytoplankton carbon.

Conclusions

During a year of high freshwater inflows in San Antonio Bay, terrestrial carbon was distributed throughout the bay as POM and was traceable in food webs to fish and shellfish that are the subject of the state's freshwater inflow management efforts. Terrestrial carbon was traceable in several Lavaca Bay trophic groups during the two years previous to the high-flow year in San Antonio Bay, but the terrestrial material was not distributed as widely in POM during these years of lower freshwater inflow. Terrestrial carbon transported by freshwater inflow does not seem to play a large role in the carbon budget of Nueces and Corpus Christi bays where phytoplankton, seagrass, and benthic algae appear to be more important carbon sources.

The ∂^{13}C value of sediment carbon in open bays does not provide an accurate picture of the value of inflow-transported terrestrial carbon in Texas estuaries. The sediment ∂^{13}C values range from -20 to -18, the midpoint between extreme values for bay carbon sources. The bays have four or five carbon sources with ∂^{13}C values ranging from -30 to -6. Mixing and physiological processes associated with metabolism and carbon recycling tend to drive the sediment ∂^{13}C values toward the middle of this range, which is coincidental with phytoplankton isotope ratios. Without additional stable isotope tracer information, it is not possible to use sediment organic matter to evaluate the long-term contribution of various carbon sources to the bay system. Nevertheless, the studies by Jones et al. (1986) and Parker et al. (1989) have shown that terrestrial carbon carried by freshwater inflow from rivers can occasionally be distributed throughout the bay system. Some organisms directly use this river-transported terrestrial carbon when it is available, and the terrestrial carbon signal can be traced to higher levels of the food web in the bays. Organisms use river-transported terrestrial carbon, but it is not yet possible to evaluate its importance to bay carbon budgets.

5.7 CONCLUSIONS

Primary producers. Both phytoplankton and estuarine vascular plants show direct responses to freshwater inflows. During periods of low inflows and high salinities, bay phytoplankton are dominated by diatoms; during high-inflow and low-salinity periods, flagellates are frequently the most numerous phytoplankton species. Changes in phytoplankton species composition occur rapidly with increases in freshwater inflows, generally in a matter of days. The return of phytoplankton populations to their pre-inflow status is generally rapid, depending on the resumption of previous salinity conditions.

Vascular plant species composition varies in response to salinity change due to freshwater inflows. Freshwater and low brackish marsh and submerged communities in the upper regions of estuaries and in the river deltas show a critical dependence on low salinity conditions and inundation from flooding. Submerged plant communities in the lower estuary can change from obligate halophyte seagrasses to the euryhaline widgeongrass when inflows result in salinities of less than 20‰. Unlike phytoplankton, however, the period of exposure of vascular plants must be prolonged (a growing season of several months or longer) before plant species' composition changes. Vascular plants integrate the effects of salinity, nutrients, underwater light levels, and sediment inputs over a long period of time.

The effect of inflow on phytoplankton productivity is more difficult to generalize. Increased loading of nutrients by inflows can stimulate phytoplankton growth, but the close coupling of phytoplankton and zooplankton consumers can make it difficult to show from abundance data alone that phytoplankton production has been stimulated. Complicated productivity measurements are usually needed, and a large historical base of this information is not available. With very high freshwater inflows, displacement of phytoplankton from the bay by high flows may obscure the effects of enhanced production.

Vascular plants also display a complicated productivity response to increased inflows. To some degree, productivity is stimulated by the heightened nutrient availability, and in the case of intertidal wetland plants, added soil moisture. Production of submerged species may be dampened by light limitation due to increased turbidity, stimulation of epiphyte growth, and algal blooms that may accompany inflow increases. Sediments transported in the inflows are also critically necessary to maintain the proper elevations for production of both intertidal emergent and submerged species. This factor assumes added significance in those bay systems currently undergoing land subsidence.

The high degree of variability in phytoplankton abundance, much of which is unrelated to freshwater inflow, makes phytoplankton abundance an uncertain measure of the effect of inflows. Vascular plant abundance is potentially a better gage of the effects of freshwater inflows because of the stationary nature of rooted plant communities. Sample-to-sample variability is reduced compared to phytoplankton. However, we do not have adequate site-specific measurements for Texas estuaries where inflows have been monitored to provide quantitative relationships between inflow and vascular plant abundance. Nevertheless, information on salinity limits, preferences, and optima of target estuarine species can profitably be used to specify ranges of acceptable salinity conditions needed to sustain vascular plant growth in estuaries.

Consumers. Zooplankton and benthic organisms are major consumers of the phytoplankton produced in Texas estuaries and the organic matter imported into them from terrestrial and delta marsh sources. Both consumer communities show distinct responses to freshwater inflows. Zooplankton species composition depends on the salinity and level of inflow. Tidal and brackish portions of rivers flowing into estuaries usually have a mixture of estuarine and predominantly freshwater zooplankton species. After flood flows, freshwater species may abound in the upper reaches of estuaries until salinities begin to rise. In general, zooplankton abundance is greater in areas away from deltas and river mouths, where salinities are higher.

Production of micro- and macrozooplankton appears to be stimulated by the occurrence of freshwater inflow pulses, although there is a point at which the displacement of macrozooplankton by water flowing through and out of the estuary seems to counterbalance production increases. In estuaries that normally receive inflows of five or six bay volumes per year, inflow increases above this level reduce annual zooplankton abundance. In estuaries that receive inflows of less than one bay volume per year, large increases in inflow are accompanied by large increases in annual zooplankton abundance.

Benthic communities also respond to increases in inflows with changes in species composition. Meiobenthic populations of nematodes decrease in density and diversity, but juvenile mollusk population numbers increase substantially. Macrobenthic organisms adapted to brackish conditions may flourish with high inflows, while, typically, the more marine species with narrower salinity tolerances are decimated until higher salinity conditions return. Recovery of the more marine benthic populations after large inflows may take months, but populations often rebuild to elevated abundances and then decline to pre-inflow levels. An incompletely tested hypothesis for regulation of benthic populations by inflows has been suggested, though not confirmed.

Other measurements of benthic community response have also shown marked effects due to freshwater inflow. The rate of consumption of bacteria and microalgae by meiofauna increased dramatically after increases in inflow, probably as the result of high microalgal and bacterial production. In addition, the rate of nutrient regeneration by the benthic community seems to be directly influenced by freshwater inflow. Regeneration rates increase during and immediately after high inflows, although unusually high or prolonged inflows may result in very low regeneration rates.

Zooplankton have many of the same drawbacks as phytoplankton as quantitative indicators of freshwater inflows. Zooplankton populations are highly variable due to local turbulence from currents, diurnal vertical migration, and generally patchy distribution. Even though zooplankton data for a few years is available for several estuaries, variability and the lack of a clear understanding of the regulation of zooplankton populations by inflows hampers use of zooplankton as an indicator of inflow effects.

There is very limited information available about micro- and meiobenthic organisms, but macrobenthos changes may provide a useful measure of the effect of inflows. Insufficient data is available to elucidate inflow-population cycles and provide quantitative relationships for assessment purposes. Information on salinity limits, preferences, and optima is available for many benthic species, and distribution patterns are generally known. As in the case of vascular plants, this information can be used to select acceptable salinity ranges in various regions of estuaries that would nurture target benthic species production.

Nutrient cycling and use of terrestrial and delta carbon. Aspects of the nutrient cycling process appear to depend on the level of freshwater inflow. In general, the components of an estuarine ecosystem that use and store nitrogen assimilate higher levels of this nutrient during periods of high inflow than during periods of low inflow. In contrast, the efficiency of nitrogen transfer between ecosystem components per unit of nitrogen input to the system decreases during high inflows. During periods of low inflows, the efficiency increases and the material is cycled more times before being lost to burial, denitrification, harvest, or export to the Gulf. While our knowledge of nutrient cycling is not yet complete enough to provide an analytical tool for evaluating freshwater inflows, some of the relationships between inflow and various nutrient processes can be used to create a nutrient budget that is useful in assessing inflow requirements (see sections 7.3 and 7.4).

Stable carbon isotope measurements of biota have shown that terrestrial or delta-produced carbon can be traced up the food chains to higher trophic levels. At times following large inflows, terrestrial or delta-produced particulate organic material has been observed throughout the estuary. The terrestrial and delta carbon signal in the sediment is somewhat obscured since sediment isotope measurements average the effects of carbon from various sources through time, and are affected by the results of differential degradation of materials during catabolism. Nevertheless, sediment isotopes clearly show that terrestrial and delta carbon is carried by inflow some distance into the estuary from the river. Stable carbon isotope information does not provide an analytical tool for evaluating inflows, but it does show that inflow brings organic material into the estuary and this material is utilized by the organisms living there.

CHAPTER 6: THE RELATIONSHIP BETWEEN INFLOW AND RELATIVE ABUNDANCE, DISTRIBUTION, AND PRODUCTION OF FISH AND SHELLFISH

6.0 INTRODUCTION

While the previous chapter concentrated on the relationship between inflow and the lower levels of the estuarine food chain, this chapter will focus on the organisms that are "economically important and ecologically characteristic of sport or commercial fish and shellfish species" [TEXAS WATER CODE 11.147(a)]. Generally, these are the animals that most people associate with Texas bays and estuaries, especially for commercial and recreational fishing.

As noted in an earlier section (5.3), most of the larger fish and invertebrates begin life as part of the plankton, but eventually grow large enough that they can no longer be considered planktonic. Many of these animals change their feeding behaviors and preferred habitats at various times during their life cycles. Therefore, to get a complete picture of the effects of freshwater inflow on these species, we must consider the effects of inflow on various life stages and sizes of animals.

Distribution and abundance are among the most familiar kinds of measurements made of juvenile and adult finfish and shellfish. This type of information is useful in an overall sense because it provides a clear indication of the use and avoidance of particular sites and habitats. In some instances, it is clear what environmental conditions govern the use of particular areas. But often, detailed studies involving physiological measurements of reactions to environmental conditions that may be affected by inflow are needed to reveal the relationship between inflow, distribution, and abundance.

The major goal of this chapter is to demonstrate whether there are clear relationships between inflow and the abundance, distribution, and production of fish and shellfish. Where possible, the relationships will be quantitatively expressed so they can be used in an assessment methodology to determine inflow needs of fish and shellfish.

Larval transport through barrier island passes. A number of species spawn their young in the Gulf. The eggs and larvae, which are members of the meroplankton, are transported toward the estuaries by currents and must be carried through Gulf passes to the bays where they can develop and grow. Section 6.1 presents the results of a study on the transport of larvae which identifies the major factors that appear to influence larval transport; freshwater inflow is one of the factors examined.

Distribution and abundance of larval and juvenile fish and shellfish. Larval and juvenile fish and shellfish have specific preferences for habitat and environmental conditions. Section 6.2 reviews the literature and presents information about preferences and limits for eggs, larvae, and juveniles of several abundant species in Texas estuaries. In addition, catch data showing spatial distribution, abundance, salinity zone, and bottom type preference are presented for several shellfish species in San Antonio Bay.

Wetland habitat use by juvenile organisms. Section 6.3 examines the use of wetland habitats by juvenile organisms. The studies discussed in this section compare the abundance of organisms in wetland and adjacent bare-bottom habitats to evaluate wetland use by estuarine species. Several studies that were reviewed allowed comparison of habitats with similar vegetative structures but different salinity regimes. In addition, an analysis of the effects of flooding on wetland habitat use by estuarine species is presented.

Effect of salinity on adult fish metabolism. Adult fish have the ability to move substantial distances to find suitable food. Unlike many benthic species, they are not at the mercy of local environmental conditions because they are mobile. They can move into unfavorable environments, forage, and move out because of the mobility their high levels of metabolism permit. Section 6.4 examines the metabolism of several adult fish species and describes their metabolic scope—the energy available for physiological processes beyond maintenance levels—as a function of salinity. The salinity range for a species' metabolic scope is an indication of how difficult it would be for the species to survive if the salinity regime were substantially altered as the result of inflow variation.

Effect of salinity on adult fish reproductive development. One aspect of metabolic scope is the amount of energy that can be put into reproduction. Section 6.5 examines the effect of different salinities on reproduction in several fish species. This serves as another indicator of potential species' survival should changes in inflow significantly alter the salinity regime. The section reviews recent experiments on the effects of salinity on adult reproductive development, egg fertilization and hatching, and larval development.

Effect of inflow on adult fish and shellfish abundance. As in the case of larval and juvenile forms, adult finfish and shellfish have particular environmental and habitat preferences. Section 6.6 reviews the preferences and limits documented in the literature and discusses adult distribution and historical relationships between abundance and inflow.

Differences in the relative abundance of fish and shellfish among estuaries. The inflows to the estuarine systems along the Texas coast vary substantially from year to year as does the pattern of inflow throughout the year. The section on vascular plants noted sizable differences in wetland and seagrass habitats among the estuaries that are related to differences in freshwater inflow and the resultant salinity. Given these differences in inflows among the estuaries, Section 6.7 investigates whether there are significant differences in the finfish and shellfish relative abundances among several estuaries. The section discusses observed abundance differences that occur, and evaluates the sensitivity of groups of species to changes in salinity regime.

The effect of inflow on the harvest of adult fish and shellfish. Fishery harvest data for seven major species have been collected for many years. Harvest is analogous to productivity, the performance variable for estuarine organisms specified in the TEXAS WATER CODE 11.147(a), which the water permitting process must maintain. While fishery harvest has its drawbacks as a measure of productivity, data have been collected for several decades over a wide range of inflow conditions. Section 6.8 uses the harvest and inflow data along with information about the life histories of the seven species to create regression equations relating annual harvest to inflow conditions over several seasons or, in some cases, years.

6.1 EFFECT OF FRESHWATER INFLOW ON LARVAL TRANSPORT

Introduction

Many marine fish and shellfish that spawn in the Gulf of Mexico or near Gulf inlets to the bays have eggs, larvae, and juveniles that depend on estuarine nursery habitats for their survival, growth, and development. Most of these young organisms are incapable of strong swimming action and are thus dependent on passive transport by prevailing water currents. Several environmental factors affect transport of these animals, but knowledge of how these forces operate is incomplete. In general, transport of the young depends on prevailing winds, astronomical tides, local rainfall, and circulation patterns. Some of these forces may be affected by or work in concert with freshwater inflows to the bays and estuaries.

A pioneering effort was made to provide insight into the environmental mechanisms governing larval transport through the passes and channels of Matagorda Bay by Darnell and McEachran (1989). Of primary concern were three species of penaeid shrimp, one portunid crab, seven species of sciaenid fish (croaker, drum, and seatrout), and two species of other marine fish (Table 6.1.1). Of secondary concern were additional species of sciaenid fish that occur in Texas. These species were selected on the basis of their economic importance to the state's sport and commercial fisheries. Field collections of larval fish and invertebrates in Matagorda Bay were made from four study sites. Two sites were selected to intercept eggs, larvae, and juveniles passing from the Gulf of Mexico into Matagorda Bay (i.e., the Matagorda Ship Channel and Pass Cavallo). Due to sampling difficulties, the Pass Cavallo station was later deleted. The two remaining sites are located along the west side of Matagorda Bay where connections exist with Espiritu Santo and San Antonio bays (i.e., Saluria Bayou and the Gulf Intracoastal Waterway). These sites were designed to study the transport of eggs, larvae, and juveniles from one bay to another. The biological data were lumped into six major

Table 6.1.1. Classification of study species of fish and shellfish.

Classification	Biological Group	Common Name	Scientific Name
Estuarine	Invertebrates	Eastern oyster[a]	*Crassostrea virginica*
		Blue crab[a]	*Callinectes sapidus*
		Brown shrimp[a]	*Penaeus aztecus*
		Pink shrimp[a]	*Penaeus duorarum*
		White shrimp[a]	*Penaeus setiferus*
Estuarine	Fish	Bay anchovy[a]	*Anchoa mitchilli*
		Darter goby[a]	*Gobionellus boleosoma*
		Gulf menhaden[a]	*Brevoortia patronus*
		Hogchoker[a]	*Trinectes maculatus*
		Naked goby[a]	*Gobiosoma bosci*
		Sharptail goby[a]	*Gobionellus hastatus*
		Striped anchovy[a]	*Anchoa hepsetus*
		Inland silverside[a]	*Menidia beryllina*
Marine	Sciaenid fish	Atlantic croaker[a]	*Micropogonias undulatus*
		Banded croaker	*Larimus fasciatus*
		Black drum[a]	*Pogonias cromis*
		Gulf kingfish	*Menticirrhus littoralis*
		King whiting	*Menticirrhus saxatilis*
		Red drum[a]	*Sciaenops ocellatus*
		Sand seatrout[a]	*Cynoscion arenarius*
		Silver perch[a]	*Bairdiella chrysura*
		Silver seatrout[a]	*Cynoscion nothus*
		Southern kingfish	*Menticirrhus americanus*
		Spot[a]	*Leiostomus xanthurus*
		Spotted seatrout[a]	*Cynoscion nebulosus*
		Star drum	*Stellifer lanceolatus*
Marine	Non-sciaenid fish	Atlantic spadefish[a]	*Chaetodipterus faber*
		Blackcheek tonguefish	*Symphurus plagiusa*
		Fringed flounder	*Etropus crossotus*
		Gulf butterfish	*Peprilus burti*
		Least puffer	*Sphoeroides parvus*
		Scaled sardine	*Harengula jaguana*
		Sheepshead[a]	*Archosargus probatocephalus*
		Southern flounder[a]	*Paralichthys lethostigma*
		Striped mullet[a]	*Mugil cephalus*
		White mullet[a]	*Mugil curema*

[a] Commonly occurs in Texas bays and estuaries.

categories: shrimp, crabs, fish eggs, estuarine fish larvae, marine fish larvae, and marine sciaenid fish larvae. Analyses were also carried out on individual species which appeared in the samples with sufficient frequency.

Stepwise and multiple regressions were used to analyze the relationship between environmental parameters and catch data. Five environmental parameters served as independent variables in the analyses for all sites: depth, temperature, salinity, wind velocity, and current velocity. Tidal height was added as an additional independent parameter for Saluria Bayou, and light and tidal height were included in the Matagorda Ship Channel analysis (see Table 6.1.2).

Results of the Study

Only a few of the observations that were made concerning the recruitment of larval fish and invertebrates were related to salinity, thus indirectly related to freshwater inflow. An added complication was the high variability in the abundance of larval and juvenile organisms in the passes. This is particularly true of fish larvae which are much less abundant than fish eggs or crab larvae.

Prediction of abundance for locality. The abundance of larval and juvenile organisms at one locality cannot be reasonably predicted on the basis of any combination of the

Table 6.1.2. Relationships between environmental parameters and positive and negative regression coefficients.

Parameter	(+) Positive relationship	(-) Negative relationship
Depth abundance (D)	High abundance at the bottom	High abundance at the top
Temperature (T)	High abundance at high temperature	High abundance at low temperature
Salinity (S)	High abundance at high salinity	High abundance at low salinity
Wind direction (W)	High abundance when the wind vector is upstream (i.e., from the Gulf toward the bay)	High abundance when the wind vector is downstream (i.e, from the bay toward the Gulf)
Current direction (C)	High abundance when the current vector is upstream (i.e., from the Gulf toward the bay)	High abundance when the current vector is downstream (i.e., from the bay toward the Gulf)
Light (L)	High abundance during the daylight hours	High abundance during the night
Tidal height (TH)	High abundance at high water levels (calculated from NOAA tide tables)	High abundance at low water levels (calculated from NOAA tide tables)

environmental parameters analyzed from another locality. The findings indicate that biological abundance is differentially influenced at the three stations studied, and that effects on these organisms from environmental parameters should be considered separately for each station.

Environmental parameters were found to be significant in determining biological abundance of larval and juvenile organisms in the passes. The largest number of variables were related to biological abundance in the Matagorda Ship Channel; a smaller set of variables was related to biological occurrence in Saluria Bayou; and the smallest set of variables was related to biological occurrence in the Gulf Intracoastal Waterway (GIWW).

Biological groups at the Matagorda Ship Channel. Results of stepwise and multiple regression analysis of fish and shellfish larvae and egg data from the Matagorda Ship Channel (Table 6.1.3) indicate that the number of statistically significant parameters contributing to biological abundance varies from two (in the case of crab larvae) to all seven (in the case of fish eggs). Low salinity of water in the pass is a parameter that has a direct relationship to freshwater inflow. Low salinity exhibited a significant relationship only to marine sciaenid larvae, the most economically important family of fish on the Texas coast. High salinity was significantly related for fish eggs, estuarine fish larvae, and marine fish larvae.

Multiple regression models relating abundance with all physical environmental variables explained only 9 to 42% of the variation in abundance. Table 6.1.4 shows the direction of the relationship with the variables for each biological group. Since the R^2 values were generally low, the models appear to have low predictive values. Non-independence of the independent variables usually creates inflated, not reduced, R^2 values. One reason for this is that each biological variable includes a group of species whose seasonal appearance and behavior may be quite distinct from one another. Thus, the search for environmental variable correlates within groups characterized by diverse biological phenomena will require more analysis at the species level.

Biological groups at Saluria Bayou. From the stepwise regressions, the number of statistically significant parameters related to biological abundance was less than that observed for the Matagorda Ship Channel, but greater than the number found at the GIWW site (Table 6.1.3). Low salinity was significant for estuarine fish larvae but for no other biological groups. Other significant parameters included up-channel current (fish eggs), depth abundance (shrimp larvae and fish eggs), temperature (fish eggs and marine sciaenid fish), tidal height (fish eggs), and direction (shrimp, fish eggs, and marine fish larvae).

Multiple regression models relating abundance and physical environmental variables explained 11 to 52% of the variance in abundance. As in the case of the Matagorda Ship Channel, the factors controlling the abundance of these biological groups are more complicated than simple linear relationships with the environmental variables.

Table 6.1.3. Environmental parameters significantly related to biological abundance from stepwise regression. The order of significance is numerically ranked with the highest level given a value of 1. Abbreviated information is as follows: C = current direction, D = depth abundance, L = light conditions, S = salinity gradient, T = temperature gradient, TH = tidal height, W = wind direction.

Location and biological group	C	D	L	S	T	TH	W
Matagorda Ship Channel							
Shrimp larvae		3				2	1
Crab larvae	1				2		
Fish eggs	4	5	6	2	3	7	1
Estuarine fish larvae		4	1	3	2	5	
Marine fish larvae		6	1	3	2	4	5
Marine sciaenid fish			3	4	1	2	
Saluria Bayou							
Shrimp larvae		2					1
Crab larvae	(No parameters significant)						
Fish eggs	2	4			3	1	
Estuarine fish larvae		2		1			
Marine fish larvae	(No parameters significant)						
Marine sciaenid fish					1		
Intracoastal Waterway							
Shrimp larvae	1						
Crab larvae	(No parameters significant)						
Fish eggs	(No parameters significant)						
Estuarine fish larvae		1					
Marine fish larvae	1				2		
Marine sciaenid fish	(No parameters significant)						

Biological groups at the GIWW. From the stepwise regression analysis, very few environmental variables were significant at the GIWW site (Table 6.1.3). Although the reason for this is uncertain, the shallow channel is subject to frequent agitation by boat and ship traffic which could confound results because of these disturbances. Prior to entering the GIWW, the eggs and larvae of marine organisms must traverse at least a portion of the lower part of the Matagorda Estuary. This delay in entering the GIWW, coupled with the fact that the organisms are somewhat older and thus more capable swimmers (in the case of larvae), may also have had an effect on the results. In no case did stepwise regression analysis for biological groups of organisms produce a model with more than two variables (Table 6.1.3). Current, depth, and temperature were the only significant factors from this analysis. Crab larvae, estuarine fish larvae, and marine sciaenid fish did not have significant regressions.

Multiple regression models relating abundance to five physical environmental variables explained 14 to 38% of the variance in abundance. Considering the low R^2 values, little of the observed variation in abundance can be explained by simple relationships with these environmental variables.

Summary analysis on biological groups. Darnell and McEachron (1989) assessed the effect of the physical environmental variables on biological groups by comparing the number of positive and negative coefficient signs from multiple regression equations involving all variables. The signs of the coefficients (Table 6.1.4) indicate the direction of the relationship between abundance and the variable. A positive sign indicates a direct relationship, while a negative sign indicates an inverse relationship. They used the z-test for binomial proportions (see Snedecor and Cochran 1967, pp. 211 to 212), with the null hypothesis that the lack of a relationship between abundance and the variables would result in a 50:50 distribution of signs for individual variables. With data from the Matagorda Ship Channel, Saluria Bayou, and ICWW, two variables were statistically significant: upchannel current ($P < 0.05$) and upchannel wind ($P < 0.01$). All other physical variable sign comparisons were not significantly different from the expected ratio. Thus, they concluded that abundance of biological groups in passes and channels correlates most frequently with upchannel current and upchannel wind. The effect was most pronounced on larval forms of shrimp, crab, and estuarine fish.

Individual species analysis. Stepwise regression was used to evaluate the significance of physical environmental variables on the abundance of individual species and larval stages. Three life stages of crabs, four life stages of shrimp, and five larval fish species provided enough data for analysis. The predominant variables that were significant in stepwise regressions included channel current direction, channel wind direction, tidal height, and depth abundance. Low salinity resulting from freshwater inflow was significant in only one instance, the equation for spotted seatrout larvae (*Cynoscion nebulosus*).

Multiple regression analyses on all available environmental variables produced regressions that explained 1 to 97% of the variance. Most regressions explained less than 30% of the variance. Comparing multiple regression coefficient signs for equations involving all environmental variables allowed Darnell and McEachron (1989) to assess the importance of physical factors in transport of individual

Table 6.1.4. Summary of variables resulting in the best statistical regression models, their positive or negative relationships and highest R^2 values for biological groups. All models were derived by omitting zero values. Abbreviated information is as follows: C = current direction, D = depth abundance, L = light conditions, S = salinity gradient, T = temperature gradient, TH = tidal height, W = wind direction.

Location and biological group	C	D	L	S	T	TH	W	Highest R^2 values
Matagorda Ship Channel								
Shrimp larvae	+	+	+	-	-	+	+	0.19
Crab larvae	+	-	+	-	+	-	+	0.09
Fish eggs	-	-	-	+	+	-	-	0.41
Estuarine fish larvae	+	+	-	+	+	+	+	0.41
Marine fish larvae	+	+	-	+	+	+	+	0.42
Marine sciaenid larvae	-	+	-	-	-	+	+	0.20
Saluria Bayou								
Shrimp larvae	+	-		-	-	-	+	0.44
Crab larvae	+	-		+	-	-	+	0.23
Fish eggs	-	-		+	+	+	+	0.50
Estuarine fish larvae	+	+		-	-	+	+	0.11
Marine fish larvae	+	-		+	+	-	+	0.11
Marine sciaenid larvae	+	+		-	-	+	+	0.52
Intracoastal Waterway								
Shrimp larvae	+	-		-	+		+	0.23
Crab larvae	+	-		+	+		+	0.22
Fish eggs	+	+		+	-		+	0.16
Estuarine fish larvae	-	+		+	+		-	0.14
Marine fish larvae	+	+		+	+		-	0.19
Marine sciaenid larvae	+	-		-	-		+	0.38

species and life stages. With the same method used for biological groups and the data from Table 6.1.5, tidal height was also a statistically significant factor ($P < 0.05$).

Discussion

Limitations due to high variance. Analytical models associated with biological and physical systems are subject to a high level of variance due to the inherent nature of the species themselves and the physical dynamics of the environment. For example, the exact spawning sites of many species are not known, but some spawn in the estuary, some in the Gulf, and some may spawn in both areas. Shelf spawners may release the eggs nearshore or at mid-shelf where they are subject to a variety of environmental influences. This will result in differences in condition and age of larvae entering the passes. The breeding season may peak quickly in some species, while it may extend over a longer period of time and exhibit more than one peak in other species. Additionally, channel differences modify environmental influences on organisms (e.g., differences in physical setting, channel orientation, depth, cross-sectional area, relationships with adjacent water bodies, disturbances by boat and ship traffic, and reactions to chemical pollutants). These differences were inherent in the three channel sites selected. As a result, the eggs, larvae, and postlarvae of a given species may display different behavioral patterns and respond differently to the collecting gear. Therefore, Darnell and McEachron (1989) lumped samples to increase sample sizes and control the variance.

Although a high level of variance in the sample data occurred, it has been possible to reach some conclusions concerning the mechanisms even though specific causal relationships have not been established. On the basis of lumped and individual species relationships, the orientation of the channel with respect to wind and current, and possibly tidal elevation, influence larval transport through the passes.

Effects of freshwater inflow on fish transport within the estuary. While current direction, wind direction, and tidal height are indicated as mechanisms influencing egg and larval transport through passes, behavioral or physiological mechanisms related to salinity (and therefore indirectly to inflow) can play a role in transport within the estuaries. For example, the fertilized eggs of the Atlantic croaker (*Micropogonias undulatus*), spotted seatrout (*Cynoscion nebulosus*), and red drum (*Sciaenops ocellatus*) are buoyant over a narrow range of salinities (Thomas and Boyd 1989). Since buoyancy influences egg movement by estuarine water currents, the transport of buoyant eggs is related to the salinity gradient and the currents of the bay water to which they are exposed. Salinity extremes may impair the egg buoyancy function and the growth rate of developing larvae of these species (Holt and Banks 1989). Other species with non-buoyant, passively transported eggs tend to have short hatching times (two to three days) (Jones et al. 1978; Wang and Kernehan 1979). Species with buoyant eggs often have longer hatching times and can be transported farther into the estuary before hatching occurs; salinity, which is influenced by inflow, maintains the buoyant relationship. Some fish with demersal eggs are known to migrate to the more constant salinity conditions in the region of estuary mouths to breed (Jones 1962; Wang and Kerneham 1979; P. R. Dando 1984). Since their eggs are demersal, freshwater inflow has little influence on egg transport. These species may depend on

Table 6.1.5. Summary of signs of coefficients of multiple regression equations relating abundance to all environmental variables measured at the Matagorda Ship Channel (7 variables), Saluria Bayou (6 variables), and Gulf Intracoastal Waterway (5 variables). Data for some stages or species was not available for all sampling locations. Multiple regressions were derived by omitting zero values. Abbreviated information is as follows: C = channel current, D = water column segregation by depth, L = light conditions, S = salinity gradient, T = temperature gradient, TH = tidal height, W = channel wind, ns = not significant, 0.05 = significance level ($P < 0.05$).

Stage or species	C	D	L	S	T	TH	W	Average R^2
Shrimp—protozoea	+++	+--	+	-++	++-	+-	++-	0.57
Shrimp—mysis	+-+	+++	+	---	++-	++	+++	0.56
Brown shrimp—post-larvae	---	---	+	--+	--+	++	-+-	0.32
Shrimp—post-larvae	+-+	+-+	-	-+-	++-	-+	++-	0.12
Crab—zoea	+++	---	+	-++	+--	--	+++	0.28
Blue crab—megalops	+-+	+-+	+	++-	---	++	---	0.09
Crab—juveniles	+	+	-	-	-	+	+	0.11
Silver perch	+-	+-	+	++	-+	+	+-	0.60
Sand seatrout	++	++	-	--	-+	+	++	-0.08
Spotted seatrout	++-	++-	+	--+	--+	+-	-+-	0.39
Black drum	-+	-+	-	--	--	++	++	0.22
Star drum	-	+	-	-	-	+	+	0.25
Total sign distribution	19+/10-	16+/13-	8+/4-	11+/18-	11+/18-	15+/5-	19+/10-	
Number of observations	29	29	12	29	29	20	29	
Percent positive signs	65.5%	50.0%	66.7%	37.9%	37.9%	75.0%	65.5%	
Statistical significance	ns	ns	ns	ns	ns	0.05	ns	

behavioral adaptations to ensure survival of their young such as guarding their nests or fanning their eggs, and not wide distribution of their eggs and larvae by currents.

Darnell and McEachran (1989) did not provide evidence supporting the existence of a relationship between freshwater inflow or salinity and transport of fish eggs or larvae through the passes from the Gulf to the estuaries. Nevertheless, the relationship between fish egg buoyancy and salinity and the preference of fish eggs and larvae for particular salinity regimes is sufficient to justify concern about maintenance of salinity gradients, even if transport from the Gulf is not a factor.

Effects of inflow or salinity on shrimp movement. Shrimp exhibit different degrees of preference to salinity at different stages of their life cycle. Hughes (1969) indicated that tidal transport of postlarvae may be initiated by increases in salinities of flood tides into low salinity estuaries where juveniles prefer to grow, until they finally leave the bays and enter the open sea (Williams 1955).

The eggs of brown shrimp are semibuoyant, while those of the white and pink shrimp are demersal (Kutkuhn 1966a, b; Anderson 1966; Ewald 1965). The transport of eggs released into the water column in the Gulf vary as a result of differences in buoyancy. Since salinity affects egg buoyancy in the water column, it may influence egg transport in the Gulf by currents, tides, and other physical factors. Within 24 hours, the eggs hatch into nonfeeding planktonic nauplii, which are carried by prevailing currents while they undergo several molts over the next 24 to 36 hours to become free-feeding larval protozoea. Jones et al. (1970) and Kennedy and Barber (1981) reported that larvae may use tidal currents to enter estuarine nursery grounds. Postlarvae also enter estuarine and coastal bay nursery areas (Copeland and Truitt 1966), their abundance increasing with increasing velocity of flood tides (Tabb et al. 1962).

In Texas, shrimp postlarvae enter estuarine nursery areas from February until November, with peaks varying by species (Klima et al. 1982). Christmas and Etzold (1977) reported that major alterations or losses of estuarine shrimp nursery habitat have resulted in Texas from dredging, spoil disposal, and impoundments which alter circulation patterns, habitats, or timing and quantity of freshwater inflow. LaFleur (1968) found that increased salinities have adversely affected white shrimp nursery grounds. Christmas and Etzold (1977) also suggested that the increases in salinity have caused shifts in dominance from white shrimp to brown shrimp production.

Emigration of juvenile and adult shrimp from estuaries appears to be governed by size of the shrimp and the environmental conditions within the estuarine system (Klima et al. 1982). In Texas coastal waters, a positive relationship between white shrimp production and increased rainfall has been attributed to a sharp increase in low salinity nursery areas. Annual white shrimp catches from 1927 to 1964 in waters off Texas showed a strong statistical correlation (R = 0.656) with rainfall in both of the preceding years (Gunter and Edwards 1969). A highly significant correlation (R = 0.85) between May to June freshwater inflow and white shrimp catches and commercial landings was demonstrated by Williamson (1977) in 1959 to 1975 in San Antonio Bay, Texas. The Texas Water Development Board (1982) reported 14 significant multiple regression equations explaining an average 69% of the variance for seasonal relationships of freshwater inflow to the commercial harvest of white shrimp on seven Texas estuaries and the Gulf coast during the period 1959-1976.

These studies show that the role of freshwater inflow in creating favorable habitat within the bays and estuaries is complex. If there is a role for freshwater inflow in transporting larvae from the Gulf into the bays and throughout the estuaries, it is probably not a direct one.

Possible indirect roles through freshwater effects on circulation processes within the bays or through modification of larval behavior patterns have not been investigated. Coastal invertebrates and fish display a great diversity of spawning seasons, selection of spawning grounds, and distributional relationships with depth, temperature, salinity, and light conditions. However, the major "bottleneck" for all the estuary-related species is the problem of traversing the passes, and with regard to life history problems, Darnell and McEachran (1989) observed a commonality in adaptations of the various species with respect to their involvement with upchannel current and upchannel wind in moving the larvae through the tidal passes. In addition, the larvae may not be entirely passive. Behavior may play a significant role particularly among the older larvae and the juvenile stages, in reaction to other environmental variables.

Conclusion

Current direction, wind direction, and tidal height are the dominant factors affecting larval transport through the passes and channels. The role of salinity and freshwater inflow in transporting eggs and larvae through these areas has not been demonstrated. However, physiological and behavioral mechanisms related to salinity and (indirectly) freshwater inflow have been reported to affect the transport of these life stages within the bay.

6.2 EFFECT OF FRESHWATER INFLOW OR SALINITY ON THE ABUNDANCE AND DISTRIBUTION OF JUVENILE FISH AND SHELLFISH

Estuaries as Nurseries

Texas estuaries change rapidly in response to changes in freshwater inflow, tidal currents, and atmospheric conditions due in part to their shallow water depths. Despite these changing conditions, flora and fauna in these estuaries are productive and show rich species diversity (Flint 1985).

Biological production in Texas estuaries has been related to rainfall and freshwater inflows (Gunter and Hildebrand 1954), but the functional relationship between rainfall and estuarine production is currently not fully understood. Estuarine organisms are adapted to a wide range of environmental conditions with respect to temperature and salinity. Laboratory experiments suggest that animals grow best within a narrow range of environmental conditions, but the evidence in nature is that estuarine organisms are commonly found in a wide range of salinities and temperatures. They shun areas only when salinities or temperatures are very high or very low.

Salinity has been reported to affect metabolism, activity, and the endocrine system of estuarine organisms (Holliday 1972). The classic study by Bull (1938) demonstrated that some fish could discriminate among salinities differing by about 0.5‰. The goby (*Gobius flavescens*) was even able to discriminate salinities of 0.06‰. Responses to salinity level or changes in salinity differ at different life stages, and are often based on different mechanisms. In general, most estuarine species spawn in the Gulf of Mexico or in bay-Gulf passes where salinity concentrations remain near seawater levels (greater than 34‰). Larvae then move into estuarine habitats to grow and seek refuge from marine predators. This means that most estuarine organisms start life in high salinities and relatively stable temperatures. While moving into the estuarine nursery area, they develop the ability to tolerate lower salinities and to cope with more rapid changes in salinity and temperature.

This section discusses the relative abundance of selected juvenile fish and invertebrates in Texas estuaries, and the response of those juvenile organisms to salinity variations. Data from the Guadalupe Estuary may illustrate patterns common in other Texas estuaries.

Blue Crab (*Callinectes sapidus*)

Blue crabs are distributed from Nova Scotia to northern Argentina, including Bermuda and the Antilles (Williams 1974). Along the Gulf and Atlantic coasts of the United States, they occur in almost every estuary, including all Texas estuaries.

Life Cycle. Blue crabs occur in different habitats within the estuary, depending on their life stages, sex, and spawning status. Adult females mate in the estuary and then migrate out of the estuary into the Gulf of Mexico to spawn. Eggs hatch and develop into zoeal stages in offshore waters. Recruitment of larvae back into bays and estuaries occurs during the megalopal stage. The mechanisms of larval transport of blue crab in Texas coastal waters are not well defined, but are believed to be related to wind direction and tidal currents.

Eggs and larvae. Blue crabs produce about two million eggs per spawning (Churchill 1921) and eggs hatch within 14 to 17 days following fertilization at 26 °C (79 °F). Relatively high salinities (23 to 30‰) are required for successful hatching. Costlow and Bookhout (1959) reported that in salinities lower than 15‰, no hatching resulted. Newly hatched blue crabs normally develop through seven zoeal stages before metamorphosing into megalopae. Mortality is usually high for the first two zoeal stages. The maximum survival rate for crab reared from hatched zoea to first crab stage in the laboratory was reported to be 40% (Millikin and Williams 1984). According to Sulkin and Epifanio (1975) and Bookhout et al. (1976), the optimum salinity and temperature combination for zoeal development is 25 °C (77 °F) and 30‰ salinity. Metamorphosis through the seven zoeal stages takes 31 to 49 days at 25 °C (77 °F) and 26‰. Optimum salinity and temperature requirements remain the same during the development of megalopae. The average metamorphosis time was 8.4 days, with a range of 6 to 12 days (Costlow 1967).

Juveniles. Laboratory studies have shown that young crabs are tolerant to a wider range of both temperature and salinity compared with their larvae. Holland et al. (1971) reported that salinity ranging from 6 to 21‰ did not affect growth and food conversion of laboratory-reared juvenile crabs (6 to 38 mm or 0.25 to 1.5 inches carapace width). Salinity less than 1‰ caused high mortality at 29 °C (84 °F) but not at 15 °C (59 °F). This suggests that extremely high or low temperatures significantly affect the blue crab's ability to tolerate salinities outside of its preferred range.

Juvenile crab sampled from Galveston Bay showed maximum weight gain at 30 °C (86 °F) in the laboratory over a period of 45 days (Holland et al. 1971). However, when temperature was higher than 30 °C (86 °F), crab growth declined rapidly. Crab ceased molting when the temperature dropped below 16 °C (61 °F) in Chesapeake Bay (Graham and Beaven 1942).

Based on laboratory studies (Costlow and Bookhout 1959), blue crabs underwent eight larval molts (seven zoeal stages and one megalopal stage), followed by 18 to 20 postlarval molts for females and 21 to 23 molts for males (Van Engel 1958). Intermolt periods of blue crabs increase with decreasing temperature, increasing salinity, and age; but they are generally shortened by ample food, near-mesohaline waters (10 to 20‰), and high water temperature. Growth per molt frequently decreased as intermolt period decreased (Leffler 1972). Tagatz (1968), however, reported a different growth pattern in crab from the St. Johns River, Florida, where crab growth per molt was constant regardless of temperature (summer vs. winter).

Studies were recently conducted to determine the importance of submerged aquatic vegetation to juvenile crab and shrimp in the Guadalupe Estuary (Academy of Natural Sciences 1989). Blue crabs were most abundant and closely associated with vegetated habitats. Low abundance or absence of crabs near delta sites was ascribed to oligohaline or freshwater conditions. Mean salinities at the delta stations (Lucas Lake) ranged from 0.25 to 1.25‰ during the period of fall 1986 through fall 1987, when inflow was high. Similar results were also observed in Galveston Bay (Zimmerman et al. 1990b). In San Antonio Bay, TPWD trawl samples show significantly higher densities (catch/sample) in areas with mean salinities of less than 20‰ or in the region most influenced by Guadalupe River inflows (Figures 6.2.1, and 6.2.2). A regression analysis using catches (Log (catch +1)) for all years 1982 through 1988 as the dependent variable, and salinity measured at the same site and time, as the independent variable, was highly significant ($P < 0.0001$; $R^2 = 0.42$). This relationship was negative, showing higher catches from lower salinity areas.

Dummy variables were used in the analysis to adjust for different densities which occurred in different years and different months.

Oyster (*Crassostrea virginica*)

Three species of oysters are commercially utilized in the United States. The eastern oyster, *C. virginica*, is the most abundant and is mainly found in brackish waters of the bays and inlets along the Atlantic and Gulf coasts (Stanley and Sellers 1986). In the Gulf of Mexico, eastern oyster production is led by Louisiana accounting for 65% of the harvest, followed by Texas at 8% (TPWD 1988). While they occur in all Texas bays, their abundance varies among bays. Approximately 90% of reported commercial Texas landings are harvested directly from reefs in the Galveston, Matagorda, and San Antonio Bay systems.

Life cycle. Oysters are sessile bivalve mollusks. Unlike most other organisms, the oyster spends its entire life in the estuary. Oysters in Texas spawn year-round, with peak spawning recorded in June and July after temperatures have reached and stayed higher than 20 °C (68 °F) and salinities stay above 10‰ (Hofstetter 1977, 1983).

Eggs and larvae. The earliest part of the oyster's life is spent as free-floating zooplankton larvae. Later, they become sessile organisms when they attach to hard surfaces and, under the right conditions, to each other, forming large reefs. The hatched larvae undergo several developmental stages including trochophore, veliger, and spat before they settle down and attach to substrate for benthic living. The period from hatching to settling is usually seven to ten days depending on factors such as temperature, salinity, and substrate (Loosanoff 1953). Hopkins (1931) reported a salinity of 20 to 21‰ to be a critical level for oyster settling. Data from 23 years of reef sampling in Galveston Bay suggests that the best spat setting occurs when spring salinities reached 17 to 24‰. The poorest sets were observed when salinities dropped below 8‰ (Hofstetter 1983). In Louisiana, setting intensity was high when salinities were 16 to 22‰, with the peak occurring between 20 to 22‰ (Chatry et al. 1983).

Salinity also affects the temperature tolerance of oyster larvae. At the optimum salinity, larvae can survive a wide range of temperatures. Survival rates of 70% or more have been reported at salinities ranging from 10 to 27.5‰ and at temperatures from 27.5 to 32.0 °C (82 to 90 °F) (Davis and Calabrese 1964). Under laboratory conditions, larvae held at 30 °C (86 °F) begin setting 10 to 12 days after fertilization, while those held at 24 °C (75 °F) set after 24 to 26 days. Few larvae held at 20 °C (68 °F) set within 35 days (Loosanoff and Davis 1963).

Oysters require the correct substrate to successfully settle. According to Loosanoff (1953) and Menzel (1955), oysters in salinities greater than 20‰ tend to attach to substrate in the intertidal zone, while those in salinities less than 20‰ set subtidally. In the laboratory, Hidu and Haskin (1971) showed that the presence of spat on cultch shell attracted other larvae and stimulated them to set. Field observations also indicated that an existing reef provided the best and most attractive place for setting. Soft sand or shifting sand and mud were unsuitable substrate for oyster setting (Galtsoff 1964).

Oyster larvae growth rates are initially high during the first six months after setting and thereafter gradually decrease throughout the life stages (Heffernan 1962). Minimum temperature for growth of oyster larvae was 17.5 °C (64 °F) and optimum salinities for spat growth were 15 to 22‰ (Chanley 1957). Loosanoff and Nomejko (1949) reported that oysters in Milford Harbor, Connecticut, did not increase in size or biomass during winter. The estimated growth rate for Texas oysters between 2 to 48 months of age ranged from 2.6 to 10 mm (0.1 to 0.4 inch) per month (TPWD 1988).

Juveniles. Various environmental factors affect juvenile oyster survival and spatial distribution in estuarine environments. Optimum temperature and salinity for survival were 20 to 30 °C (68 to 86 °F) and 10 to 30‰ respectively (Gunter and Geyer 1955). Oysters were capable of surviving salinities ranging from 3 to 44‰ (Copeland and Hoese 1966). High salinity was suggested to be the factor limiting the abundance of oysters on the lower Texas coast, where salinities in excess of 45‰ were common (TPWD 1988). On the other hand, low salinities (less than 3‰) also affect oysters feeding and increase mortality. For example, oysters acclimated to 27‰ showed no feeding in salinity less than 3‰ (Loosanoff 1953). Oysters in Trinity Bay, Texas have survived salinities of 5‰ at temperatures ranging from 24 to 27 °C (75 to 81 °F) for two to three weeks, but they experienced high mortalities (greater than 90%) when exposed to water at the same temperature with salinities of less than 2‰ for three weeks (Benefield 1966).

Oysters demonstrated an inverse relationship with salinity and temperature. Oysters can tolerate low salinities at temperatures lower than 5 °C (41 °F) for relatively long periods but can survive only a few days under the same conditions at 15 °C (59 °F) (Andrews 1982). Gulf coast oysters generally survived freezing water temperatures (Cake 1983). Obviously, this can be affected by very low salinities.

Other factors which affect juvenile oyster survival and growth include food concentration, dissolved oxygen, and water flow rates. The latter factor is important because water

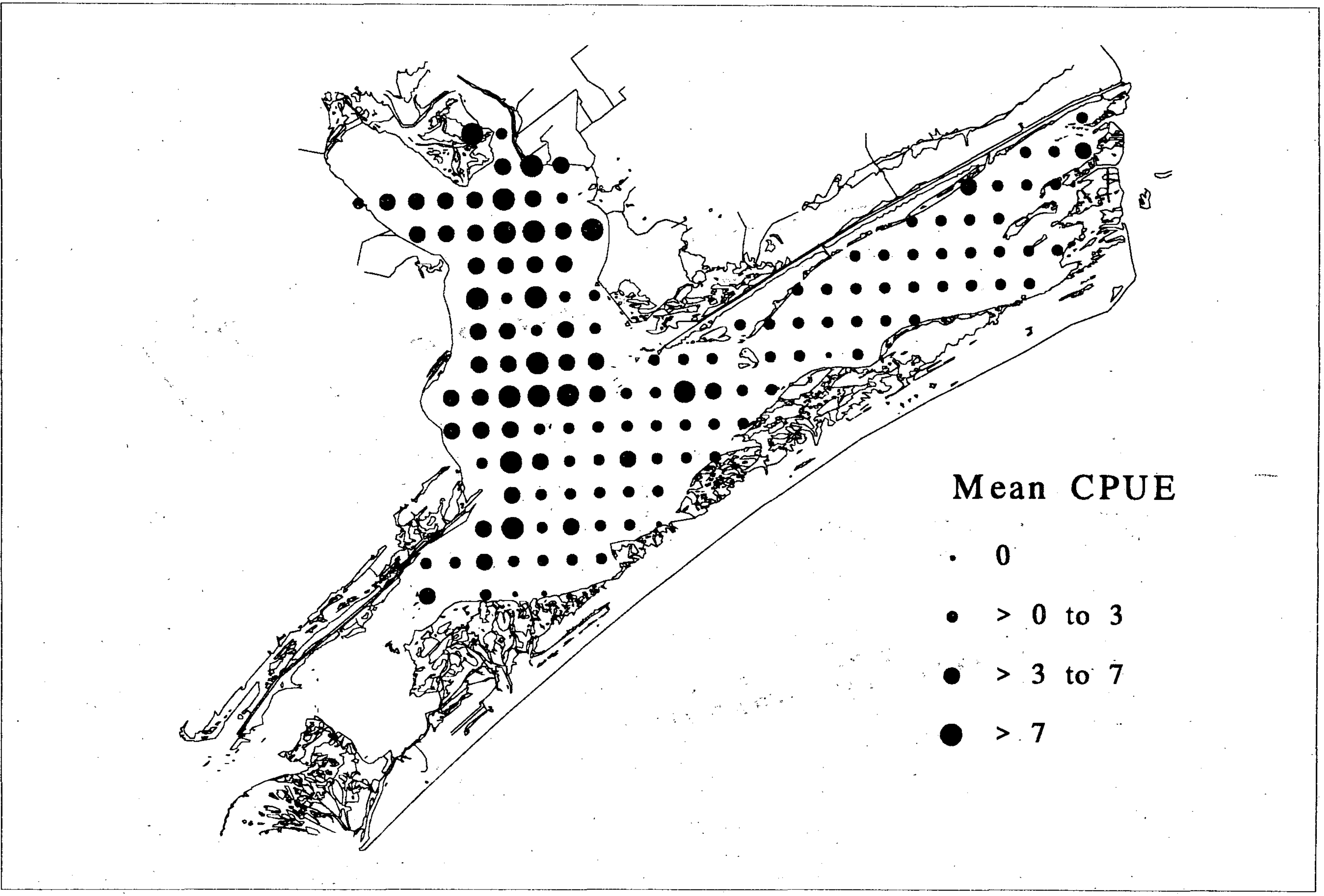


Figure 6.2.1. Relative abundance of juvenile blue crab (*Callinectes sapidus*) in the Guadalupe Estuary, from trawl samples.

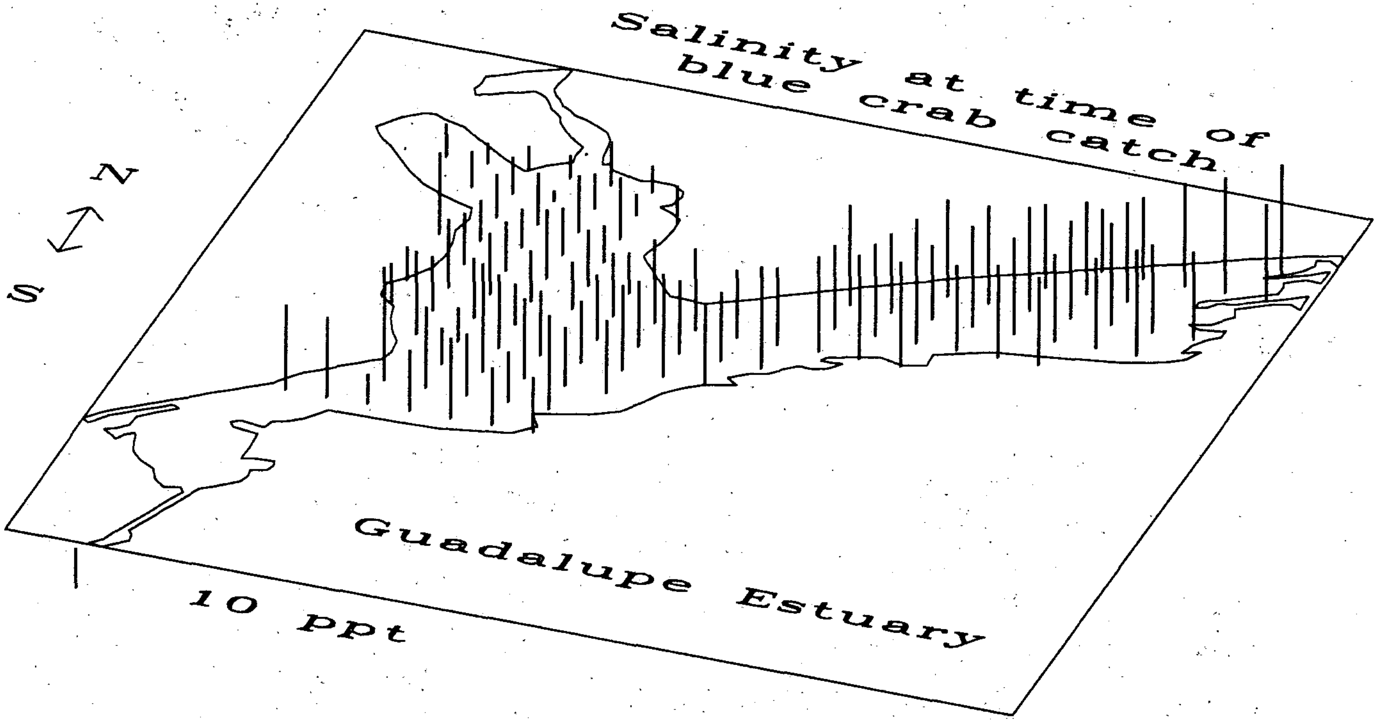


Figure 6.2.2. Salinity distribution at the time of blue crab (*Callinectes sapidus*) capture in the Guadalupe Estuary, from trawl samples.

current not only provides food and oxygen, but also dissipates waste and disperses larvae to colonize new areas. Oysters in Delaware Bay were found most abundant in regions of high water velocity (Keck et al. 1973). However, extremely high velocities (above 150 mm/s or 6 inches/s) may cause loosely attached oysters to loosen and be killed by tumbling along the bottom (Mackenzie 1981).

Oysters depend on changing salinities to control predators. Oyster predators, including fish, crustaceans, sponges, and other mollusks, are usually plentiful in oyster reefs (Zimmerman et al. 1990b). Most oyster predators are favored when salinities are high for prolonged periods. When conditions remain favorable for predators, they can proliferate enough to significantly increase oyster mortality. Annual oyster mortality due to oyster drills (*Thais haemostoma*) ranged from 50 to 85% in Louisiana (Schlesselman 1955), and 50 to 100% in Mississippi (Chapman 1959). In the laboratory, small drill (50 mm or 2 inches) were able to consume about 85 spat per day, and one to five market-size oysters (greater than 75 mm or 3 inches) per month. Fortunately, oyster drills and other predators are reduced by low salinities. Salinities of less than 15‰ can benefit oysters by reducing their predators' abundance (Menzel 1966). Therefore, periodic freshwater flooding is needed to control oyster predators. Pierce and Conover (1954) also reported that oysters exposed to fluctuating salinity in the normal range (10 to 30‰) grew faster than those held at a relatively constant salinity. The largest and most numerous oyster reefs in the Guadalupe Estuary are located in areas having salinities commonly ranging from 10 to 24‰ (Diener 1975).

White Shrimp (*Penaeus setiferus*)

White shrimp are widely distributed along the North American East Coast. They are found in estuaries from Long Island Sound to Campeche, Mexico (Perez-Farfante 1988). In Texas, white shrimp are found in estuaries where temperatures range from 5.2 to 39 °C (41 to 102 °F) and salinities from 0.2 to 45.3‰ (Copeland and Bechtel 1974). They are most abundant in estuaries having mean salinities of less than 22‰.

Life cycle. Adult white shrimp generally inhabit nearshore waters of the Gulf about 14 m (45 ft) in depth and with salinities greater than 27‰ (Cook and Murphy 1969). Eggs are demersal and larvae are planktonic. White shrimp enter estuaries during the second postlarval stage at about 7 mm (0.28 inch) and thereafter begin a benthic existence in shallow waters with soft bottoms and marsh grasses. The time between hatching and migration to estuaries is estimated to be two to three weeks. As shrimp grow, they leave marsh habitat for deeper portions of the estuary prior to their journey back to the Gulf waters.

Eggs and larvae. White shrimp spawn from late spring to early fall. Peak spawnings are usually observed in June and July. From 0.5 to 1 million eggs are produced per spawn. Eggs usually hatch within 10 to 12 h. The larvae then metamorphose through five nauplius, three protozoea, and three mysis stages before they become postlarvae. It takes two to four weeks to metamorphose from newly hatched nauplii to the first postlarvae, depending on temperature, salinity, and food availability. In the Trinity-San Jacinto Estuary, white shrimp postlarvae are first caught during May in the Galveston Channel, and peak during early summer (Klima et al. 1982).

Juveniles. Juvenile white shrimp tolerate a wide range of temperatures, but growth is slow when water temperature is less than 20 °C (68 °F). Zein-Eldin (1965) reported that postlarvae held in the laboratory could attain maximum growth rates of about 1.4 mm (0.06 inch) per day at 32 °C (90 °F), 1.1 mm (0.05 inch) per day at 25 °C (77 °F), and negligible at 11 °C (52 °F).

Juvenile white shrimp use a wide variety of available estuarine habitats. They are caught during all months of the year, but are most abundant from July through December (Mueller and Matthews 1987). In the field, white shrimp catch ratios were found to be positively related to temperature up to 40 °C (104 °F) (Copeland and Bechtel 1974). Catch ratios here refer to the number of successful catches divided by the number of attempts and is an index of percent successful catch. Commercial catch data suggests that white shrimp are most abundant in low-salinity waters. Gunter et al. (1964) summarized data collected from Texas waters and concluded that white shrimp occurred most abundantly in areas with salinities of less than 10‰. These investigators suggested that salinity was a limiting factor to the distribution and abundance of shrimp in the coastal waters. However, laboratory studies also suggested that white shrimp could tolerate much higher salinities because they could be successfully raised at salinities of 8 through 34‰ (Perez-Farfante 1969). Copeland and Bechtel (1974) also indicated that no apparent relationship existed between catch success and salinity. White shrimp appeared to occur over the entire range of estuarine salinity. Parker (1970) even stated that high catches of shrimp at low salinities might result because low salinities occurred during the period of peak abundance and in areas where shrimp were abundant (i.e., marshes and shore areas).

TPWD bag seine monitoring data was analyzed to determine if the spatial distribution of white shrimp in the San Antonio Bay system were affected by salinity and substrate type at the sampling site (Lee et al. 1990). This analysis was conducted with the assumption that shrimp generally remained in areas where conditions were most favorable for their survival and growth, and that higher catch rates would be associated with favorable areas. Estuarine areas were characterized according to observed salinity levels and the data was analyzed using multiple regression to determine which salinity zones resulted in the highest catch rates. Dummy variables were used to reduce variations caused by different abundances in different years and different months. Small white shrimp (less than 50 mm or 2 inches) showed no differences in catch per sample among different salinity zones (Figure 6.2.3). Larger white shrimp (greater than 50 mm or 2 inches) showed higher catch per sample in less saline areas. A posteriori comparison of mean catches indicated that catches from high salinity zones (24.2 to 26.2‰) were significantly lower ($P < 0.05$) than those of the lower salinity zones. Mean trawl catches plotted by area in the Guadalupe Estuary also showed a clear pattern of higher catches occurring in areas with lower mean salinities (1 to 21‰) (Figures 6.2.4 and 6.2.5). A significant inverse relationship between salinities and trawl catches was detected ($P < 0.0001$, $R^2 = 0.29$). Again, dummy variables were used to adjust for different densities which occurred in different years and different months.

Gunter and Hildebrand (1954) were the first to report a positive relationship between rainfall and white shrimp production in Texas. Gunter and Edwards (1969) also found white shrimp abundance to be positively correlated with the previous two years' rainfall. Mueller and Matthews (1987) examined this relationship, although their results were not consistent with the earlier studies. They found poor or no correlation between catches and annual inflow. However, they did find significant correlation between shrimp catch and selected monthly river flow. They hypothesized that high spring inflows loaded the system with nutrients and detritus, thus stimulating a spring-early summer plankton bloom which benefited newly arrived postlarval shrimp. The reason for a positive influence of increased flows in October is less obvious. Mueller and Matthews, however, speculated that it might act as a flushing mechanism to move shrimp offshore. If true, inflows may not increase production, but may merely move shrimp to areas where they are more available to fishermen.

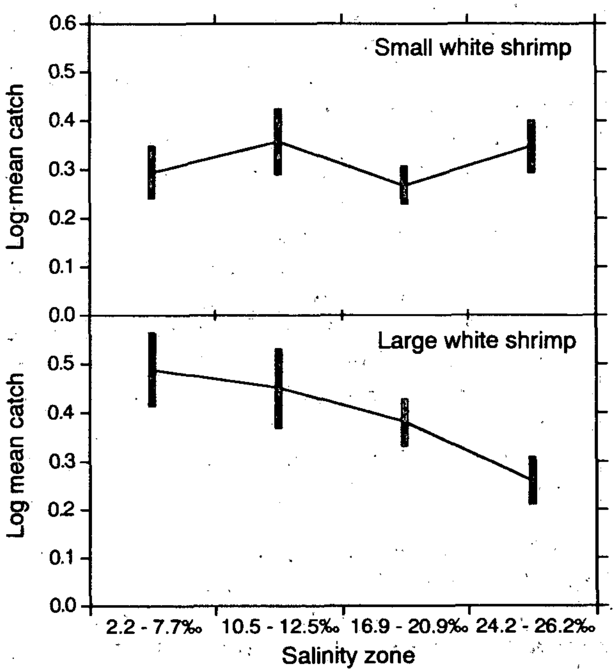


Figure 6.2.3. Relative abundance of juvenile white shrimp (*Penaeus setiferus*) along salinity gradients in the Guadalupe Estuary, from bag seine

Zein-Eldin and Renaud (1986) suggested that salinity-temperature interactions may have more pronounced effects on shrimp than either factor alone, and believed that low salinity-low temperature combinations were the most detrimental to white shrimp. Laney (1971) reported that a critical thermal maximum for white shrimp was influenced by life stage and acclimation to temperature, but was less dependent on salinity. The reported 80% survival curve for white shrimp, over the temperature and salinity ranges possibly encountered in estuaries, was not well defined, but increased survival was noted in low salinity water when the temperature was above 25 °C (77 °F) (Copeland and Bechtel 1974). The latter is a common condition in the Guadalupe Estuary when white shrimp are present.

Brown Shrimp (*Penaeus aztecus*)

Brown shrimp range from Martha's Vineyard, along the entire East Coast, through the Gulf of Mexico, down to the Yucatan Peninsula, Mexico. Brown shrimp have population centers along the Texas-Louisiana coast (Perez-Farfante 1988). They are the most important commercial shrimp on the Gulf coast and are found in all Texas bays. Unlike white shrimp, brown shrimp are abundant in lower coastal bays in Texas where there is less rain and higher salinities (Meador et al. 1988).

Life cycle. Brown shrimp have a life cycle similar to white shrimp, and nursery grounds of both species overlap within the bays. Adult brown shrimp spawn from September through May of the following year in offshore waters ranging from 14 to 110 m (44 to 341 ft) (Renfro and Brusher 1982). As eggs hatch and develop into postlarvae, they move into estuaries on incoming tides as plankton and become demersal. Transformation to juveniles occurs in estuaries when the postlarvae are approximately 25 mm or 1 inch in total length (TL) (Look and Lindner 1970). Juvenile brown shrimp remain at the water-marsh interface or in seagrass beds for two to three months. At a length of 60 to 70 mm (2 to 2.5 inches), they move into deeper open waters and normally begin their seaward migration when they reach lengths of 80 to 100 mm TL (2.5 to 4 inches) (Van Lopik et al. 1979). Emigration to the Gulf appears to be a direct route

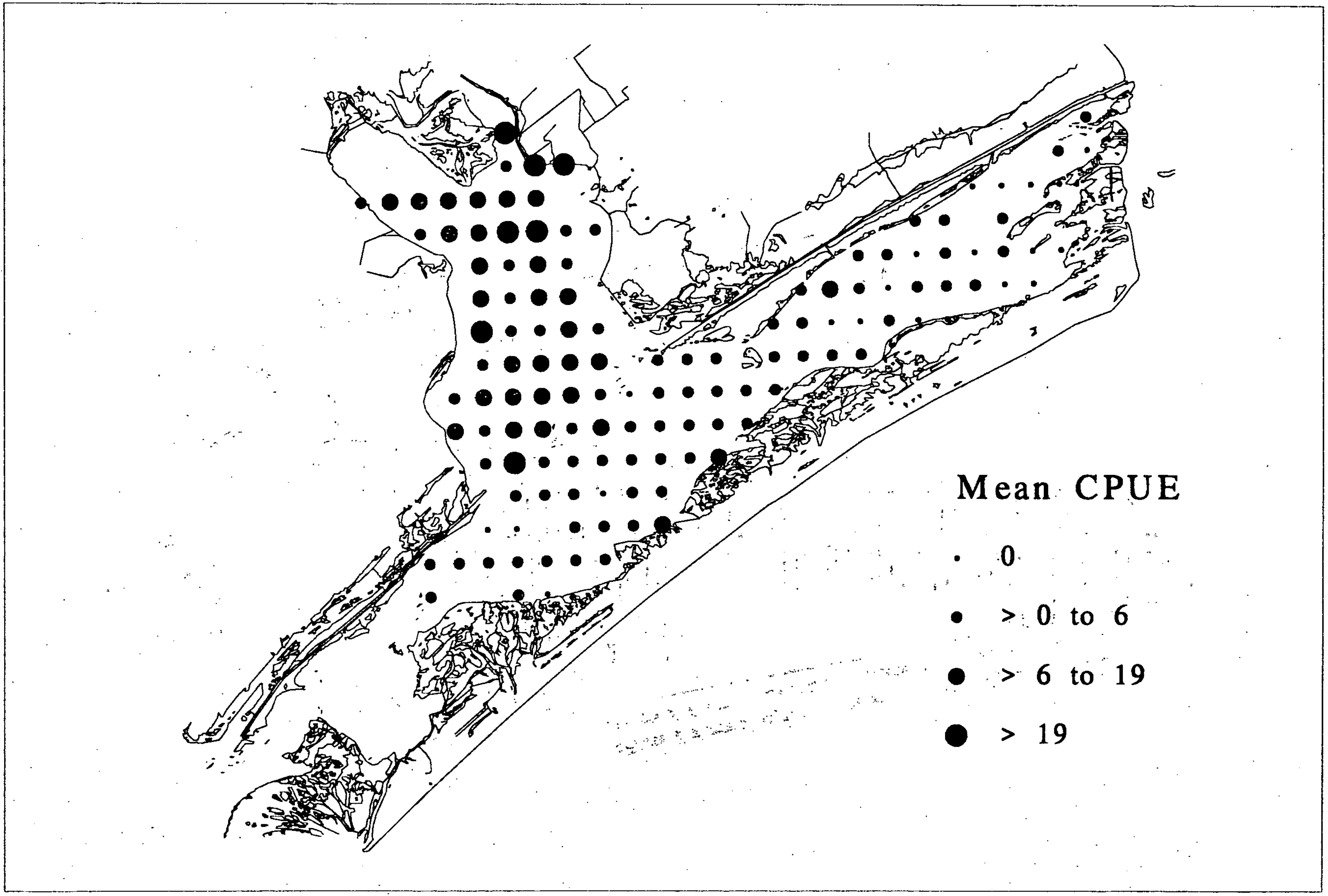


Figure 6.2.4. Relative abundance of juvenile white shrimp (*Penaeus setiferus*) in the Guadalupe Estuary, from trawl samples.

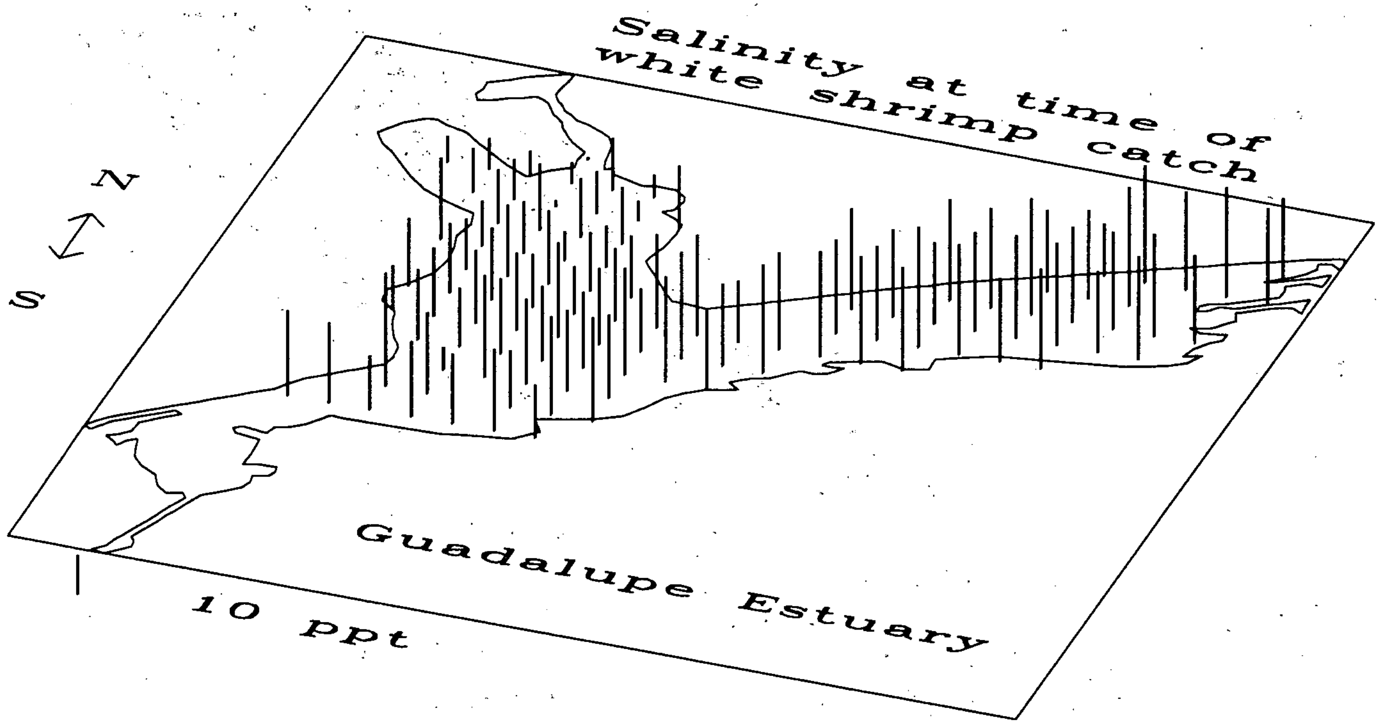


Figure 6.2.5. Salinity distribution at the time of white shrimp (*Penaeus setiferus*) capture in the Guadalupe Estuary, from trawl samples.

to passes and not a random search. The movement usually begins in late May, peaks in June, and is completed by July (Parker 1970). Brown shrimp mature in neritic waters when they are about 140 mm (5.5 inches) TL.

Eggs and larvae. Each gravid female releases about 250,000 eggs. Eggs hatch within 24 hours and then pass through nauplius, protozoea, and mysis stages before developing into postlarvae. Shrimp growth rates change with life stages, temperature, and food availability (Perez-Farfante 1969). Growth rates are low during larval stages and become significantly greater during postlarvae and juvenile stages. Postlarvae growth reached a maximum between 25 and 27 °C (77 and 81 °F), and growth ceased when temperatures were higher than 30 °C (86 °F) (Zein-Eldin and Renaud 1986). In general, mean growth rates of postlarvae and juveniles in the laboratory were less than 1 mm (0.04 inch) per day compared with 1 to 1.5 mm (0.04 to 0.06 inches) per day observed in the field during the primary growing seasons of late spring and early summer (St. Amant et al. 1966).

Juveniles. Brown shrimp have been collected over a wide range of temperatures and salinities in estuaries. While white shrimp continued to grow and survive at a constant temperature of 35 °C (95 °F), survival of juvenile brown shrimp decreased above 30 °C (86 °F). Maximum growth, survival, and food utilization for brown shrimp were found at 26 °C (79 °F) (Venkataramiah et al. 1972). In the field, Copeland and Bechtel (1964) reported that juvenile catch ratios increased when temperatures were above 15 °C (59 °F), and decreased when temperatures were above 30 °C (86 °F). The minimum temperature for significant catches was found to be 15 to 20 °C (59 to 68 °F). At both extremes (less than 15 °C or 59 °F, or greater than 30 °C or 86 °F), no brown shrimp were taken. The overall catch ratio in Texas bays was 0.53. Compared with white shrimp, brown shrimp were more tolerant of temperatures below 15 °C (59 °F).

Lassuy (1983) reported that juvenile brown shrimp were euryhaline and had been collected in waters with salinities ranging from 0 to 45‰. In the field, they have been found most abundant in waters of 10 to 20‰ (Gunter et al. 1964). A tagging study by White and Boudreaux (1977) also showed that reduced salinities in marsh areas due to freshwater inflow would cause juveniles to migrate to deeper, more saline waters. Bag seine data collected by the TPWD during 1977-1987 also pointed to similar results; shrimp less than 51 mm (2 inches) TL preferred higher salinity zones in the estuaries (Lee et al. 1990). In San Antonio Bay, areas with a mean salinity range of 24.2 to 26.2‰ yielded more catches of small shrimp (less than 50 mm or 2 inches) than areas with lower mean salinities (Figure 6.2.6). Low catches were found to be associated with both low and variable intermediate salinities. However, there were no differences in mean catches of shrimp larger than 50 mm (2 inches) TL among different salinities. Holt and Arnold (1989) estimated brown shrimp density to be 20 to 30 per 10 m^2 in lower San Antonio Bay compared with a density of less than 5 per 10 m^2 in the upper bay. They concluded that salinity had a positive association with brown shrimp density. According to a report by the Academy of Natural Science (1989), upper San Antonio Bay became too fresh to be used by either brown shrimp or crabs during periods of high freshwater inflow. If true, the overall area available as nursery ground would diminish during periods of high inflow.

Copeland and Bechtel (1974) found no apparent relationship between brown shrimp catch ratio and salinity. The catch ratios were virtually the same for all salinity classes from 0 to 40‰ in estuaries, suggesting that salinity preference of brown shrimp extended over the entire range of estuarine salinities. Parker (1970) also showed that brown shrimp in Galveston Bay were most abundant in areas with salinity of about 5‰. The average relative density at sites with 0 to 4.9‰ salinities was about two times higher than at sites with 20 to 24.9‰ salinities. Some of the conflicting conclusions reached in these studies may be the result of not analyzing the data separately by size class.

Analysis of TPWD trawl data indicated that high mean catches were more common for brown shrimp than for white shrimp or blue crabs in the Guadalupe Estuary. The same data also pointed to a highly significant nonlinear

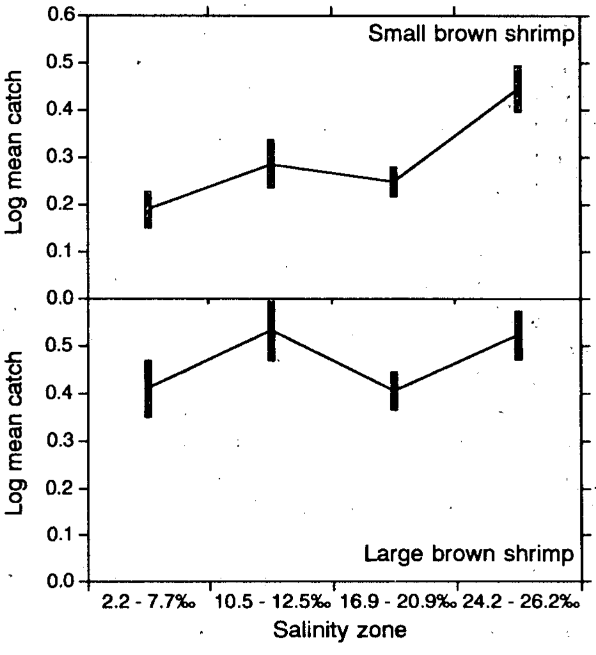


Figure 6.2.6. Relative abundance of juvenile brown shrimp *(Penaeus aztecus)* along salinity gradients in the Guadalupe Estuary, from bag seine samples.

relationship ($P < 0.0001$; $R^2 = 0.31$) between trawl catches and salinity. Brown shrimp catches were higher in the part of the estuary that had intermediate salinities of 15-25‰ (Figures 6.2.7 and 6.2.8). Mean lengths of these shrimp were 76 to 102 mm TL (3 to 4 inches) (Meador et al. 1988).

Brown shrimp tolerate a wide range of temperature and salinity combinations. While the optimum combinations for shrimp growth and survival were defined at 26 °C (79 °F) and 8.5‰ to 17‰, the worst conditions were at low temperature-low salinity extremes (Zein-Eldin and Renaud 1986). Venkataramiah et al. (1972) suggested that brown shrimp could tolerate a wider salinity range at 26 °C (79 °F) than at higher or lower temperatures. The range of temperature tolerance also increased with increasing salinity (Zein-Eldin and Aldrich 1965). Compared with white shrimp, brown shrimp survive better in low temperatures at 25‰ salinity (Lassuy 1983). Over the range of conditions normally encountered in estuaries, the 80% survival curve for brown shrimp encompassed a temperature range of 4 to 35 °C (39 to 95 °F) and a salinity range of 5 to 40‰, except in the combined low temperature-low salinity regime (Copeland and Bechtel 1974).

Juvenile brown shrimp in San Antonio Bay were mostly found on soft bottom, shallow water areas near or in marshes or seagrass beds (Zimmerman and Minello 1984; Academy of Natural Sciences 1989; Holt and Arnold 1989). Williams (1958) experimentally demonstrated in the laboratory a significant preference by settling postlarvae for soft, muddy substrate with decaying vegetation. In Louisiana, Turner (1977) found that the total shrimp yield was directly proportional to marsh areas and acres of seagrass. In San Antonio Bay, the soft bottom, on the average, produced 40% more catches than other sites with mud-shell or sand-shell-rock (Figure 6.2.9); the same held for white shrimp (Figure 6.2.10) (Lee et al. 1990).

Brown shrimp may be favored by variable seasonal freshwater inflows although the mechanism involved is not fully understood. Van Lopik et al. (1979) considered the relationship of brown shrimp harvest to temperature and salinity, and concluded that high brown shrimp landings could be expected if nursery areas had experienced a warm spring with relatively high salinities (about 20‰).

Red Drum (*Sciaenops ocellatus)*

Red drum have a wide distribution along the North American East Coast, ranging from the Gulf of Maine off Massachusetts to Florida and down to Tuxpan, Mexico. The species is one of the most highly prized food and recreational fish caught along the Gulf Coast.

Life cycle. Spawning adult red drum typically inhabit offshore waters and are only occasionally found in shallow bays. Annual spawns begin in late summer and end in early winter, with peaks recorded during September and October (Matlock 1987). Spawning generally occurs inshore on the oceanic side of barrier islands or shoreline, and is initiated after sunset (Holt et al. 1985). Eggs hatch in 19 to 20 hours at water temperatures of 24 to 28 °C (75 to 82 °F). The hatched larvae or postlarvae then ride tidal currents into shallow waters of the primary bay.

As they grow older, the postlarvae spread farther into secondary bays, and are found along with juveniles among patchy seagrass meadows and in water depths up to 3 m (10 ft). In later life stages, juvenile red drum tend to move into slightly deeper open waters. This movement can be accelerated by sudden drops in water temperature, which often occur during the fall and early winter. Subadults may remain in the bay for two to three years before they migrate offshore in late fall and winter to join the adult stock (Reagan 1985).

Eggs and larvae. Red drum eggs develop best in salinities greater than 25‰, while larvae (less than 8 mm or 0.3 inch) and postlarvae (8 to 15 mm or 0.31 to 0.59 inches) are stenothermal and are reported to be very sensitive to salinity changes. In Florida waters, they were collected from areas with temperatures ranging from 18.3 to 31.0 °C (65 to 88 °F) and in salinities ranging from 16 to 36.4‰ (Peters and McMichael 1987). Off the Alabama coast, larval red drum were caught in 27 to 29 °C (81 to 84 °F) and 18 to 35‰ Gulf waters (National Marine Fisheries Service 1986). Similar temperature-salinity ranges were reported for successful red drum hatching and development in the laboratory. Optimum combinations were 25 °C (77 °F), and 30‰, respectively (Holt et al. 1981). They also noted that at 25 °C (77 °F), some larvae were able to develop into feeding postlarvae at salinities ranging from 10 to 40‰. However, when salinities dropped to less than 20‰, eggs tended to sink, and losses from either bacterial infection or low oxygen level were high (Vetter et al. 1983). Conversely, they found high salinities along with high temperatures to be detrimental to yolk-sac larval survival.

Juveniles. Juvenile red drum (longer than 15 mm or 0.6 inch) are considered to be euryhaline and are collected in a wide variety of habitats ranging from seagrass meadow edges through open waters with slightly muddy bottoms to oyster reefs (Reagan 1985). Field sampling has shown juveniles in waters with temperatures of 2 to 34.9 °C (36 to 95 °F) and salinities of 0 to 45‰. They appear to prefer temperatures in the 10 to 30 °C (50 to 86 °F) range and salinities from 20 to 40‰ (Overstreet 1983). Simmons and Breuer (1962) reported the range of 30 to 35‰ as the

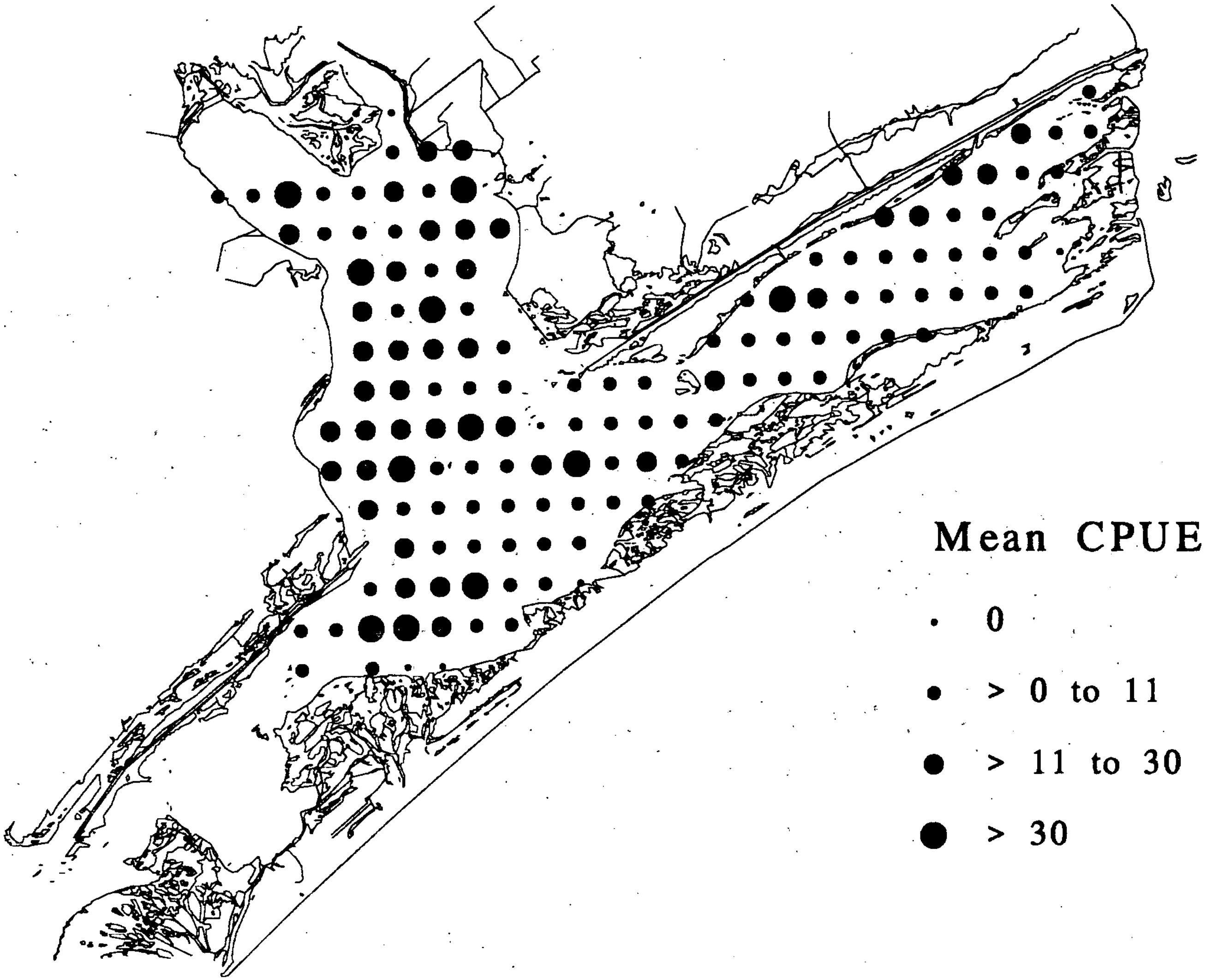


Figure 6.2.7. Relative abundance of juvenile brown shrimp (*Penaeus aztecus*) in the Guadalupe Estuary, from trawl samples.

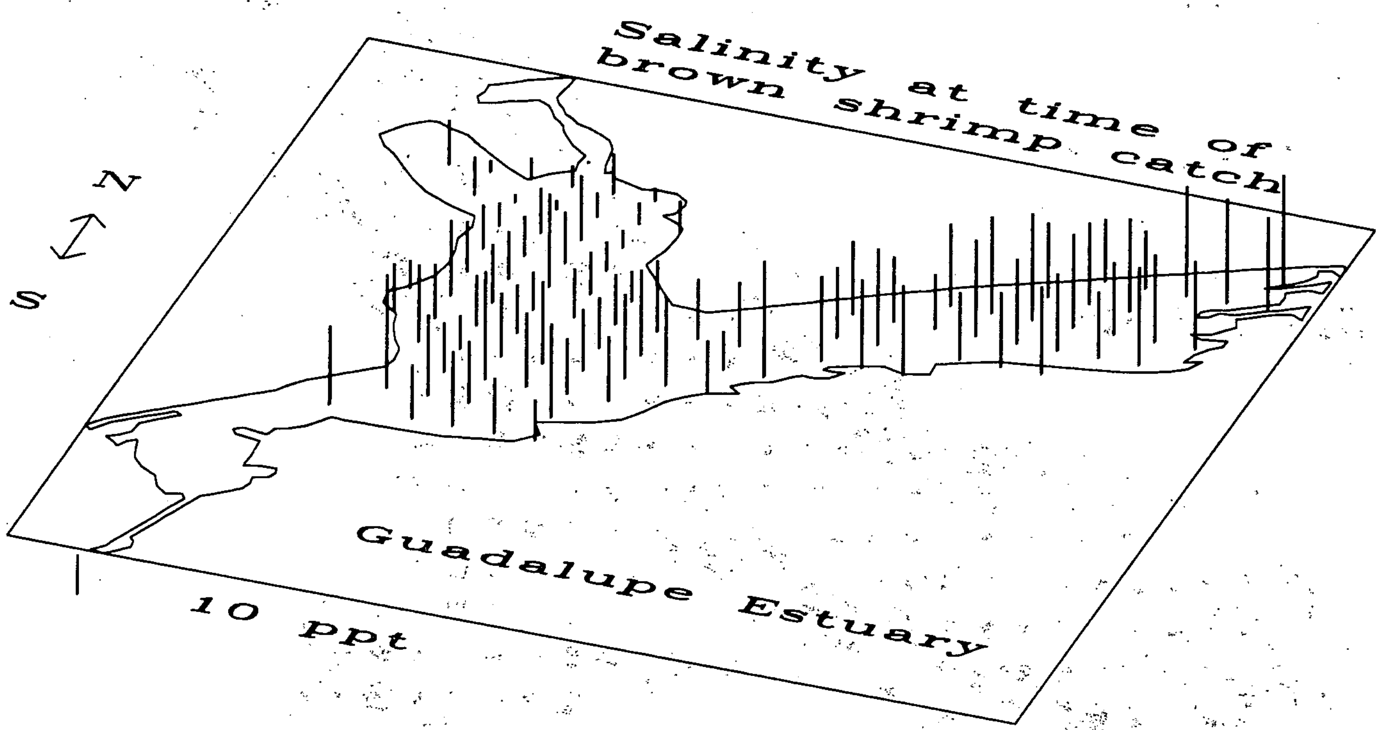


Figure 6.2.8. Salinity distribution at the time of brown shrimp *(Penaeus aztecus)* capture in the Guadalupe Estuary, from trawl samples.

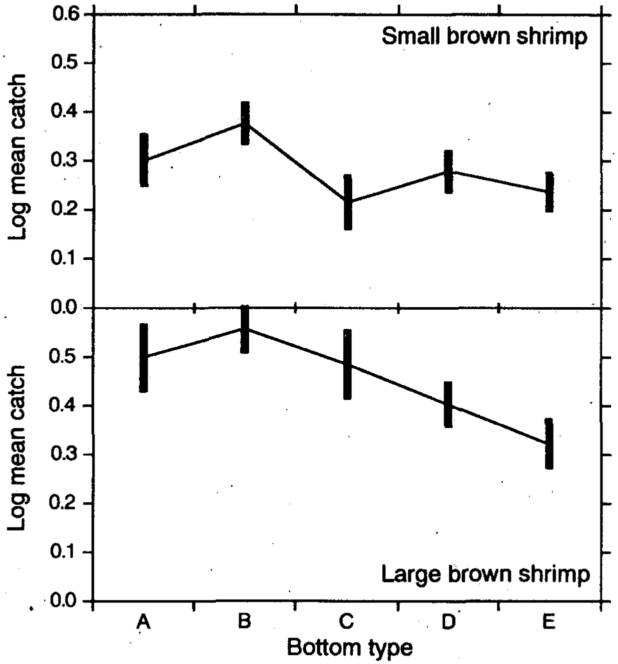


Figure 6.2.9. Relative abundance of juvenile brown shrimp (*Penaeus aztecus*) among the different sediment types (A: mud, B: mud-sand, C: mud-shell, D: mud-sand-shell, E: sand-shell-rock) in the Guadalupe Estuary, from bag seine samples.

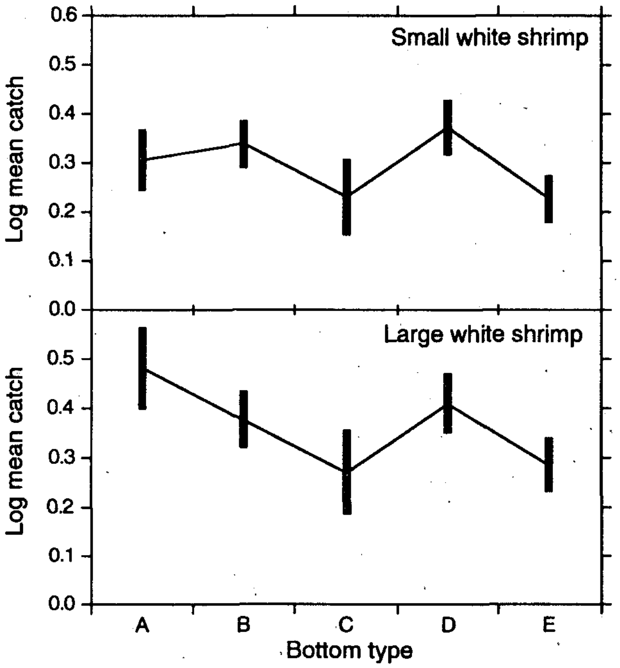


Figure 6.2.10. Relative abundance of juvenile white shrimp (*Penaeus setiferus*) among the different sediment types (A: mud, B: mud-sand, C: mud-shell, D: mud-sand-shell, E: sand-shell-rock) in the Guadalupe Estuary, from bag seine samples.

optimum salinity for juveniles in Texas waters. Yokel (1966) reported a direct relationship between salinity and red drum size, stating that juveniles appeared more tolerant of low salinity while adults were more tolerant of high salinity.

In Caminada Bay, Louisiana, red drum juveniles were collected in salinities ranging from 4 to 27‰; 91% of the catch, however, came from areas with salinities between 16 and 25‰. The highest mean catch rates were recorded at stations with mean seasonal salinities from 15 to 20‰, followed by stations with mean salinities from 20 to 25‰, and then 10 to 15‰ (G.W. Peterson, Louisiana State University; pers. comm.). Red drum juveniles showed no preference for high or low river flow in the Matagorda Bay system, although a weak association was detected between juvenile biomass and river inflow in the Tres Palacios region (Wetzel and Armstrong 1987). In San Antonio Bay, TPWD bag seine monitoring data also indicated no relationship between small red drum 30 to 70 mm (1.2 to 2.8 inches) TL catch and salinity at the site and time of sampling (Loeffler et al. 1990). Temperature and depth were the only two factors influencing small red drum catch in the bay. Larger red drum (71 to 150 mm or 2.8 to 6 inches) however, were abundant in areas having either high salinities (20 to 32‰) or low salinities (4 to 10‰). They tended to avoid areas with salinities ranging from 12 to 18‰.

Gulf Menhaden (*Brevoortia patronus*)

Gulf menhaden are not only commercially valuable but also ecologically important along the Gulf coast. In estuaries, they are important prey for red drum and seatrout. In coastal waters, the population supports the largest single fishery (by weight) in the United States. Their distribution is restricted to the Gulf states and centered in Louisiana and Mississippi waters up to 4.8 km (3 mi) offshore (Nelson and Ahrenholz 1986).

Life cycle. Gulf menhaden have a life cycle similar to other estuary-dependent species (i.e., red drum and white shrimp). It begins in coastal waters from 2 to 128 m (6 to 420 ft) deep, where spawning takes place from October to March. Menhaden larvae spend three to five weeks at sea and then gradually move inshore. Larvae are about 15 to 25 mm (0.6 to 1 inch) TL when they migrate into the estuaries (Warlen 1988). The reported timing of their movement extends from October to May. Once in the estuaries, they transform to juveniles in four to six weeks, and remain in the upper bay nursery area. The reported duration of their stay in estuaries is variable, but fishery data suggests that most menhaden migrate to offshore Gulf waters to spawn when they are about one year old (fork length, 100 mm or 4 inches), and this offshore movement generally occurs during the fall and winter (Lewis and Roithmayr 1981).

Eggs and larvae. The females spawn 40,000 to 700,000 eggs, depending on both age and size of the fish. Menhaden are considered to be intermittent spawners and may spawn up to six times during the season (Etzold and Christmas 1979). Eggs hatch within two days at 20 °C (68 °F) and 30‰ salinity. Growth rates at the early life stage average 0.3 mm (0.01 inch) per day at 20 °C (68 °F) (Lewis et al. 1987). While eggs and early larvae are stenohaline and associated with the higher salinities of Gulf waters, postlarvae appear to be euryhaline and most often associated with low salinity areas.

Juveniles. Juveniles occupy a wide range of salinities in the estuary, ranging from fresh to hypersaline. In general in Texas, juvenile menhaden were reported most abundant at salinities of 0 to 12‰ and at locations from secondary to tertiary bays (Wetzel and Armstrong 1987). Menhaden in Lavaca Bay were found year-round, with seasonal peaks observed in winter and spring. Juveniles preferred freshwater sections of both Lavaca Bay and Tres Palacios Bay. Tagatz and Wilken (1973) reported that young menhaden often entered fresh water but never penetrated far (1.6 km or 1 mi) beyond the interface with low salinity estuarine waters. The catches in Lavaca Bay varied annually, and ranged from 1.6 to 14.2 catches per unit effort (CPUE) during the 1973 to 1975 period (Gilmore et al. 1976). Highest monthly mean catch was 23.3 in March compared with 0.04 in December of 1974. In Mississippi Sound, high catches of menhaden were consistently restricted to waters over 20 °C (68 °F), particularly 25 to 34.9 °C (77 to 95 °F), and between salinities of 5 and 25‰. The highest catches came from locations with salinities of 5 to 9.9‰ and temperatures of 30 to 34.9 °C (86 to 95 °F).

Copeland and Bechtel (1974) analyzed the relationship between Gulf menhaden and environmental factors including temperature, salinity, season, and location within the Gulf estuaries. Catch ratio, instead of abundance, was used to determine the range of conditions within which organisms were collected. Of the total 2,377 samples taken along the Gulf coast, 1,284 samples contained menhaden specimens. The catch ratios were reported to be positively correlated with temperature when it was higher than 20 °C (68 °F) while juvenile catch ratios displayed a negative correlation with salinity. When salinity was greater than 25‰, the catch ratios dropped to nearly zero. Wetzel and Armstrong (1987) concluded that the optimum temperature-salinity combination for Gulf menhaden was 25 to 30 °C (77 to 86 °F) and 0 to 12‰. The combination of low temperature and low salinity was found most detrimental to survival and was avoided by juveniles. Lewis (1966) indicated that when acclimation temperature increased, the upper and lower tolerance limits of salinity also increased. Lichtenheld and Hettler (1968) further suggested that salinity could be a factor controlling menhaden abundance in estuaries when temperatures were higher than 20 °C (68 °F).

TPWD bag seine samples from San Antonio Bay did not show any correlation between relative density and salinity (Loeffler et al. 1990). For small menhaden (less than 35 mm or 1.41 inch), significant negative correlations with bottom type and wind speed were deleted at the 1% level. With larger menhaden (36 to 57 mm or 1.42 to 2.24 inches) caught in bag seines, a significant negative correlation with barometric pressure ($P < 0.05$) and positive correlations with dissolved oxygen squared, turbidity, and moon phase were obtained.

Spotted Seatrout (*Cynoscion nebulosus*)

Spotted seatrout, found in estuaries along the East and gulf coasts of the United States, is primarily a southern species and is most abundant in Florida, Mississippi, and Alabama (Tabb 1966). Unlike other sciaenids, spotted seatrout spawn and grow within shallow estuaries. They seldom move far from their territory unless conditions become intolerable (Perret et al. 1980). In estuaries, they are found in a wide variety of habitats including upper tidal marsh, tidal creek, beaches near inlets, seagrass meadows, sounds near oyster reefs, shell reefs, and submerged islands (Tabb 1966).

Life cycle. Spotted seatrout are considered to be estuarine species; they depend on the bays and estuaries for feeding, spawning, and nursery ground. Along the Gulf coast, fish have been found to spawn continuously from February through September (Brown-Peterson et al. 1989). Two spawning peaks have been observed during the year, one in the spring and the other during late summer. Simmons (1951) reported that spawning in Texas did not begin until the water temperature reached 21 °C (70 °F), while Jannke (1971) indicated 24 °C (75 °F) to be the critical temperature for heavy spawning in Florida. Spotted seatrout school when they reach 25 to 51 mm (1 to 2 inches) TL; schooling behavior continues through adult stages. Seasonal movements have generally not been reported, but one tagging study from Galveston Bay showed cyclic movement from the upper portion of Bastrop Bayou in the fall back to San Luis Pass during the spring and summer (Baker et al. 1986).

Eggs and larvae. Females may produce up to 1.5 million eggs in deep channels adjacent to grass flats (Tabb 1966) or near offshore barrier islands (Christmas and Waller 1973). Following spawning, eggs may sink to the bottom or remain bouyant depending on spawning and ambient salinities (Tabb 1966). They generally hatch within 24 hours, when water temperatures are higher than 25 °C (77 °F).

Newly hatched larvae are rarely collected and described from field studies. In Tampa Bay, Florida, larval seatrout up to 8 mm (0.31 inch) were caught in water having temperatures ranging from 20.4 to 32.9 °C (69 to 91 °F) and salinities of 18 to 32‰ (McMichael and Peters 1989). In general, larvae in Tampa Bay preferred salinities of 24 to 30‰. In the laboratory, the optimum temperature-salinity combination was reported to be 28 °C (82 °F) and 28‰ (Taniguchi 1980). He predicted 100% larval survival from salinities ranging from 18.6 to 37.5‰ and temperatures from 23 to 33 °C (73 to 91 °F). Tabb (1966) postulated that low salinities caused by strong freshets in the southern states may cause mass mortalities of larvae and juveniles.

Juveniles. Juvenile seatrout (less than 250 mm or 10 inches) are generally abundant in areas having salinities ranging from 15 to 35‰. In the Laguna Madre, juveniles were most abundant in the fall and were collected in areas having salinities greater than 60‰ (Simmons 1957). Abrupt decreases in either temperature or salinity often cause movement of juveniles to more stable environments such as deep channels or tidal passes. In Louisiana, juveniles were caught in salinities ranging from 6 to 31‰, but the majority were from waters of 12 to 25‰ salinities. Like red drum, the highest mean catches were at stations with a mean seasonal salinity from 15 to 20‰ (Peterson, pers. comm.). Loman (1978) reported high catches from waters with temperatures from 25 to 30 °C (77 to 86 °F), while Tabb (1958) suggested 15 to 27 °C (59 to 81 °F) as suitable temperatures in Florida. Along the Georgia coast, juvenile spotted seatrout began moving back into shallow waters as temperatures warmed to approximately 17 °C (63 °F) (Mahood 1974).

Holt and Arnold (1989) conducted a two-year study on finfish in the Lavaca-Colorado Estuary and a one-year study in the Guadalupe Estuary. They found that spotted seatrout were rarely encountered in Lavaca Bay in either year. During the Lavaca study, the salinity in year two (1986) was much higher than in year one (1985). In Mesquite Bay during the Guadalupe study, juveniles smaller than 30 mm (1.2 inch) were present from spring to fall at a density of 0.5 to 1.3 per 10 m^2 compared with a density of 0.1 to 0.5 per 10 m^2 in San Antonio Bay. Juveniles were not captured in trawl samples taken from open waters in either bay. Comparing these studies, it is clear that Lavaca Bay and its delta were not used by spotted seatrout, and the upper San Antonio Bay did not produce spotted seatrout catches during high-flow periods. In contrast, Roger et al. (1984), in a study of a salt-marsh estuary in Georgia, found that several fish including Atlantic croaker, spot, and flounder commonly used upper estuarine areas as nursery grounds during periods of high river inflow (up to 100 days of freshwater encroachment).

When species density was regressed on salinity and bay, Holt and Arnold (1989) found that juvenile abundance was positively related to salinity. Densities were usually the lowest at vegetated sites with lower salinities, while densities were highest at vegetated sites with higher salinities. Based on oxygen consumption rates of the fish collected from Aransas Bay, Wakeman and Wohlschlag (1978) reported that optimum salinity for juvenile spotted seatrout was 20 to 25‰ at 25 °C (77 °F). Metabolic scope was reduced above or below that salinity range. The fish were stressed if salinity dropped below 10‰ or rose above 40‰. Data from TPWD bag seine samples agreed well with these findings. In San Antonio Bay, small spotted seatrout (30 to 70 mm or 1.2 to 2.8 inches TL) were most abundant at 20 to 25‰, and decreased in their relative densities at salinities above 30‰ or below 25‰. Larger juvenile spotted seatrout (71 to 150 mm or 2.8 to 5.9 inches TL), however, were mostly caught at less saline sites with salinities ranging from 5 to 10‰.

Conclusion

Groups with different life cycle patterns. Based on life history associated with spawning migration, two groups can be defined for the species examined in this study. The first group spends its entire life cycle in the bays and estuaries and seldom moves to the Gulf; species of this group depend entirely on estuaries for spawning or feeding. These species are potentially most affected by salinity variations in an estuary. Examples belonging to this group are the eastern oyster and spotted seatrout. The second group spawns in coastal waters or passes to the Gulf, and then migrates to estuaries as they grow into postlarvae or early juveniles. Examples include blue crab, shrimp, red drum, and gulf menhaden. Adults of this second group are affected less by variations in estuarine salinity because they spend all or most of their adult life in the sea.

Salinity in Gulf of Mexico waters is relatively high and less variable compared to estuarine waters. Therefore, eggs and larvae of the second group are more or less stenohaline and require salinity of 30‰ or greater for successful survival and growth. However, juveniles of both groups are confined to estuaries and are directly affected by freshwater inflow or tidal currents. In order to survive, grow, and take advantage of the high productivity in estuaries, their physical tolerance to variations in salinity, temperature, and possibly other abiotic factors must be high. In general, residents in estuaries are euryhaline and are able to tolerate a wide range of salinities. The relationship between juveniles and freshwater or salinity is summarized for each species below.

Blue crab. Juvenile blue crabs are generally most abundant in the vegetated habitats in the lower and middle bay where salinities range from 6 to 25‰, and temperatures range from 20 to 30 °C. In San Antonio Bay, trawl samples show significantly higher catches both in areas with mean salinities of less than 20‰ and in areas influenced by Guadalupe River flow. They avoid the delta areas when freshwater inflow is high and mean salinity is less than 1.5‰. Juveniles exposed to salinity below 1‰ during the summer usually encounter heavy mortality.

Oyster. Oysters survive best in temperatures of 20 to 30 °C and in salinities of 10 to 30‰. In San Antonio Bay, the largest and most numerous oyster reefs are located in areas with salinities of 10 to 24‰. Salinity of about 15‰ is the suggested optimal for survival and reproduction. In addition, fluctuating salinity in the range of 10 to 30‰ often promotes more rapid oyster growth than relatively constant salinity.

White shrimp. White shrimp occur over a wide range of salinities, but were most abundant in less saline areas of the estuarine system. In San Antonio Bay, they are significantly more abundant in areas with low mean salinities (2.2 to 7.7‰) than in areas with high mean salinities (24.2 to 26.2‰). White shrimp are reported to grow fastest in temperatures between 20 and 32 °C (68 to 90 °F). In addition, a positive relationship between rainfall and shrimp production has also been documented for this species in Texas.

Brown shrimp. Brown shrimp prefer zones of high salinity in estuaries. In San Antonio Bay, the polyhaline areas (24.2 to 26.2‰) yielded significantly greater shrimp catches than the less saline areas. Juvenile brown shrimp density was estimated to be 20 to 30 per 10 m^2 in the lower bay, but was less than 5 per 10 m^2 in the upper bay. Peak abundance in the field was collected in temperatures between 15 and 33 °C (59 to 91 °F).

Red drum. Red drum are considered to be euryhaline and are collected in a wide variety of habitats. They have been found in waters with temperatures of 2 to 34.9 °C (36 to 95 °F) and salinities of 0 to 45‰, but appear to prefer salinities of 20 to 35‰. In San Antonio Bay, juvenile red drum (71 to 150 mm or 2.8 to 5.9 inches TL) were abundant in either polyhaline areas having salinities 20 to 32‰ or less saline areas with salinities of 4 to 10‰. Areas having salinities from 12 to 18‰ had lower catch rates.

Menhaden. Juvenile menhaden occupy a wide range of salinities in estuaries, ranging from oligohaline to hypersaline, but are reported most abundant at 0 to 12‰ salinity and in secondary or tertiary bays. In Texas waters, catch successes are reported to be negatively correlated with salinity but positively related to temperature. A low temperature-low salinity combination (e.g., 6 °C or 43 °F and 5‰ salinity) was found most detrimental to survival and, hence, was avoided by juveniles.

Spotted seatrout. Spotted seatrout spend most of their life in estuaries. They are collected in various habitats including upper tidal marsh, seagrass meadow and shell reefs. They seldom move far from their territory unless conditions become intolerable. Abrupt decreases in salinity often cause movement to deep channels or tidal passes. Spotted seatrout are generally abundant in areas with salinity of 15 to 35‰. In Texas and Louisiana waters, high mean catches were found in areas with a mean seasonal salinity between 20 to 25‰ and a mean temperature of 25 °C (77 °F).

The effect of salinity on juveniles. Salinity is a major factor influencing species distribution and abundance in estuaries. However, salinity and other factors interact, and salinity alone seldom explains large portions of the variation in species abundance and distribution. Factors such as temperature, vegetation, sediment, and water depth of the habitats must be included in any attempt to understand the mechanisms underlying the population dynamics of estuarine species.

Each species discussed in this section has its unique life history parameters. Their niches may partially overlap, but in general they are temporally and spatially separated, forming a dynamic mosaic of estuarine communities. These animals have become adapted to the "normal" (highly variable) seasonal changes of salinity in estuaries. Salinity appears to have a dominating effect only when it varies in extremes for extended periods such as in strong freshets or droughts. These effects seem to be intensified when accompanied by very low or very high temperatures.

6.3 WETLAND HABITAT USE BY ESTUARINE ORGANISMS

Introduction

In estuaries worldwide, many organisms use wetland habitats during part of their life cycle. The effect that freshwater inflow has on wetlands and the species that use them has not been definitively demonstrated; however, there is very good evidence that a relationship exists. Odum (1974) has extensively documented primary productivity in saltwater marshes; Deegan (1986) has shown that plant growth in estuarine marsh communities, even for saltwater species, is stimulated by freshwater inflow; Turner (1977) demonstrated positive correlations between fisheries production and areal extent of marsh; and Powell (1979), through regression analysis, showed that the commercial harvest of fish was related to freshwater inflow. Some portion of all these relationships could be the result of inflow effects on wetland habitats.

Until recently, the differential use of bay habitats by species or different size groups of the same species was not well-established. Most investigations have focused on one habitat at a time or simple salinity gradients across an estuary (Peterson 1984; Renfro 1960; Schwartz et al. 1982). The studies that will be discussed in this section show that species inhabit spatially separated estuarine habitats during the same time period. The studies also show how similar habitats with different salinity regimes are used by estuarine species, and how the occupancy of wetland habitats changes during and after floods.

To determine how different habitats were used by small fishes and decapod crustaceans, and how use varied within and among the habitats with respect to salinity regime, studies were initiated which sampled habitats in several areas in the estuary within a short time period (less than one month), with the same standardized gear. This permitted direct comparisons among habitats during the same time periods. An understanding of how species use different areas of an estuary is necessary to accurately infer causal relationships with environmental variables, and to improve predictions of the effects that altering freshwater inflows to the estuary will have on fishery organisms and other life on which they depend.

Study Locations and Methods

Four different field studies (Zimmerman et al. 1989, 1990a, 1990b, and 1990c) were completed in three estuaries (Trinity-San Jacinto, Lavaca-Colorado, and Guadalupe). The field studies compared the occurrence of fishery species in various habitats (emergent marsh, seagrass, oyster reef, and bare bottom) and salinity regimes, and assessed the effect of flood events on different estuarine species. No single estuary in which the studies were conducted had all plant species or habitat types present, but collectively the estuaries studied represented most of the dominant plant communities found within Texas estuaries.

In laboratory experiments, Minello et al. (1990) examined the effects of vertical structure, turbidity, light, temperature, bottom type, food, and salinity on white shrimp and brown shrimp behavior. Minello et al. (1989) compared the stomach contents from fish caught in the delta and outer bay areas of Lavaca Bay during October (1985), May (1986), and August (1986). The results were used to help interpret data from the field studies.

Trinity-San Jacinto Estuary. Sampling in the Trinity-San Jacinto Estuary was done in three areas: the Trinity Delta, a mid-bay area (Smith Point), and a lower-bay area (Figure 6.3.1). In the Trinity Delta, sampling was conducted in areas comprised mainly of emergent marsh species (chiefly bulrush, *Scirpus maritimus*) and submerged vascular plant species (widgeongrass, *Ruppia maritima*; water nymph, *Najas* sp.; and wild celery, *Vallisneria* sp.) (Zimmerman et al. 1990a). In the mid-estuary region, sampling was conducted in areas with smooth cordgrass (*Spartina alterniflora*), needlerush (*Juncus roemerianus*), big cordgrass (*S. cynosuroides*), and saltgrass (*Distichlis spicata*). There was no submerged vascular vegetation present within the sampled mid-bay area. In the lower-estuary region, sampling was done at sites that were mainly comprised of emergent marsh species including smooth cordgrass, perennial glasswort (*Salicornia virginica*), and saltwort (*Batis maritima*). At one lower-bay site (Christmas Bay), plants consisted of emergent and submerged species: perennial glasswort, shoalgrass (*Halodule wrightii*), widgeon grass, turtle grass (*Thalassia testudinum*), and the seagrass *Halophila engelmanni*.

Lavaca-Colorado Estuary. The Lavaca Bay study (Lavaca-Colorado Estuary) had sample sites located in two different emergent marsh communities within Lavaca Bay (Figure 6.3.2). The vegetation of the three deltaic sites was

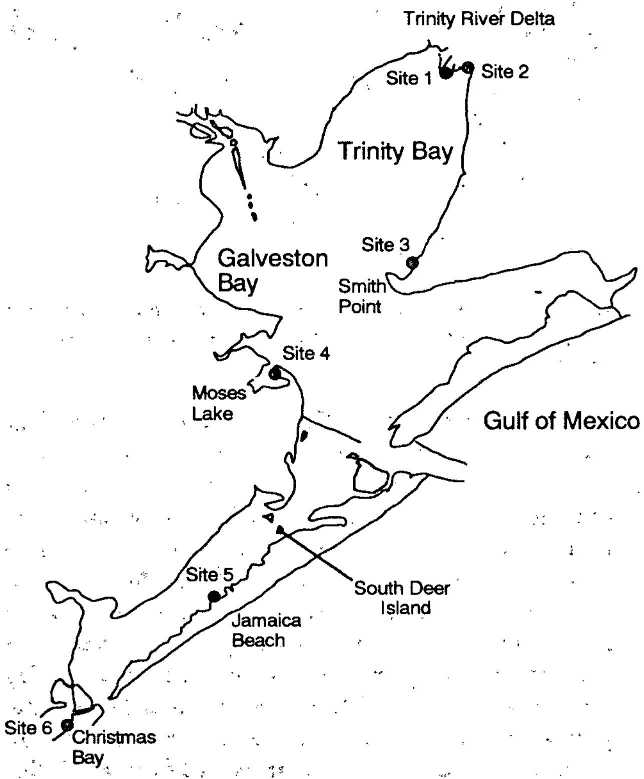


Figure 6.3.1. Sample sites within the Trinity-San Jacinto Estuary for comparison of marsh use by small fish and decapod crustaceans.

dominated by needlerush, and the marshes at the three sites located in secondary bays were predominantly smooth cordgrass. No submerged vascular vegetation was present at these sites (Zimmerman et al. 1990c).

Guadalupe Estuary. Vegetation in the Guadalupe Estuary sampling sites in the Guadalupe delta area consisted of sparse widgeon grass, water nymph (*N. guadalupensis*), and smooth cordgrass. In the lower sites, vegetation consisted of shoalgrass, widgeon grass, and smooth cordgrass (Figure 6.3.3, Zimmerman et al. 1990a).

Study design. In all three estuaries, samples were taken in bare, unvegetated bottom adjacent to the marsh, in the emergent marsh, and within submerged seagrass beds when present. Comparative sampling on oyster reef (composed mostly of *Crassostrea virginica*), marsh, and unvegetated habitat was accomplished only in the vicinity of Confederate Reef near South Deer Island in the eastern end of West Bay (Figure 6.3.1, Zimmerman et al. 1990c).

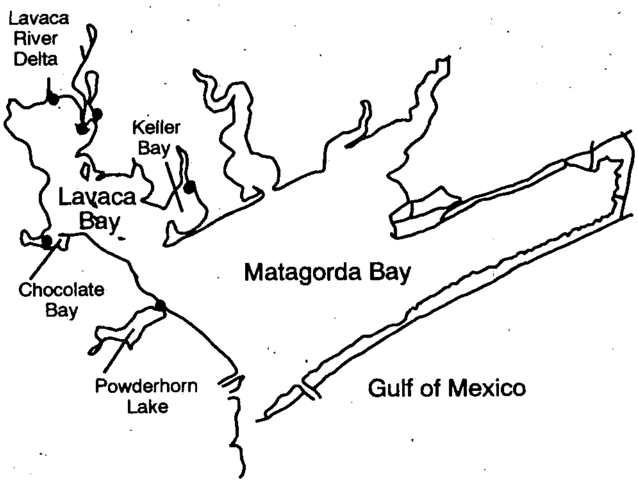


Figure 6.3.2. Sample sites within Lavaca Bay, lower *Spartina* marshes, and delta *Juncus* marshes for comparison of marsh use by small fish and decapod crustaceans.

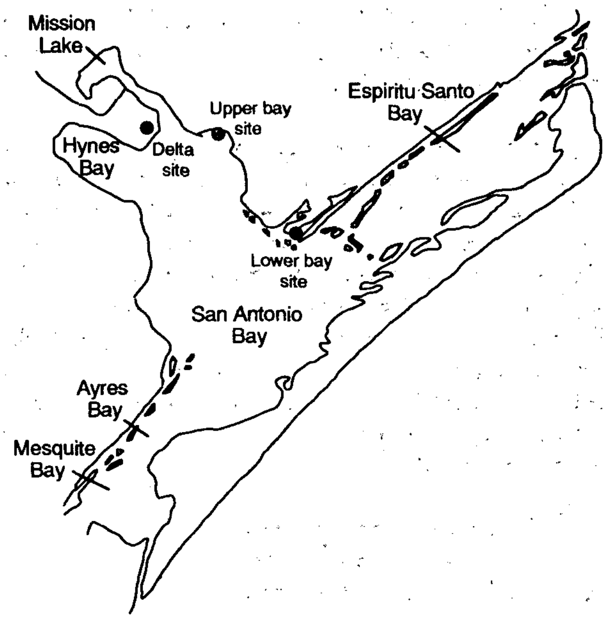


Figure 6.3.3. Sample sites within the Guadalupe Estuary for comparison of marsh use by small fish and decapod crustaceans.

Salinity regimes. Salinity varied over a wide range during these studies and included oligohaline (0.5 to 5‰), mesohaline (5 to 18‰), and polyhaline (18 to 30‰) conditions and transitions. Within the Trinity-San Jacinto Estuary, salinities ranged from 0 to 11‰ in the Trinity Delta area (Figure 6.3.4, sites 1 and 2), to 24 to 32‰ at Christmas Bay site 6. Sites located within the estuary between these two extremes generally had salinities which were intermediate to these values. Salinities at Lavaca Bay sites (Lavaca-Colorado Estuary) ranged from 11 to 28‰, generally mesohaline to polyhaline conditions (Figure 6.3.5). Salinities in the Guadalupe Estuary ranged from oligohaline at the delta to polyhaline in lower San Antonio Bay (Figure 6.3.6). The Lavaca Delta sites were slightly less saline, but they were never as fresh as the sampling sites in the Guadalupe Delta or the Trinity Delta areas except after a major flood event. Salinities in the Lavaca Delta area decreased by 12 to 22‰ during the period May 15 through May 25, 1987, to 1‰ by June 7, 1987 (Figure 6.3.7).

Collecting gear. A cylindrical drop trap 1.8 m (6 ft) in diameter was used as the collecting gear in these studies. The trap caught a full range of organisms. However, 99% of the fish and 99.9% of the invertebrates captured were less than 50 mm (2 inches) TL. Consequently, the results of these studies apply mainly to postlarvae and juveniles, and the interpretations of results from these must be restricted to the small organisms represented in the samples studies (Zimmerman and Minello 1984).

Laboratory studies. The investigators used rectangular tanks in laboratory studies filled with seawater, with green plastic drinking straws placed vertically in the sediment to simulate vegetation. Half of the tank was "planted" with the simulated vegetation in a regular pattern resulting in equally spaced clumps of four straws each. Tanks were randomly assigned to experimental treatments and various combinations of food, substrate, salinity, light, turbidity, and predators were presented to brown and white shrimp test subjects. Behavior was recorded several times throughout the day (generally five times a day).

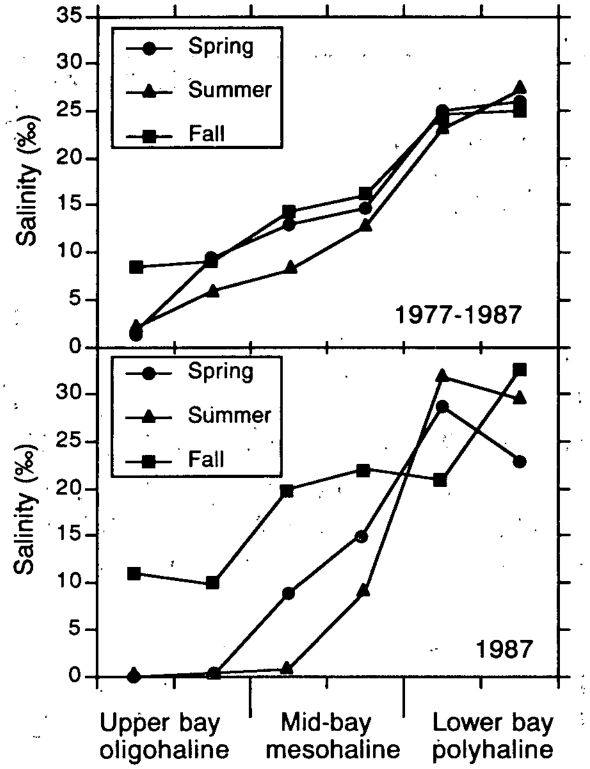


Figure 6.3.4. Mean salinities at Galveston Bay drop-sampling sites in 1987 compared to means from TPWD monitoring data taken within 1 km of each site during the period 1976 through 1987.

Results of the Field Studies

Abundance pattern in salinity zones. These studies have shown that many estuarine species including most of Texas' economically important fishery organisms make extensive use of emergent marsh surface and seagrass beds. They also demonstrated that the greatest densities of organisms were in areas having mesohaline or polyhaline conditions (figures 6.3.8 and 6.3.9). For finfish, the densities observed in oligohaline or polyhaline areas were about the same. However, the species composition between the two differed: red drum, spotted seatrout, and flounder were more abundant in polyhaline areas while Atlantic croaker, sheepshead minnow and Gulf killifish were more abundant in oligohaline areas. Decapod crustaceans had lowest densities in oligohaline areas, intermediate densities in polyhaline areas, and highest densities in mesohaline areas (figures 6.3.8 and 6.3.9).

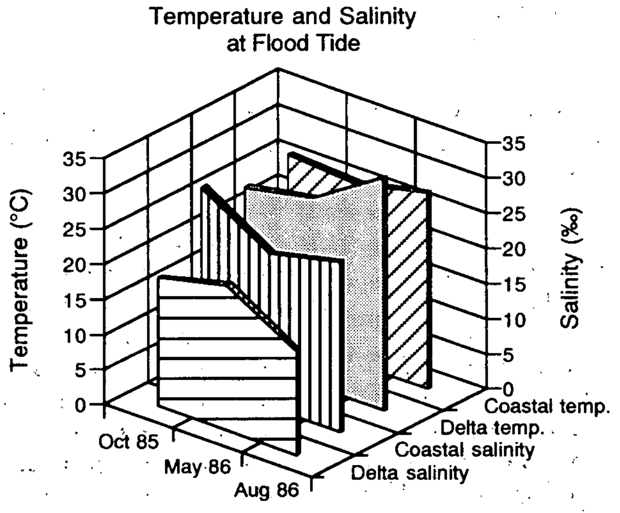


Figure 6.3.5. Temperature, salinity, and water depth associated with *Spartina* (coastal) marshes and *Juncus* (delta) marshes in Lavaca Bay.

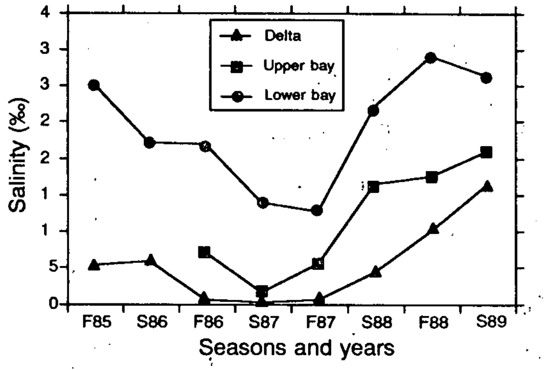


Figure 6.3.6. Salinity in the Guadalupe Estuary during drop-trap sampling from fall 1985 to spring 1989, by area.

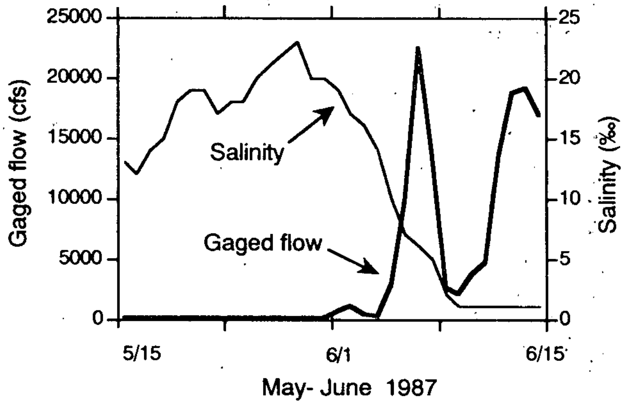


Figure 6.3.7. Salinity change in upper Lavaca Bay during flooding of the Lavaca River associated with high rainfall in May and June of 1987.

Use of oyster reef habitat. Oyster reef was used extensively by estuarine organisms but generally not by any commercially important juvenile finfish or decapod crustaceans (figures 6.3.10 and 6.3.11, Zimmerman et al. 1990c). The stone crab (*Minippe mercenaria*) was the most abundant crustacean in the oyster reef habitat. This species is commercially exploited in other states to a much greater extent than in Texas. Oyster reef samples showed abundances and diversities similar to emergent saltmarsh and seagrass meadows, but they were comprised of a different set of species.

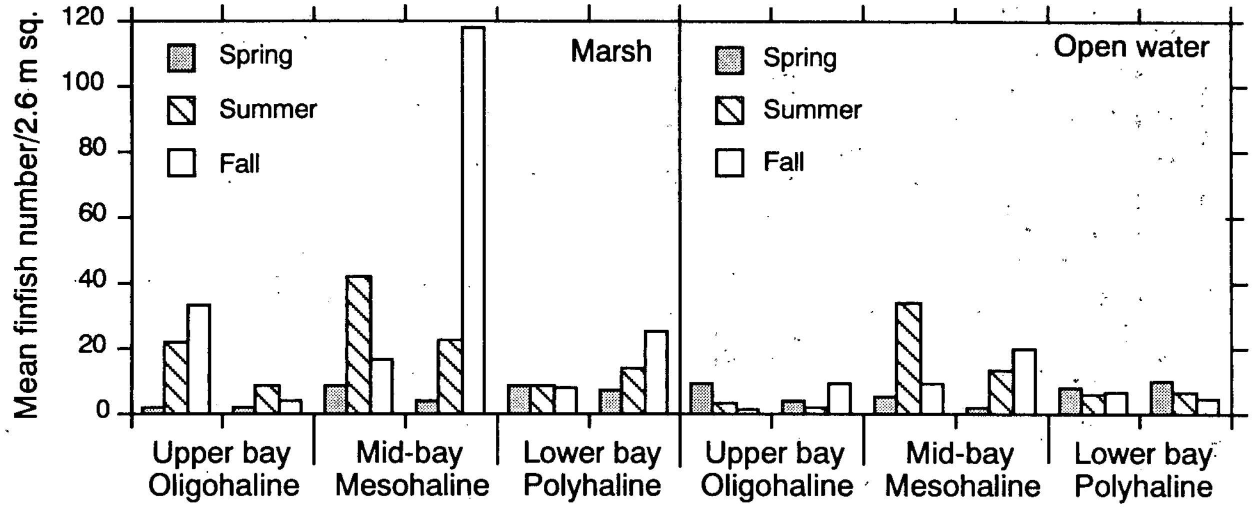


Figure 6.3.8. Mean finfish densities within the emergent marsh and adjacent nonvegetated areas in the Trinity-San Jacinto Estuary determined by drop-sampling surveys during 1987.

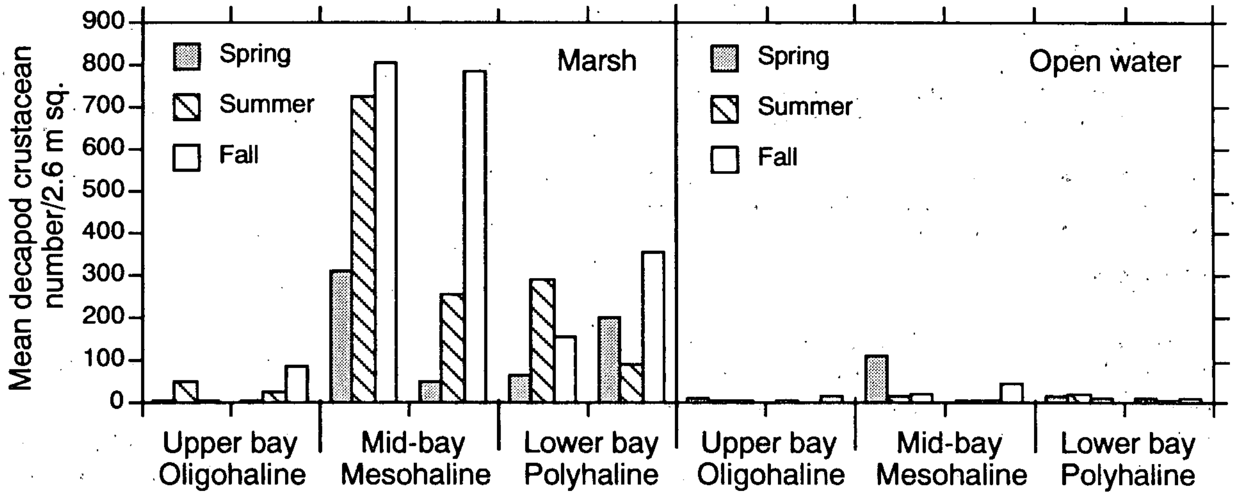


Figure 6.3.9. Mean decapod crustacean densities within the emergent marsh and adjacent nonvegetated areas in the Trinity-San Jacinto Estuary determined by drop-sampling surveys during 1987.

Forage species habitat use. Finfish species caught at the highest densities and in the most different places included the bay anchovy (*Anchoa mitchelli*), Gulf killifish (*Fundulus grandis*), naked goby (*Gobiosoma bosci*), striped mullet (*Mugil cephalus*), and bay silversides (*Menidia beryllina*). These species were low in the food chain and served as forage for larger and more economically important species.

Habitat use by economically important finfish. None of the three major economically important finfish caught with the drop trap (red drum, *Sciaenops ocellatus*; spotted seatrout, *Cynoscion nebulosus*; and southern flounder, *Paralichthys lethostigma*) were captured at sites having the lowest salinity levels (Figure 6.3.12). These three species had about the same densities in emergent marsh and open bay bottom indicating that the use of these two areas was about equal.

Redfish, spotted seatrout, and southern flounder are obligate carnivores (high on the food chain) and never reach the abundance levels of omnivorous species which use plants, detritus, and animal material as their food sources. Because of the low densities and the modest amount of data available from these studies, it is not possible to conclusively state that these carnivorous fish do not use the freshest areas of the estuaries. The evidence at this time, however, suggests that they do not. This is consistent with observations by Peterson (1984) who stated that spotted seatrout (5 to 140 mm or 0.25 to 6 inches TL) in Caminada Bay, Louisiana

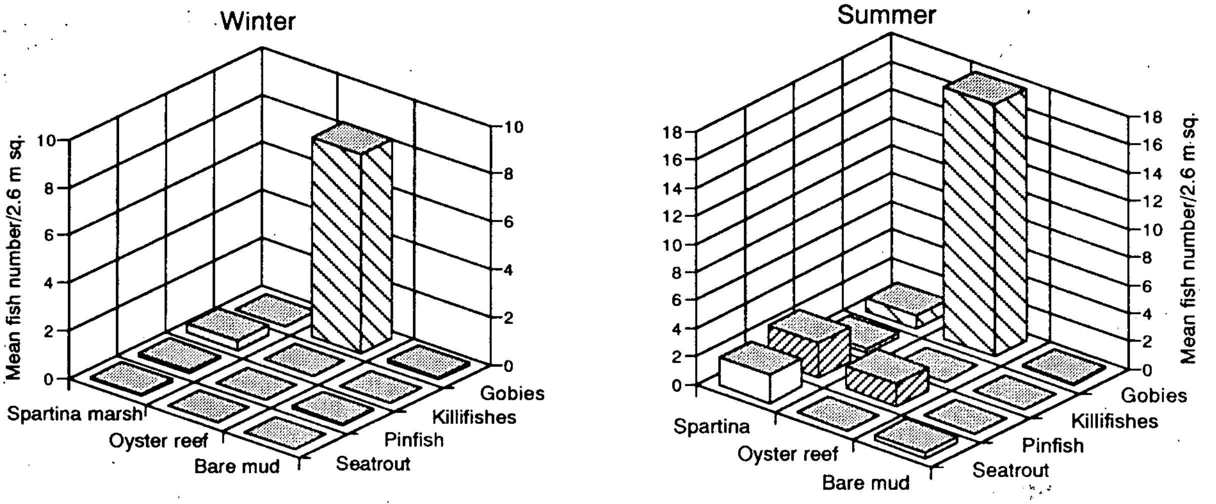


Figure 6.3.10. Relative densities among predominant fish in oyster reef, salt marsh, and bare mud habitats in West Bay (Trinity-San Jacinto Estuary) during December 1988 and July 1989.

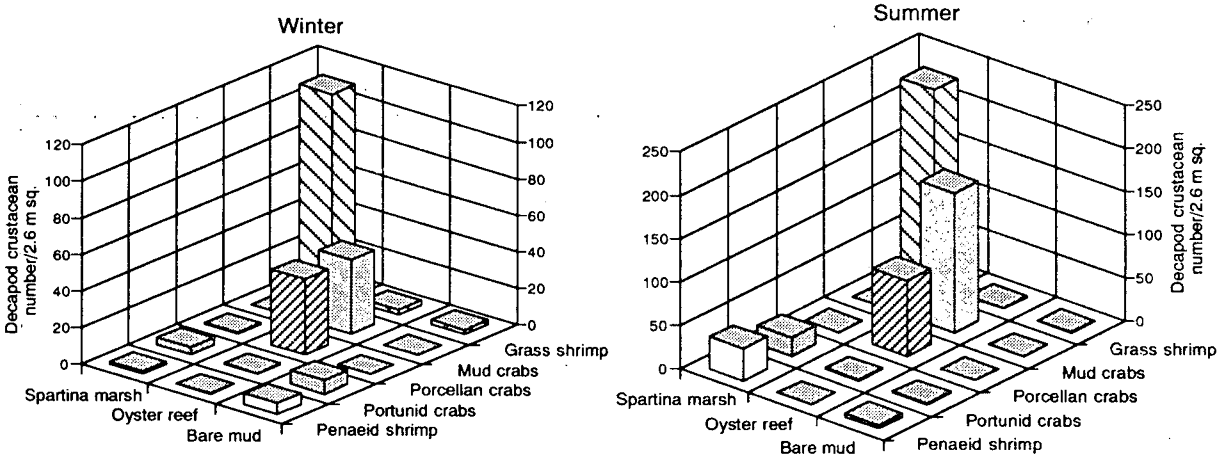


Figure 6.3.11. Relative densities among predominant decapod crustaceans in oyster reef, salt marsh, and bare mud habitats in West Bay (Trinity-San Jacinto Estuary) during December 1988 and July 1989.

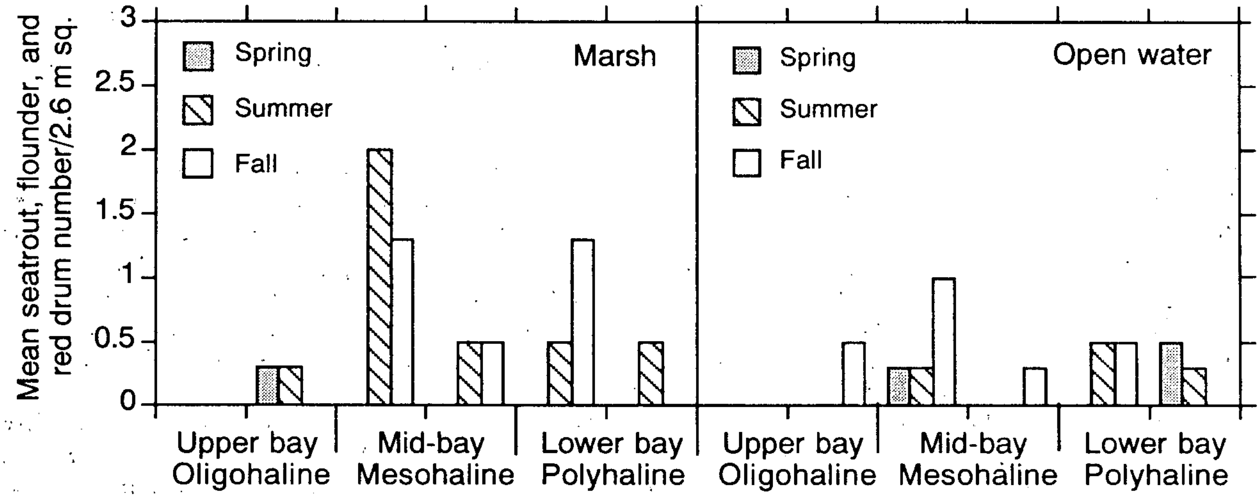


Figure 6.3.12. Mean relative densities of spotted seatrout, southern flounder, and red drum by site (salinity conditions) and presence (marsh surface) or absence (open water) of vegetation within the Trinity-San Jacinto Estuary.

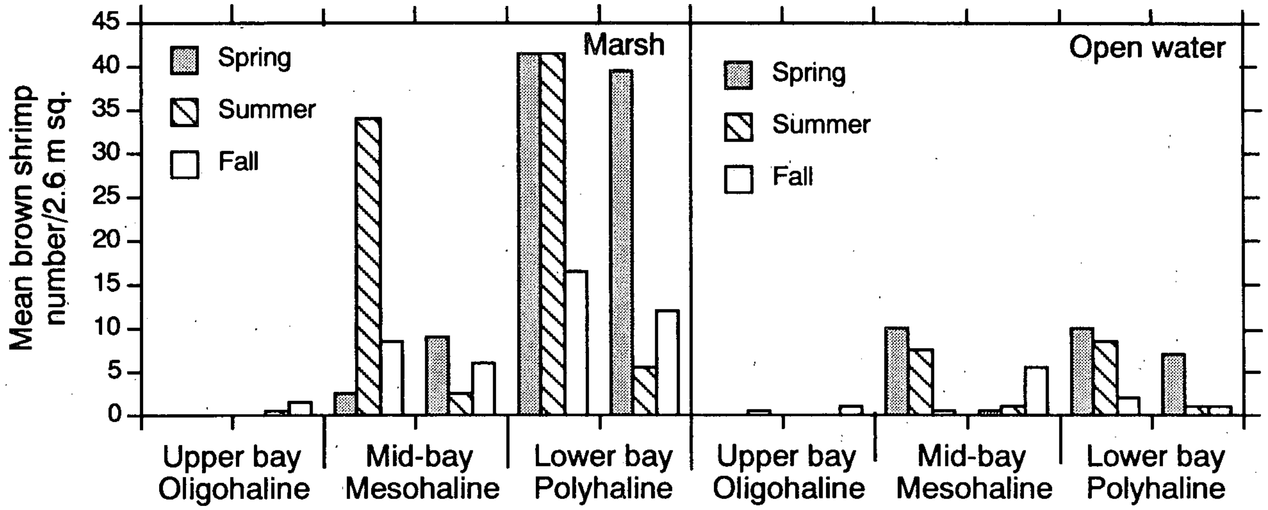


Figure 6.3.13. Mean densities of brown shrimp by site (salinity conditions) and presence (marsh surface) or absence (open water) of vegetation within the Trinity-San Jacinto Estuary.

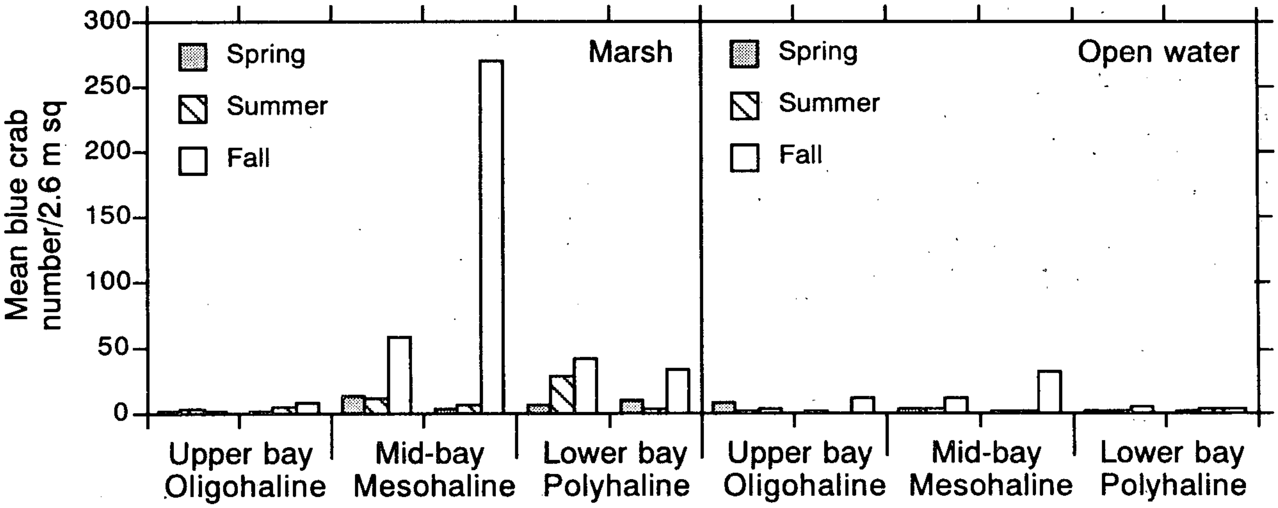


Figure 6.3.14. Mean densities of blue crab by site (salinity conditions) and presence (marsh surface) or absence (open water) of vegetation within the Trinity-San Jacinto Estuary.

preferred shallow- to medium-depth water in saltmarsh with broken shorelines. He also stated that all catches were made in salinities ranging from 6 to 31‰. All red drum catches in this study occurred in salinities ranging from 4 to 27‰.

At least one species important to Texas recreational fishing, Atlantic croaker (*Micropogonias undulatus*), does use the oligohaline area. This species was captured at every sampling station throughout the estuary.

Habitat use by shrimp and blue crabs. Brown shrimp (*Penaeus aztecus*), white shrimp (*P. setiferus*), and blue crabs (*Callinectes sapidus*) had their lowest observed densities in the oligohaline areas and their highest densities in the mesohaline or polyhaline areas. However, brown shrimp and blue crabs were caught at every station in all estuaries, indicating a very wide tolerance to salinity conditions and use of available habitat (figures 6.3.11, 6.3.13, and 6.3.14). White shrimp were also caught at every station except one, the station having the lowest salinity concentration. A prominent forage species, grass shrimp (*Palaemonetes pugio*), had a very similar distributional pattern as these other species (Figure 6.3.15). The investigators reported that differences in abundance between the oligohaline and mesohaline areas were statistically significant; abundance differences between mesohaline and polyhaline areas were inconsistent, significant in some tests but not in others.

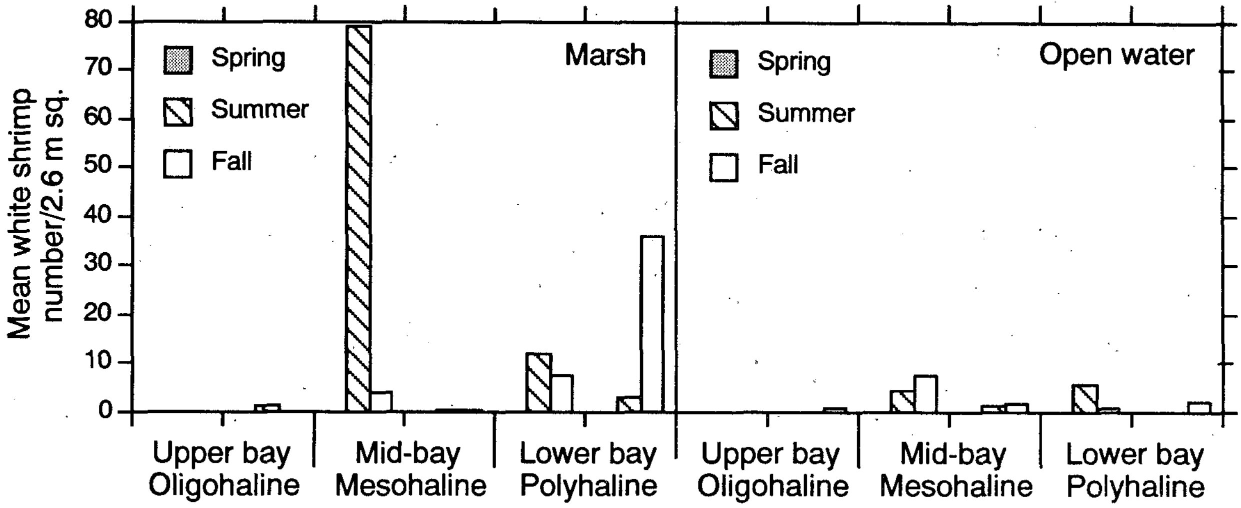


Figure 6.3.15. Mean densities of white shrimp by site (salinity conditions) and presence (marsh surface) or absence (open water) of vegetation within the Trinity-San Jacinto Estuary.

Use of vegetated versus bare habitat areas. Estuarine organisms make extensive direct use of marsh and submerged aquatic vegetation habitat. In general, postlarvae and juvenile brown shrimp reach their highest densities in marsh in mesohaline areas and polyhaline areas. Brown shrimp also reached high densities in seagrass meadows when it was well established. They use bare intertidal bottom habitat at a much lower rate (indicated by lower densities) unless vegetated habitat is not available (i.e., at extremely low tides). Grass shrimp have the same preferences as brown shrimp except they were more abundant. Juvenile white shrimp were about equally abundant in vegetated and nonvegetated habitat. This pattern was not uniform, however. During the summer, white shrimp were more abundant in vegetated marsh in mesohaline areas, and in the fall they were more abundant in the vegetated marsh in polyhaline areas.

The Zimmerman et al. (1990a) study reported significantly higher densities in nonvegetated areas for spot (*Leiostomus xanthurus*) and Atlantic croaker. Gulf menhaden (*Brevoortia patronus*) and bay anchovy showed a significant preference for open water, whereas bay silverside and naked goby used the two areas (vegetated and nonvegetated) about equally. Spotted seatrout and pinfish *(Lagodon rhomboides)* showed higher densities in marsh habitat than in bare-bottom habitat. Red drum appeared to have equal densities between marsh and nonvegetated habitat.

Effects of floods on habitat use. The field studies were able to take advantage of a period of flood flows into Lavaca Bay to determine the immediate effects of floods on the distribution of estuarine species. Zimmerman et al. (1990c) compared species densities before and after these floods. They found no significant differences for finfishes except for bay anchovy and gulf menhaden after salinities had dropped from 23 to less than 1‰ in Lavaca and Matagorda bays and remained low for more than a week (Figure 6.3.7). Both species were more abundant after the salinities dropped (Figure 6.3.16). Bay anchovy and gulf menhaden are filter feeders (Govoni et al. 1983) and may have migrated into the area to take advantage of an increased density of detritus caused by the flooding. Bay anchovy were more abundant mid-way through the flood sequence, while the bays maintained moderate salinity. The investigators also reported that brown shrimp and blue crab abundance declined significantly after salinities dropped, but that white shrimp abundance did not (Figure 6.3.17). The result for white shrimp could be an artifact because there were not many white shrimp in the bay during this time with which to assess a response.

Generalization from the field studies. These studies confirm the idea that Texas estuaries contain a large number of species having wide tolerances to varying salinities, allowing them to take advantage of the energy in nutrient materials supplied by large freshwater inflows (Copeland 1966; Wadie and Razek 1985).

The pattern emerging from these studies is that many estuarine organisms move as close to a freshwater source as their physiological limitations permit to take advantage of high nutrient concentrations, high plankton densities, and large detritus loads resulting from freshwater inflows. From the generally high densities of organisms in the mesohaline areas and the very low densities in the oligohaline areas, there may be an advantage for organisms to live close to the lower limit of their salinity tolerance. They can take advantage of conditions caused by freshwater inflows, including benefits from food and cover (Minello et al. 1989, 1990). It also

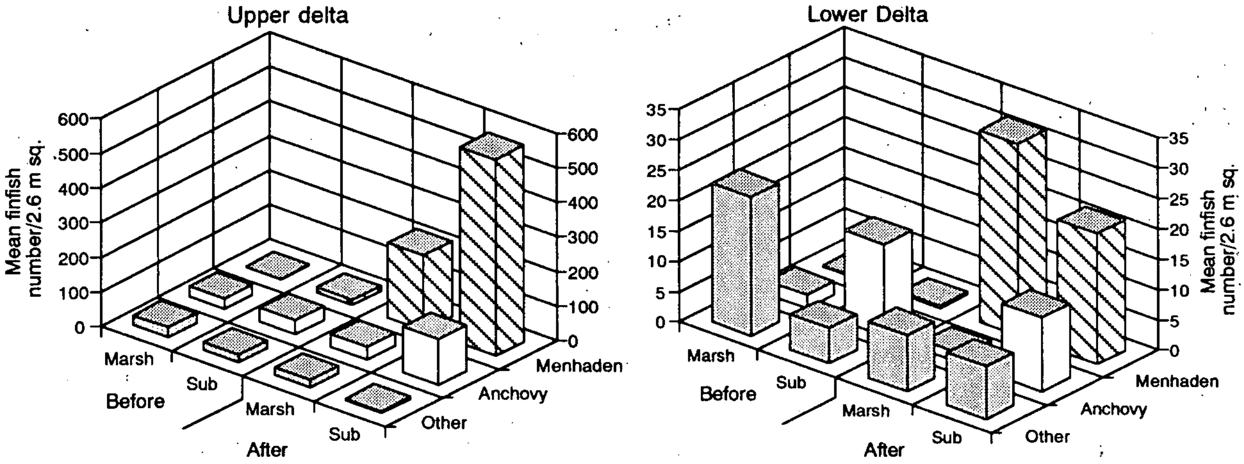


Figure 6.3.16. Mean densities among fish in microhabitats of Lavaca River delta marshes before and after freshwater flooding during May and June of 1987.

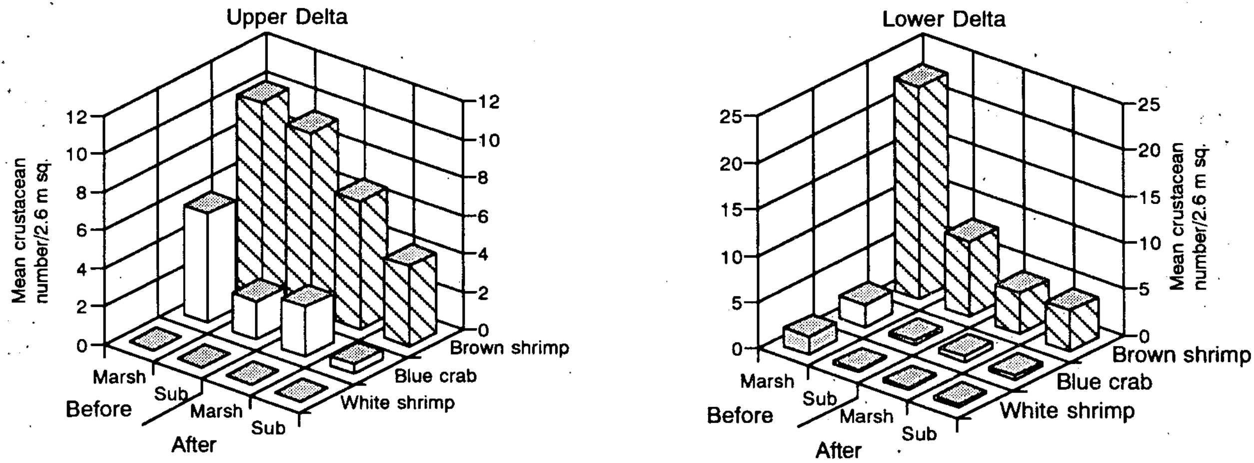


Figure 6.3.17. Mean densities among economically important crustaceans in microhabitats of Lavaca River delta marshes before and after freshwater flooding during May and June of 1987.

appears that these organisms have adapted to cope with short-term fluctuations to salinity in their environment as they did not quickly move out of recently freshened areas as Zimmerman et al. (1990c) showed by sampling before and after a flood event. This is consistent with the observations of Schwartz et al. (1982) who recorded 77 marine species that remained in water of 0‰ salinity for up to six weeks. Obviously, estuarine organisms can cope with changes in their environment, but they do not sustain populations in fresh water. Organisms may not necessarily attain their highest densities in nature in areas having abiotic conditions (i.e., temperature, salinity, oxygen, etc.) that laboratory studies define as their optima. Their distribution may be skewed from ideal conditions so that they may seek food or protection from predators.

Results of the Laboratory Studies

Minello et al. (1989) showed in the laboratory that white and brown shrimp change their behavior and distribution with respect to vegetated and nonvegetated areas with changes in their environment. They found brown and white shrimp have a strong preference for vertical structure provided by vegetation regardless of any available food. These experiments support the observations from field data that brown shrimp and, to a lesser extent, white shrimp have a strong selection for vegetated areas (81% of browns and 75% of whites were in simulated vegetated areas, respectively).

Effect of environmental change on brown shrimp distribution. The investigators found 80 to 95% of all brown shrimp to be within the vegetated areas of the tanks at all salinities (Minello et al. 1989). Attempts to alter this strong preference using food, substrate, turbidity, light, and changes in salinity did not totally remove this strong preference. At the lowest salinity (3‰), brown shrimp were more active, but they were still mainly located on the vegetated half of the tank. Decreasing the salinity from 20 to 3‰ resulted in an initial decrease in the percent of brown shrimp in the vegetated area to 50%; however, the percent in the vegetated half continued to increase during the next four hours until it was more than 75%.

The investigators were able to significantly manipulate the distribution of brown shrimp using squid as food. Placing food only on the nonvegetated side resulted in 40% of the shrimp being located on the vegetated side. However, there was no significant effect of food on activity levels. Highest mean activity was in tanks with no food (Minello et al. 1990). The lowest concentration of brown shrimp in the vegetated area (less than 30%) was attained by placing food on the nonvegetated side at night.

Minello et al. (1990) also showed that brown shrimp distribution with respect to vegetation could be altered by the availability of a substrate which enabled burrowing. Such a substrate only affected the brown shrimps' distribution during daylight hours when burrowing was more important. A sandy substrate only on the nonvegetated half into which shrimp could burrow during the day resulted in 67% of the shrimp located in the nonvegetated area. The night-time distribution was not affected by this arrangement. No measure of activity was made when shrimp were tested in high turbidity experiments, although increased activity in turbid water has been observed in other studies (Minello et al. 1987). There was no evidence for increased activity in tanks where light was reduced during the day.

Effect of environmental change on white shrimp distribution. White shrimp were tested using the same types of experiments as those used in the brown shrimp studies. White shrimp distributions were not greatly influenced by these parameters, although the experimental results from using food to alter white shrimp distributions were suspect since the investigators may not have chosen a valid food (squid) with which to test the response. The main thing that reduced the activity of white shrimp was the presence of a natural predator (flounder).

Conclusions

These studies and others demonstrate the importance of marsh, seagrass meadows, and oyster reefs to the ecological health and biological productivity of Texas estuaries. They also strongly imply that a significant loss of marsh or seagrasses would result in a loss in total productivity and increased predation; these losses would be directly observable in reductions of annual biomass in shrimp, crabs, and many forage fish on which other fishes depend.

The studies also provided a number of specific conclusions about the comparative use of habitat areas by estuarine species. Many species (but not all) were several times more abundant in marsh and seagrass habitats than in nearby open bay areas. Postlarvae and juveniles were more abundant in mesohaline and polyhaline marsh and seagrass areas than in oligohaline sites. It appears that many species tend to move as close to the oligohaline area as their physiological tolerances will allow.

Many fish species and possibly white shrimp remain in marsh habitats during post-flood, low-salinity conditions even at some metabolic cost. A few species move to these areas during or after floods, possibly to take advantage of detritus or induced plankton blooms. The abundance of brown shrimp and blue crab decline significantly after salinities drop.

In behavioral studies, brown and white shrimp preferred "vegetated" vertical structure to bare areas even when salinities dropped to low levels. This behavior was modified only if food items or preferred burrowing substrates were offered.

Oyster reefs have a similar diversity and abundance to marsh and seagrass communities. The reef community, however, contains a different set of species.

6.4 SALINITY EFFECTS ON ADULT AND JUVENILE FISH METABOLISM

Introduction

Energy use by fish. Like other animals, marine fish obtain the energy required for their life cycle from the food they eat. Conceptually, we can view the energy budget of an adult marine fish as being divided into three components—one part for body maintenance, another for growth, and the third portion for reproduction of the species. When environmental conditions become stressful, the amount of energy required for body maintenance can increase dramatically, leaving little or no energy for fish growth and reproduction (Fry 1971; Brett 1979; Brett and Groves 1979).

Environmental influences on fish energetics. Environmental factors which influence fish energetics can be classified as either controlling, masking, directive, or limiting factors (Fry 1971). For example, temperature is a controlling factor that governs chemical reaction rates; salinity is a masking factor that increases the metabolic cost of regulating internal body salts; photoperiod (the amount of daily light) is a directive factor that affects hormonal (endocrine) activity; and oxygen supply, fish size, and food availability are limiting factors that can restrict growth through several different mechanisms (Brett and Groves 1979). Although freshwater inflows can affect Texas estuaries and their living resources through many factors, perhaps the most direct and most apparent effects occur as a result of changes associated with bay salinity conditions.

Texas Study Results

Metabolic cost of ionic and osmotic regulation. The metabolic cost of regulating internal body salts (ion-osmoregulation) in the estuarine-dependent fishes of Texas has been studied by measuring fish activity levels and metabolic rates under varying salinity conditions (Kloth and Wohlschlag 1972; Cech and Wohlschlag 1975; Wohlschlag 1976; Wohlschlag et al. 1977; Wohlschlag and Wakeman 1978; Wohlschlag et al. 1980a, 1980b). A common feature of these studies was use of the fish's metabolic activity level as a natural expression of salinity stress which can occur in Texas bays and estuaries during periods of high (flood) or low (drought) freshwater inflow.

Rates of metabolism. Three kinds of metabolic rates are considered here; the distinction between them is of paramount importance in determining the ecological relevance of the study results (Holeton 1974). First is the standard rate, the lowest maintenance rate possible for completely quiescent and fasted (unfed) fish. Second is the active rate when the fish is swimming at its maximum sustained speed. The difference by subtraction between the active and the standard rates is particularly important because it defines the fish's "scope for metabolic activity." The third metabolic rate of interest is the routine rate, an ecologically operational level that lies between the standard and maximum activity levels of the fish.

Routine metabolic rate. General observations and theoretical considerations related to the optimal swimming speed of fish would support characterization of the routine activity level as a normal foraging speed of about one body length per second (Weihs 1973; Wohlschlag and Wakeman 1978). Interestingly, the routine rate also represents about the same metabolic rate exhibited by inactive, recently fed fish; therefore, it may be considered roughly equivalent to the metabolic energy level required for digestion and assimilation of food (Wohlschlag 1976; Wohlschlag et al. 1977; Wakeman 1978). Moreover, when the routine metabolic level is reduced by environmental factors, such as salinity stress, below the point where normal foraging behavior and body maintenance can occur, it may be considered below the ecological maintenance level of the fish (Brett 1976). In general, a fish's routine rate must be about twice the standard rate to allow for normal feeding and growth (Wohlschlag 1976).

Metabolism in spotted seatrout. The standard, active, and routine metabolic rates of the spotted seatrout (*Cynoscion nebulosus*), as well as its maximum scope for metabolic activity, are all at optimum values when the fish is exposed under laboratory conditions to a salinity regime of about 20‰ (Figure 6.4.1). Since full-strength seawater has a salinity of 35‰, it appears that this economically important fish does best when marine waters and fresh waters are mixed in Texas bays in near equal volumes. The effects of salinity, temperature, sublethal red tide, and poor growth condition on the metabolic rates, the scope for activity, and the swimming speeds of the spotted seatrout are also shown in Figure 6.4.2 for comparison. From this research, it is clear that environmental factors like salinity can restrict a fish's

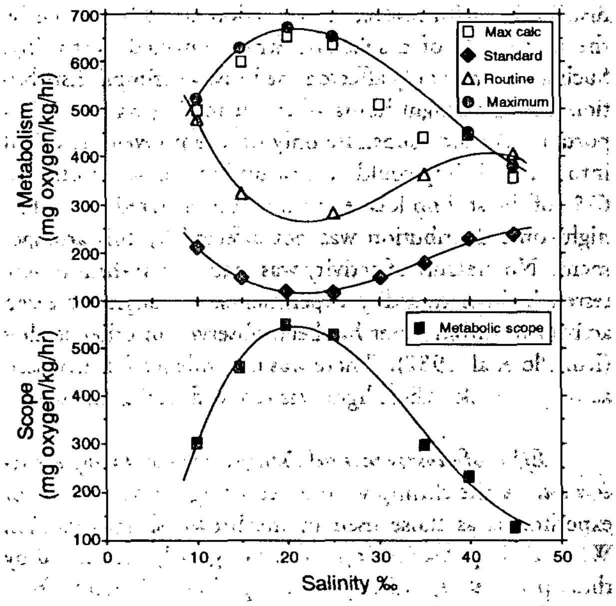


Figure 6.4.1. Upper panel—salinity stress effects on the metabolism of spotted seatrout at standard (lower line, diamonds), routine (middle line, triangles), and maximum sustained activity levels (upper line; circles are observed, squares are statistically calculated) with all fish fully acclimated to each test salinity and 28 °C temperature conditions. Lower panel—effect of salinity stress on fish scope for activity (from Wohlschlag and Wakeman 1978).

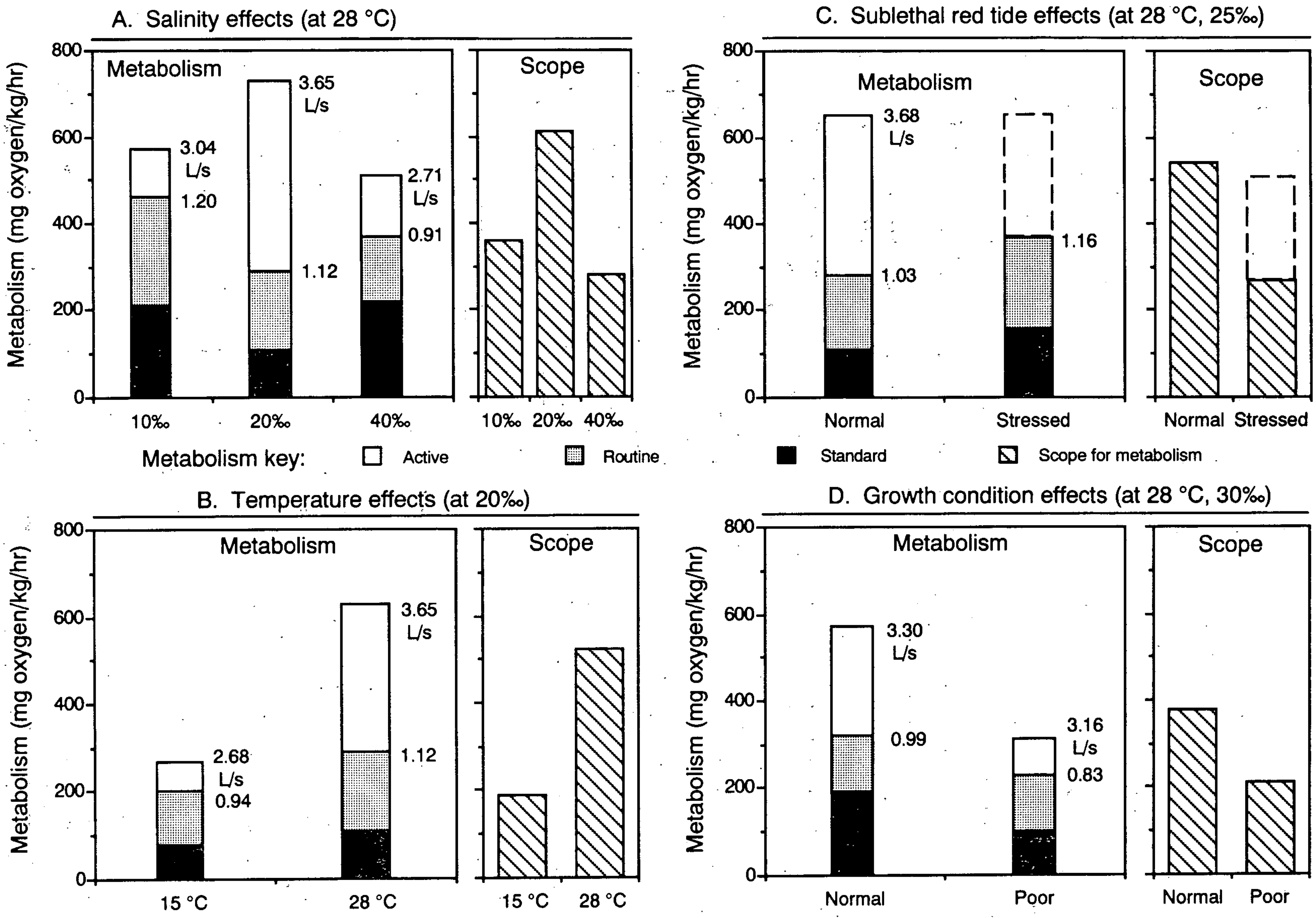


Figure 6.4.2. Panel A—Comparison of standard (lower bar), routine (middle bar), and active (upper bar) metabolic rates and swimming speeds of spotted seatrout over a range of salinities; companion graph shows the scope for activity at low, high, and optimum (20‰) salinity levels. Panel B—Temperature effects on the metabolic components under summer (28 °C) and winter (15 °C) conditions. Panel C—Effects of mild "red tide" bloom in Nueces Bay waters on the components of fish metabolism; stressed fish were not capable of "sustained" activity and stopped swimming after a few minutes; the dashed bars in the metabolism and scope graphs indicate metabolic levels had the active metabolism been the same as for normal animals; a stressed scope measurement was made, but at a lower level of activity. Panel D—Large fish in relatively poor condition compared to normal fish at near seawater (30‰) salinity (from Wohlschlag 1976).

scope for metabolic activity by either reducing the active rate, increasing the standard rate, or both (Wohlschlag 1976).

Metabolism in red drum. Similar results for the red drum or redfish (*Sciaenops ocellatus*) are given in Figure 6.4.3. Under summer temperature conditions (28 °C), metabolic rates and the maximum scope for metabolic activity are optimum at about 20‰; however, when the redfish is tested under the cooler (20 °C) conditions which typically occur in early spring and late fall, a slight shift of the optimum salinity to about 25‰ has been observed (Wohlschlag et al. 1977). The increase of the salinity optimum toward more marine conditions when temperatures are low may be a common response of many Texas coastal fish species (Wohlschlag et al. 1977). Also, since the sustained (greater than one hour) swimming speed was found to be maximum at the optimum salinity, Wohlschlag (1976) suggested that the maximum sustained speeds alone could be used to identify the optimum salinity levels of most coastal fish. At 28 °C, the maximum swimming velocity of the redfish peaks at approximately 25‰, dropping off rapidly as salinities are either increased or decreased from the optimum (Figure 6.4.4). Because the redfish, along with most other estuarine-dependent fish, is considered to be euryhaline (widely salt-tolerant), the relatively strong stenohaline (narrowly salt-tolerant) response to salinity is somewhat surprising.

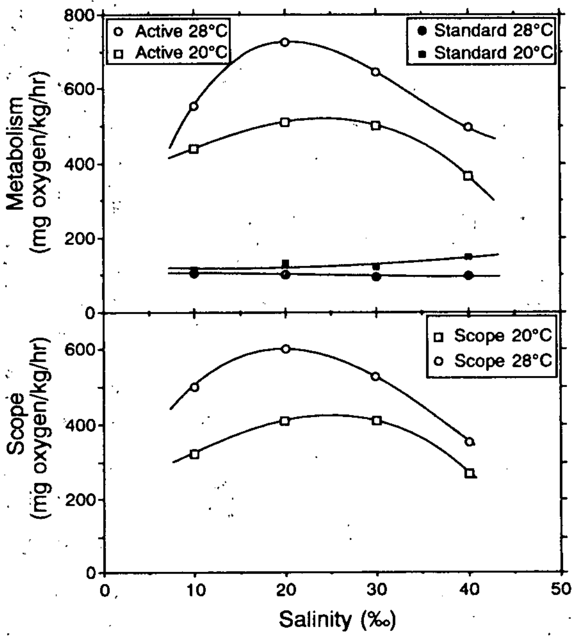


Figure 6.4.3. Upper Panel—Salinity effects on standard (lower lines) and active (upper lines) metabolic rates of red drum. Lower Panel—Effect of salinity on fish scope for activity; encircled points are observed at 28 °C, while points in squares are calculated statistically at 20 °C (from Wohlschlag et al. 1977).

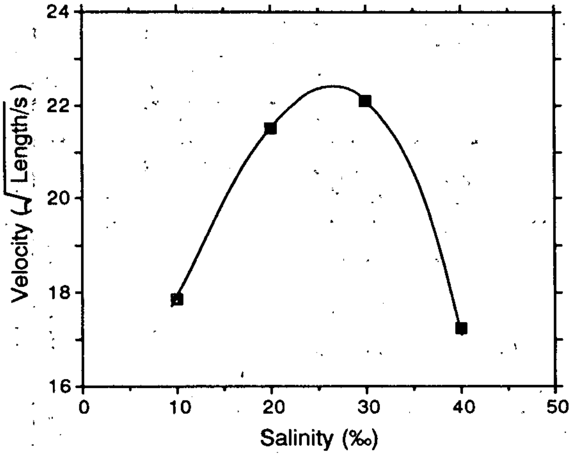


Figure 6.4.4. Effect of salinity on maximum sustained swimming speeds of red drum at 28 °C (from Wohlschlag et al. 1977).

Other estuarine fish. Using this same measure of performance, Wakeman (1978) was able to show that the maximum sustained swimming speeds of several common estuarine fish occur when salinity conditions are between 20-30‰ (Figure 6.4.5). Specifically, at 28 °C the maximum swimming speeds were recorded when salinity levels were about 20‰ for the spotted seatrout, about 25‰ for the redfish, about 25-30‰ for the sheepshead (*Archosargus probatocephalus*), and about 20‰ for the black drum (*Pogonias cromis*). In addition, the optimum salinity conditions for maximum performance of the sand seatrout (*Cynoscion arenarius*) was determined to be about 25‰, while the optimum salinity for the Atlantic croaker (*Micropogonias undulatus*) was estimated to be approximately 30‰ under cool (22 °C) temperature conditions (Wohlschlag et al. 1977).

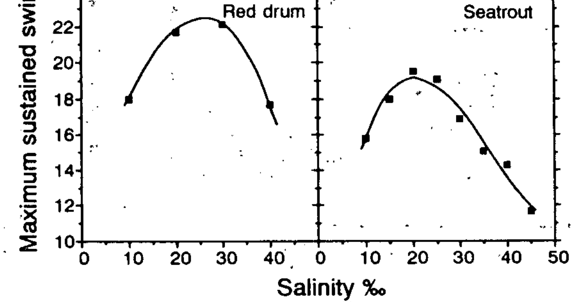


Figure 6.4.5. Comparison of salinity effects on maximum sustained swimming speeds of sheepshead, black drum, red drum, and spotted seatrout at 28 °C (from Wakeman 1978).

Effect of salinity on juvenile metabolism. Although the optimum salinity for juvenile fishes may be similar to that of the adults, there are important differences. For example, the metabolic scopes for activity in juvenile (fingerling) spotted seatrout, redfish, and Atlantic croaker are higher and broader than those exhibited by adults of the same species (Figure 6.4.6). This means that the juveniles have more energy for growth and that they are more euryhaline than adults (Wohlschlag et al. 1980a). Also, since salinity can fluctuate considerably in estuaries, the young fishes that use them as nurseries must be capable of relatively rapid acclimatization to changing salinity conditions. From experiments conducted with juvenile spotted seatrout and redfish, Wohlschlag et al. (1980b) concluded that metabolic stabilization occurs to a large extent by about 30 hours after the salinity change, and the short-term acclimatization is virtually complete by about 48 hours. In addition, it appears that rapid decreases in salinity, which can occur in Texas bays during river flooding, result in less stress on the juvenile fishes than equally rapid increases in salinity, which ordinarily occur slowly in nature (Gunter 1979). Furthermore, the smaller juveniles exhibit greater decreases in their active metabolic rates during the initial reaction phase than do larger specimens, but the time to recovery (acclimatization) is generally shorter, probably because of the relative differences in the gill surface-to-body volume ratios (Gunter 1979; Wohlschlag et al. 1980b).

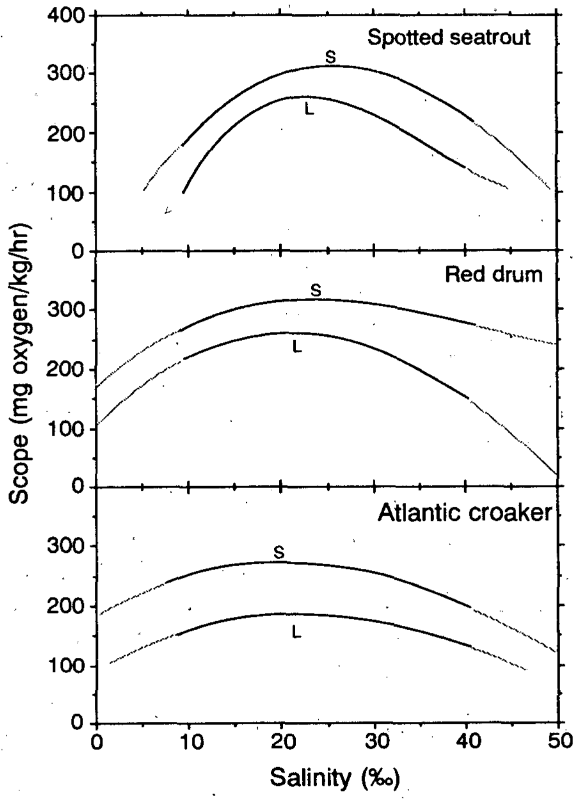


Figure 6.4.6. Comparison of salinity effects on metabolic scopes for activity of small (S) juvenile and large (L) subadult to adult spotted seatrout, red drum, and Atlantic croaker at 15 °C (from Wohlschlag 1980a).

Conclusion

Freshwater inflows can affect Texas estuarine-dependent fish in many ways, but perhaps the most direct and apparent effects occur from changes in bay salinities. Research studies on salinity effects observed in spotted seatrout, sand seatrout, red drum, black drum, sheepshead, and Atlantic croaker have demonstrated that an adult fish's scope for metabolic activity can be reduced to critical levels when salinity conditions deviate substantially from the optimum of 20 to 30‰. Although the optimum salinity for juvenile fishes is similar to that of the adults, the scope for metabolic activity is larger in the juveniles, and their optimum performance extends over a broader range of salinity conditions. These results, along with their relatively rapid (30 to 48 hours) recovery from fluctuating salinity conditions, suggests that the juveniles are specifically adapted to use Texas bays and estuaries as nursery habitats for their growth and maturation.

6.5 EFFECT OF SALINITY ON ADULT REPRODUCTION, LARVAL SURVIVAL, AND DEVELOPMENT OF FISH

Introduction

The previous section illustrates how differences in salinity affect the respiratory metabolism of estuarine boney fish (teleosts). It is clear that the salinity of estuarine water strongly influences the energy available to the organism for swimming, growth, and other metabolic functions. This section examines the effects of salinity on one of these metabolic functions, reproduction.

Background. The effects of salinity on reproduction can be divided roughly into three aspects: the effects on development of gametes in adults; the effects on eggs through hatching; and the effects on larvae after they have been released to the aquatic environment. Thomas and Boyd (1989) point out that there is almost no information on the effects of salinity on ovarian growth. Most information about the effects of salinity on reproduction have focused on physiology and development, with some work on egg and larval survival. Holliday (1969, 1971) reviewed older literature on egg and larval development, and Alderdice (1988) has reviewed more recent studies. In general, studies of egg and larval survival have shown a pattern similar to that described by Alderdice and Forrester (1968). There is a central range of salinities and temperatures over which fertilization, hatching, larval development, and survival is successful. Toward the upper and lower ends of these scales, various problems occur including egg collapse, irregular cellular development, weak larvae, and abnormalities in larval development. The Atlantic croaker (*Micropogonias undulatus*), red drum (*Sciaenops ocellatus*), and spotted seatrout (*Cynoscion nebulosus*) are among the most abundant sport and commercial fish species in Texas estuaries. These three species illustrate the range of reproductive patterns displayed by estuarine fish.

Spotted seatrout reproduction. As noted in Section 6.2, seatrout usually spend their entire life cycle within the estuary. Adults generally spawn in the bays or near the passes. Brown-Peterson et al. (1988) reported spotted seatrout spawning along the edges of seagrass beds, and Holt et al. (1990) presented evidence of seatrout spawning in the lower portion of the upper Laguna Madre far away from a Gulf pass. Spawning may occur over a long period, February through September or October, but there tend to be spring and fall spawning peaks. Eggs undergo rapid development and hatch in one to two days. Relatively little is known about larval or juvenile habits, although the latter are thought to remain in the deeper portions of the estuary during winter; both stages are frequently associated with bottom vegetation (Johnson 1978).

Atlantic croaker reproduction. Lassuy (1983d), in a review of the life history of the Atlantic croaker, noted that they do not reproduce in the estuaries like seatrout, but migrate to the Gulf to spawn offshore. Some of the ovarian development of the females may occur while croaker are resident in the estuaries. Johnson (1978) concluded that spawning of Atlantic croaker on the East Coast occurs over a wide area and extends a considerable distance offshore. He notes that spawning does not occur just around the passes, but may occur there when conditions are favorable. There is some uncertainty about the location of spawning. Bearden (1964) found ripe females 48 km (30 mi) offshore on the South Carolina coast, but Pearson (1929) and Parker (1971) suggested that spawning occurs in the Gulf near passes. Generally, adult croaker migrate in late summer and fall, and spawning occurs from October through March, with the peak period occurring in November. Spawning in the Gulf has not been directly observed, however (Lassuy 1983d). After spawning, eggs are pelagic and take less than one week to hatch. Not much is known about larval habits after hatching; Lassuy (1983d) states that larvae and post-larvae may spend some time as part of the plankton and become demersal (bottom dwelling). Many larvae return to the estuary, although small croaker are caught in offshore trawls year-round. Adult croaker reenter the bays during the spring.

Redfish reproduction. The life cycle of red drum falls between that of the Atlantic croaker and the spotted seatrout. Maturing red drum migrate to the Gulf in late summer and early fall, and spawn from mid-August through December. Their migration and spawning is earlier than the croaker's. Older adults from the Gulf return to the passes in the fall to spawn, but rarely reenter the estuaries as adults (Perret et al. 1980). Referring to several studies, Reagan (1985) noted that spawning takes place in deeper water near bay mouths and inlets, and on the Gulf side of barrier islands in some areas; Holt et al. (1985) collected eggs as far as 24 km (14.9 mi) offshore, and Lyczkowski-Shultz et al. (1988) collected larvae as far as 34 km (21.1 mi) offshore. Johnson (1978) described the spawning location to be on outer coasts near passes and channels, but not entirely confined to passes. Johnson and Funicelli (1991) reported spawning of red drum in Mosquito Lagoon, Florida, and attributed the non-oceanic spawning behavior to sea-like salinity conditions in the lagoon. Little is known about the eggs or larvae in the Gulf, although Marley (1983) reported that eggs were carried into Mobile Bay with high salinity currents. In the estuary, postlarval red drum are known to inhabit seagrass and marsh areas.

Most of the information concerning the relationship between egg or larvae and salinity for these species is descriptive and comes from studies of distribution or life history. Only indirect inferences about salinity relationships can be drawn from major life history studies involving Atlantic croaker (Hildebrand and Cable 1930; Fruge and Truesdale 1978; Lewis and Judy 1983) and spotted seatrout (Pearson 1929; Tabb 1966; Perret et al. 1980; Holt et al. 1988; McMichael and Peters 1989), largely from salinity observations when collections were made. The same sort of anecdotal information is available for the red drum (Jannke 1971; Crocker et al. 1981; Holt et al. 1983; Marley 1983; Peters and McMichael 1987; Holt et al. 1988), although Holt et al. (1981a, 1981b) conducted studies on the effects of temperature and salinity on hatching and survival of larvae for periods of up to two weeks.

From experimental studies involving some of these species, Thomas and Boyd (1989) described the effects of salinity on reproduction in adult fish. Holt and Banks (1989) and Thomas and Boyd (1989) also measured the effects of salinity on fertilization, hatching, and larval development.

Effect of Salinity on Adult Reproductive Development

Spotted seatrout. Thomas and Boyd (1989) investigated the effects of salinity on reproductive endocrine function and ovarian growth in adult spotted seatrout. Fish were captured and held at 30‰ for three weeks to acclimate to laboratory conditions. Then the salinity was altered in steps of 5‰ per day so that four salinity groups were established in holding tanks, 10, 20, 35, and 45‰. Temperature and photoperiod were chosen to reflect spring conditions, and samples were taken at the end of 30 and 60 days. The experimental design included replicate tanks for error analysis. Thomas and Boyd sampled gonadal steroids (estradiol and testosterone), made histological examinations of gonadal tissue, and used a gonadosomatic index (GSI), which is the ratio of the gonad weight to the body weight expressed as a percent, to evaluate reproductive status.

Figure 6.5.1 compares gonadosomatic indexes in successive 30-day periods for spotted seatrout. The ratio of the gonadosomatic index for animals after 30 days and the index for control animals (collected before the acclimation period) shows that initial ovarian development is sensitive to acclimation salinity. Ovarian growth was highest for animals held at 35‰, followed in order by 45, 20, and 10‰. The GSI for females held at 35‰ at the end of 30 days was significantly higher than the control GSI.

In the next 30-day period, seatrout held at 10‰ did not survive through the end of the experiment. The relative ovarian growth for fish held at 20, 35, and 45‰ was very similar, however, and was lower in the second 30-day period than in the first 30 days (Figure 6.5.1). At the end of 60 days, GSI's of both the 35 and 45‰ females were significantly higher than GSI's of females from the corresponding salinities at the end of the first 30 days.

Similar results were demonstrated by the gonadal steroid measurements, with 35‰ fish having the highest levels of both estradiol and testosterone. In relative terms, the changes in ovarian growth and gonadal steroids for the holding salinities were greater for the first 30-day period than for the second 30-day period. In addition, microscopic examination of ovaries showed that 35‰ females had a higher proportion of oocytes developing into eggs than females held at the other salinities. Thus, fecundity of 35‰ females was higher than for the other salinity groups due to greater ovarian growth and greater development of oocytes into eggs.

Thomas and Boyd concluded that ovarian growth and endocrine function in female spotted seatrout is significantly altered over the salinity range of 20 to 45‰, although the greatest salinity effect appears to be in the early period of ovarian development. Salinities of 35‰ appeared to be optimal, while lower salinities (20‰ or less) suppressed ovarian growth and caused more reproductive interference than higher salinities (45‰). They also looked at the effects of salinity on male spotted seatrout reproduction but concluded that it was not significantly changed over the 20 to 45‰ salinity range.

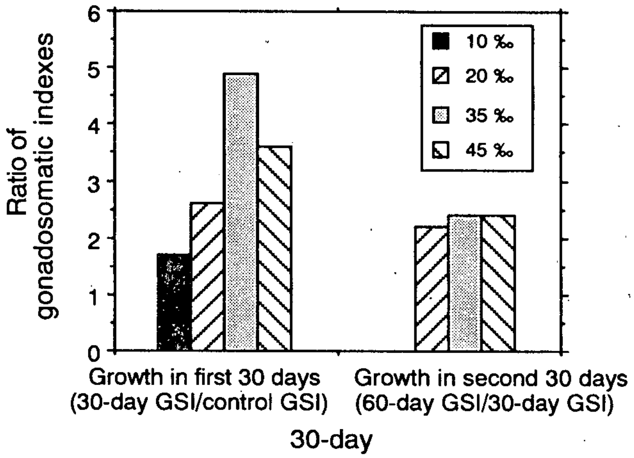


Figure 6.5.1. Ratio of the gonadosomatic index at the end of the 30-day growth period with the gonadosomatic index from the beginning of the period, showing the relative amount of growth of the ovaries compared to the body weight for spotted seatrout; the four bars on the left show the relative ovarian growth under four salinity regimes during the first 30 days of gonadal development; the three bars on the right show the relative ovarian growth during the next 30 days, before spawning; females held at 10‰ did not survive the second 30-day period (data from Thomas and Boyd 1989).

Atlantic croaker. Thomas and Boyd (1989) did similar studies with Atlantic croaker. Animals were held for three weeks at 30‰; then, salinities of five groups of animals were adjusted over a 10-day period to 5, 15, 25, 35, and 45‰. Photoperiod and temperature were selected to simulate fall conditions. The experiments, first done in 1987, were repeated in 1988; samples were taken 22 to 25 days after the holding salinities were established. Thomas and Boyd noted that they were unable to capture croaker in the earliest stages of ovarian development for a direct comparison with seatrout; in 1987, the croaker ovaries were in a more advanced stage of recrudesence (ovarian development) at the beginning of the experiment than for the spotted seatrout experiments.

In 1987, there were no significant differences among the fish held at different salinities for GSI (Figure 6.5.2) or for the gonadal steroids testosterone and estradiol. In the 1988 experiments, however, GSI was significantly lower for fish held at 25 and 45‰ than for those held at 5 and 35‰ (Figure 6.5.2). Estradiol levels and the proportion of oocytes that would develop into eggs were lower for high-holding salinities than for low-holding salinities.

Thomas and Boyd concluded that ovarian development in croaker in the last three to four weeks prior to spawning is relatively insensitive to different salinities in the 5 to 35‰ range. Holding tank salinities of 45‰ may result

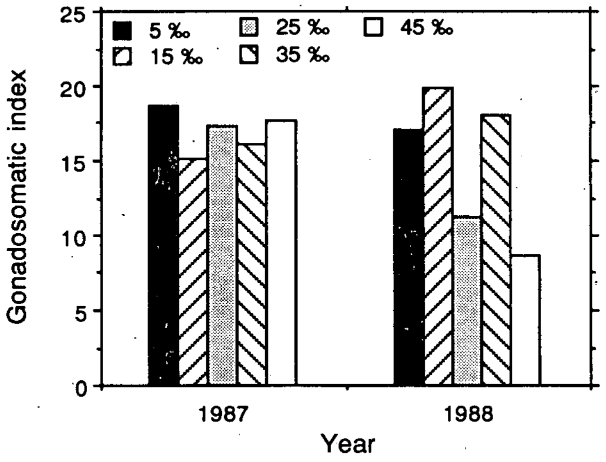


Figure 6.5.2. Gonadosomatic index of Atlantic croaker at the end of 22- to 25-day holding periods at five salinities; bars on the left were from 1987 experiments, while bars on the right were from 1988 experiments (data from Thomas and Boyd 1989).

in decreased gonadal steroid production, fecundity, and egg development, although additional experiments will be needed to clarify this point.

General effect of salinity on adult female reproduction. The GSI and gonadal steroid levels of adult female spotted seatrout differ substantially depending on salinity during the first 30 days of gonadal development, with the greatest ovarian growth occurring at 35‰ and the least growth occurring at 10‰. The second 30 days of gonadal development in the seatrout up to the point of spawning, and the 22- to 25-day period before spawning in Atlantic croaker, is relatively insensitive to the effects of holding salinity. Thus, for female fish it appears that the greatest effect of salinity on reproductive development occurs during the early stages of ovarian development, and that later stages are relatively insensitive to salinity differences.

Effect of Salinity on Egg Fertilization and Hatching

As soon as adult fish spawn, the eggs are exposed to the ambient salinity of the water. The eggs are fertilized, undergo embryological development, and hatch. Each of these developmental phases differs slightly in the manner in which the osmotic environment is regulated. A freshly spawned egg is usually subjected to a hypertonic environment, where the osmotic concentration is greater outside the egg than inside. Alderdice (1988) noted that eggs, which are relatively permeable to water and ions at spawning, go through rapid decreases in permeability during the first 24 hours after fertilization. The plasma membrane remains relatively impermeable until later stages in the embryonic development when permeability may increase again. Initially, the plasma membrane and cells of the blastula stage handle ion regulation in the developing egg. Epithelial chloride cells, particularly on the yolk sac, form shortly after the egg passes through the gastrula stage of development. They contribute to ion regulation for the embryo as development proceeds, and continue operation through hatching and into the post-hatch period. Thomas and Boyd (1989) studied the effects of salinity on fertilization and hatching of spotted seatrout and Atlantic croaker.

Spotted seatrout. Seatrout were first acclimated to laboratory conditions for three weeks at 30‰; then salinities were adjusted over a 10-day period to the test salinity levels (10, 15, 25, 35, and 50‰). The animals were held for periods of up to four months. Water temperature and photoperiod were varied to mimic local conditions from spring through summer; spawning was induced by injection with luteinizing hormone-releasing hormone. Some experiments (10, 15 and 50‰) had to be repeated because all females died before spawning.

Thomas and Boyd (1989) measured percent fertilization and hatching at the five salinities. Fertilization in spotted seatrout was high over the 10 to 35‰ salinity range, though somewhat reduced at 50‰. Fertilization at 15‰ was very low (15.8%), but Thomas and Boyd speculated this was the consequence of there being only one male in the holding tank for this experiment instead of the usual two or three. Hatching rate was also high over the 10 to 35‰ range, but much lower at 50‰. Figure 6.5.3 shows the combined effect of salinity on eggs from spawning through hatching as the product of the percent of eggs fertilized and the percent of fertile eggs hatched (at 35‰ for example, 80.8% fertilization x 80.6% hatching = 65.1% survival through hatching). Although hatching occurred over a broad range of salinities, larval abnormalities increased the more the holding salinity deviated from 35‰.

Excluding the 15‰ measurements, seatrout egg survival through hatching has a broad tolerance to salinities in the 10 to 35‰ range. Only when salinities are greater than 35‰ does survival through hatching decrease substantially. Thomas and Boyd concluded that spawning and short-term survival do not appear to be salinity-sensitive between 10 and 35‰ in spotted seatrout.

Atlantic croaker. Croaker were also acclimated to laboratory conditions and salinity was adjusted to 5, 15, 25, 35, and 45‰ over a 10-day period. Water temperature and photoperiod were manipulated to produce conditions similar to fall through early winter. Spawning was induced by hormonal injection. Fertilization and hatch rate data were collected for eggs in each treatment.

Fertilization was highest at 25 and 35‰, but was substantially lower at 15 and 45‰; fertilization was com-

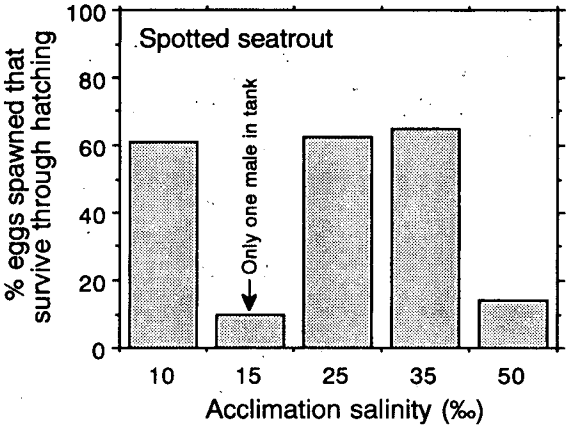


Figure 6.5.3. Percent of eggs spawned that survived through hatching for spotted seatrout at different salinities; low survival rate for 15‰ may be due to poor fertilization since only a single male was present in the tank (data from Thomas and Boyd 1989).

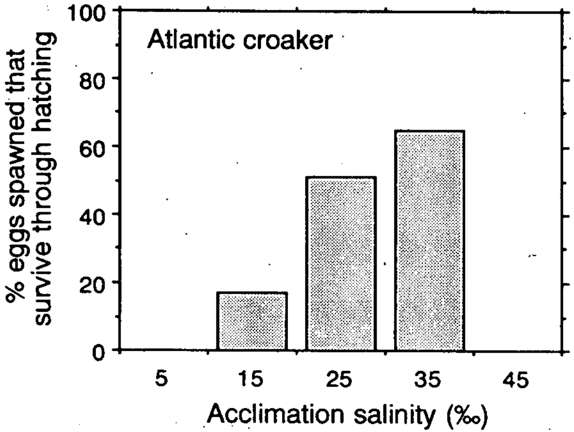


Figure 6.5.4. Percent of eggs spawned that survived through hatching for Atlantic croaker at different salinities; no eggs were fertilized at 5‰, and none of the eggs fertilized at 45‰ hatched (data from Thomas and Boyd 1989).

pletely unsuccessful at 5‰. Egg hatching was reasonably successful (60% or greater) over the 15 to 35‰ range. No eggs hatched at 5‰ since none were fertilized. Although about 9% of the eggs were fertile at 45‰, none of them hatched successfully. Combining fertilization and hatching in the same manner as for spotted seatrout, Figure 6.5.4 shows that survival through hatching was optimal at 35‰, a little less at 25‰, and substantially reduced at 15‰. Larval abnormalities were about four times as prevalent at 15‰ as at 35‰. Thomas and Boyd (1989) concluded that the salinity range for successful croaker development from egg through larva is limited to 25 to 35‰.

Red drum. Holt et al. (1981) determined that egg hatching and larval survival were optimal at 30‰. They noted that lower salinities usually resulted in lower survival. Holt and Banks (1989) measured egg hatching and 3-day development. Their methods were not directly comparable to those of Thomas and Boyd (1989) since eggs were transferred to test salinities 12 hours after they were fertilized rather than being held in constant salinities from egg development through hatching. The range of salinities over which red drum eggs hatched was broad. More than 60% of the eggs hatched at 5‰ on the low end of the salinity tolerance, and nearly 70% hatched at 60‰. The delay in transferring fertilized eggs to the test salinities apparently bypassed the most salinity-sensitive period in post-fertilization egg development.

General effect of salinity on survival through hatching. While salinity has little effect on the later stages of ovarian development in adult female estuarine fish, its effect on fertilization, development, and hatching of eggs appears to differ from one species to another and to be consistent with the fishes' life history. Survival from spawning through hatching in spotted seatrout, which remain in the estuary throughout their life cycle, was high over the 10 to 35‰ salinity range, but was substantially reduced at 50‰.

Equivalent levels of successful survival from spawning through hatching in croaker occurred only at 25 to 35‰, and was substantially reduced or zero at other salinities. Spawning, egg development, and hatching in Atlantic croaker normally occur in the Gulf at salinities around 35‰, so it is unusual for croaker eggs to experience other salinities. Thomas and Boyd's (1989) data suggest that survival from spawning through fertilization in both species is more sensitive to high and low salinities than survival through hatching after fertilization has occurred.

Experimental studies involving red drum were not directly comparable to those of seatrout and croaker since the first 12 hours after fertilization were not included in the survival test. Nevertheless, the wide salinity range over which at least 60% of the eggs hatched (5 to 60‰) probably indicates that the pattern of survival though hatching in red drum is more like seatrout than croaker. This is consistent with the spawning pattern of red drum, close to the inlets and openings of bays to the Gulf, so that eggs and larvae rapidly return to the estuary.

Effect of Salinity On Larval Development in Fish

The general osmoregulatory tendency of adult estuarine fish is to continually lose water to and gain salts from the aquatic environment in which they live. Estuarine fish have developed several physiological means to regulate their water and ionic levels (Alderdice 1988). They drink estuarine water and absorb monovalent ions and water through the intestinal wall while excreting most divalent ions; they excrete monovalent ions through the gill epithelium and both types of ions via the renal system; and they selectively take up certain ions (Na^+ and Cl^-) through the gills and renal system. When fish eggs hatch into larvae, the water-regulating capabilities of the gill epithelium, intestinal wall, and renal system are not well developed. The larvae initially depend on chloride cells in their epithelium for most of their ionic and osmotic balance until the gill s, gut, and renal organs are functional. Alderdice (1988) noted that there is little information on how ionic and osmotic regulation changes from chloride cells to the other organs in developing teleost larvae; it appears, however, that the timing of the changeover is variable. Alderdice speculated that it may be related to the life history of the species.

Holt and Banks (1989) investigated the hatch rate and survival of spotted trout, red drum, and Atlantic croaker. They used a variety of methods to ensure a supply of fertile eggs for experimental study including capturing animals migrating through the Aransas Pass and inducing them to spawn by injecting hormones, collecting ripe adults in the field and stripping the eggs and milt immediately upon capture, and culturing spawning stock in the laboratory. While Holt and Banks did a number of different types of experiments to test the effects of salinity on egg and larval development, their acute salinity tolerance tests allow the best comparisons among the three species.

Spotted seatrout. Eggs were spawned, hatched, and held at 32‰, and acute 18-hour salinity tolerance tests were conducted for larvae that were 1, 3, 5, 7, and 9 days old. Test salinities ranged from 0 to 56‰. The lethal doses (salinities) at which 50% of the larvae died (LD_{50}) were calculated for the upper and lower ends of the salinity range. Figure 6.5.5 shows the ranges determined by the experiments. The black horizontal bar (LD_{50} range) defines the salinity range over which 50% or more of the larvae survived. The gray bar encloses the LD_{50} range and shows the total salinity range where there was some larval survival. In a few cases for spotted seatrout, the total salinity range does not extend beyond the LD_{50} range. Bars that touch the vertical axes at the extreme salinities (0 or 56‰) show that some larvae survived even the most extreme salinity. If the bars do not touch the vertical axes, however, no larvae survived beyond the salinity indicated by the end of the bar.

The LD_{50} salinity range for the survival of spotted seatrout larvae is wide, varying from 2 to 6.5‰ at low salinities and 42.5 to 50‰ at high salinities, depending on larval age. An interesting point is the narrowing of the LD_{50} bar for three-day larvae. Holt and Banks noted that three days is approximately the time of first feeding by fish larvae. They suggested that the chloride cells and developing osmoregulatory organs of some larvae may not be able to regulate the mineral load taken in with food and water by larvae during initial feeding, thus increasing mortality at the upper and lower ends of the salinity scale. As the organ systems develop over the next few days, the LD_{50} bar broadens, showing improvement in larval osmoregulatory capability. Age-related differences in salinity response were not surprising and were reported in Holliday (1969) for several species of marine herring and plaice.

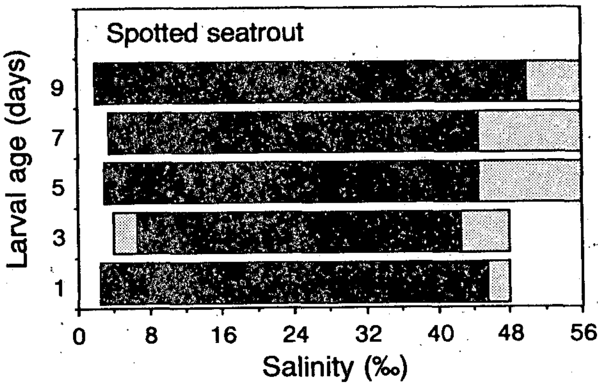


Figure 6.5.5. Survival from acute 18-hour salinity tolerance test for 1-, 3-, 5-, 7-, and 9-day-old spotted seatrout larvae that were spawned, hatched, and held at 32‰; test salinities ranged from 0 to 56‰; the black horizontal bar (LD_{50} range) defines the salinity range over which 50% or more of the larvae survived; gray areas at the end of the black bars indicate salinities for which survival was less than 50% (data from Holt and Banks 1989).

In addition to tests on larvae spawned, hatched, and held at 32‰, Holt and Banks (1989) performed acute salinity tolerance tests on larvae that had been spawned and reared at lower salinities. They compared the upper LD_{50} salinities for seatrout larvae spawned in Aransas Bay (spawning salinity 32‰) and larvae spawned in Copano Bay (spawning salinity 24‰). Larvae spawned at the higher salinity had much higher LD_{50} values for the upper limit of salinity at 1, 3, 5, and 7 days than animals spawned at the lower salinity. These results suggest that maternal acclimation salinity probably has some bearing on egg and larval survival, although differences in osmoregulatory requirements of separate fish populations cannot be ruled out.

While LD_{50} is a well-defined statistical measure for describing and comparing biological responses, it is not necessarily the most appropriate measurement for determining the salinity range that provides adequate larval recruitment and survival for population maintenance. LD_{50} deliberately excludes other factors present in the natural environment which might interact with salinity and result in tolerance differences. By combining data from several different tests, Holt and Banks (1989) defined the salinity range for larval development in which there was negligible salinity-related mortality. The range was 9 to 40‰ for spotted seatrout.

Atlantic croaker. The experimental regime for croaker was the same as for spotted seatrout except that the upper limit for testing was 50‰ and results were presented for only for 1, 3, 5, and 7 days by Holt and Banks (1989). The LD_{50} range (3.6 to 7‰ at low salinities and 33.5 to 44‰ at high salinities) was still broad, though narrower than the range for spotted seatrout (Figure 6.5.6). The total salinity range for survival was narrower, especially the 5-day range. Unlike spotted seatrout, the LD_{50} range for 3-day larvae was not narrower than for other ages. Instead, the lower LD_{50} salinity limit for 3-day larvae was elevated and the upper LD_{50} salinity limit for 5-day larvae was depressed; at this time, there is no explanation for the different pattern. Combining data from different tests, Holt and Banks defined the salinity range with negligible salinity-related mortality for Atlantic croaker to be 15 to 33‰.

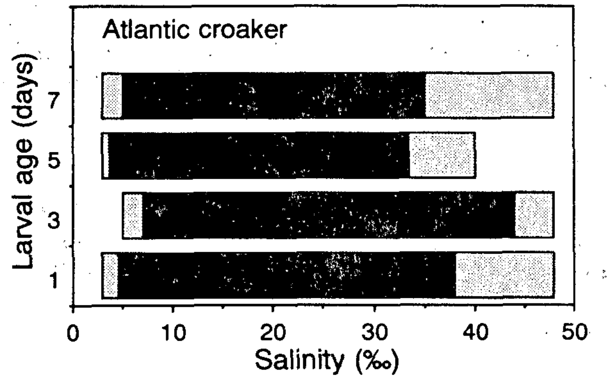


Figure 6.5.6. Survival from acute 18-hour salinity tolerance test for 1-, 3-, 5-, and 7-day-old Atlantic croaker larvae that were spawned, hatched, and held at 32‰; test salinities ranged from 0 to 50‰; the black horizontal bar (LD_{50} range) defines the salinity range over which 50% or more of the larvae survived; gray areas at the end of the black bars indicate salinities for which survival was less than 50% (data from Holt and Banks 1989).

Red drum. The same experimental regime was used for red drum as for Atlantic croaker except that results for 9-day-old drum were also presented. Figure 6.5.7 shows the LD_{50} salinity range and the total salinity range for the survival of fish larvae in the experiments. The LD_{50} range (from 4.2 to 6.5‰ at low salinities and 33.5 to 45‰ at high salinities) and the total salinity range were broad, although not as wide as for spotted seatrout; they were a little wider than for Atlantic croaker. The LD_{50} salinity range for 3-day larvae narrowed but then broadened with age, just as with spotted seatrout. Holt and Banks (1989) defined the salinity range with negligible salinity-related mortality for red drum to be 15 to 33‰.

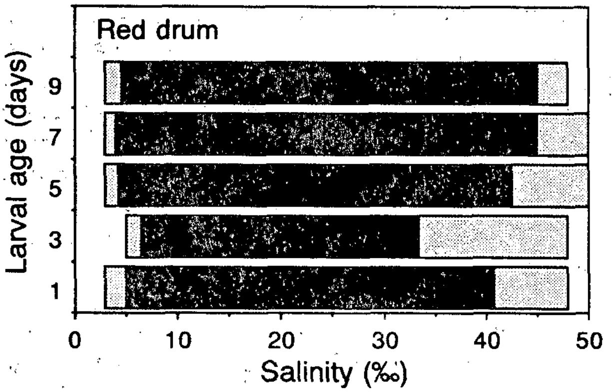


Figure 6.5.7. Survival from acute 18-hour salinity tolerance test for 1-, 3-, 5-, 7-, and 9-day-old red drum larvae that were spawned, hatched, and held at 32‰; test salinities ranged from 0 to 50‰; the black horizontal bar (LD_{50} range) defines the salinity range over which 50% or more of the larvae survived; gray areas at the end of the black bars indicate salinities for which survival was less than 50% (data from Holt and Banks 1989).

General effect of salinity on survival of larvae. Spotted seatrout, which usually live their entire lives in the estuary, have the widest range of salinities over which larval survival and development may occur with negligible short-term (18 h) salinity-related mortality (9 to 40‰). This wide salinity range allows survival of larvae over a wide area in most Texas estuaries regardless of where the larvae are transported by bay currents. Holt et al. (1990) noted that spotted seatrout reproduction apparently occurs in the lower portion of the upper Laguna Madre, far removed from access to the Gulf.

Red drum spawn in the near-shore Gulf near inlets and bay mouths. Due to the proximity of the spawning areas to the estuaries, the salinity of the water in which they spawn may be less than full-strength seawater (less than 35‰); furthermore, eggs and larvae may be swept back into the estuary by tidal currents through the inlets and passes. The range for negligible salinity-related mortality for larvae (15 to 33‰) appears reasonable and consistent with the habitat requirements for older post-larvae and juvenile red drum. These later life stages inhabit seagrass beds and marshes such as those in the lower ends of the estuaries, which have moderate to near-marine salinities. The narrower salinity range for early red drum larval survival (one to seven days), limits the area in which they may reproduce to a much greater extent than spotted seatrout.

Atlantic croaker spawn in the Gulf; they are generally thought to spawn over a wide area, not restricted to the areas near the passes. In an earlier portion of this section (concerning fertilization and hatching), salinities of 35‰ resulted in a greater proportion of Atlantic croaker eggs surviving through hatching than in 25 or 45‰ water. Since reproduction in this species is tied to Gulf waters, it seems natural to expect that there would be no salinity-related mortality at marine salinities. Therefore, it was surprising that the salinity range with negligible salinity-related mortality for Atlantic croaker was 15 to 33‰; the upper limit is a few parts per thousand below typical oceanic salinities. The measured upper LD_{50} of 5-day larvae was 33.5‰, so from the physiological experiments it appears that significant mortality may occur at normal marine salinities of 35‰. Lassuy (1983d) noted that croaker eggs were pelagic and hatched in less than a week. Lassuy also stated that larvae may remain in the plankton for some time, although it is not clear how long this period lasts. The fact that significant mortality in Atlantic croaker larvae is associated with normal marine salinities implies that croaker larvae that return to the estuaries must either encounter lower-than-normal marine salinities in the Gulf, or they must not be exposed to normal marine salinities for long periods. In the latter case, they could be spawned near the passes or transported some distance from offshore spawning grounds in a short period of time. Otherwise, very few larvae would survive the higher salinity levels of Gulf waters.

Conclusion

Spotted seatrout. Spotted seatrout represent fish whose entire life cycle is spent in the estuary. The tolerance of spotted seatrout to salinity is broad for all phases of reproduction: ovarian growth, egg fertilization and hatching, and larval development. The most sensitive aspect of reproduction was the first 30 days of ovarian growth where near-marine salinities produced the greatest ovarian growth. Thereafter, spotted seatrout reproduction seemed relatively insensitive to salinity except at very low and hypersaline levels. If freshwater inflow were to substantially decrease or increase, it does not appear that the effects on reproduction and larval development would be great as long as the species had access to higher salinity areas such as near Gulf passes.

Red drum. Red drum represent fish that go to the nearshore Gulf environment to spawn, but whose eggs and larvae quickly return to the estuary for later developmental stages. No information is available about the effects of salinity on red drum ovarian growth, and the data on survival through hatching did not include the initial 12-hour post-fertilization period which may be particularly sensitive to salinity change. Larval development without salinity-related mortality occurs over the 15 to 33‰ salinity range. As long as there is access to the Gulf through passes, it seems unlikely that the early stages of reproduction would be much affected by changes in bay salinity structure, although the effects on ovarian development have not been studied. Eggs or early larvae returning to the estuary might be affected by major changes in salinity resulting from alterations in freshwater inflow. If the salinity of the lower portion of the bay had near-marine or hypersaline conditions, larval survival might be affected since the salinity tolerance for red drum larvae is narrower than for spotted seatrout. Extreme increases in inflow could also pose a problem if lower bay salinities remained at less than 15‰ during the late fall.

Atlantic croaker. Atlantic croaker represent a third pattern of reproduction for estuarine fish, a species that migrates offshore to spawn, and whose eggs must be carried back into the estuary for further development. Ovarian development in croaker seems relatively insensitive to salinity differences, although the earliest stages of ovarian development were not investigated. The tolerance range for fertilization and hatching is fairly narrow, with best survival through hatching at 35‰, the typical marine salinity. Larval survival without salinity-related mortality occurred over the same 15 to 33‰ range as red drum. Therefore, they would experience the same sort of problems as red drum if changes in freshwater inflow altered bay salinity regimes. The return of croaker larvae to the estuary from the offshore area poses an uncertain aspect of the relationship between salinity and reproduction. Since the upper bound on salinity tolerance is below the usual level of marine salinity, successful croaker larvae must encounter lower-than-normal marine salinities, be spawned near the passes to the estuaries, or be rapidly transported from offshore back to the passes by Gulf currents.

6.6 EFFECTS OF FRESHWATER INFLOW AND SALINITY ON THE ABUNDANCE AND DISTRIBUTION OF ADULT FINFISH AND SHELLFISH

Introduction

Texas estuaries are used by many finfish and shellfish species for specific parts of their life cycles. Estuarine areas are used as paths for migration, spawning areas, nursery areas, and foraging areas. An estuary's success at fulfilling these functions can be monitored by tracking species diversity and abundance within the estuary. A decline or abrupt change in either factor over time may be a signal that environmental degradation is occurring.

In order to track species diversity and abundance within estuaries, baseline data collection programs have been established. The National Oceanic and Atmospheric Administration (NOAA) began the Estuarine Living Marine Resources (ELMR) project in 1985 to document the distribution and relative abundance of 150 fishes and invertebrates in 120 of the nation's estuaries. In Texas, NOAA has worked closely with the Coastal Fisheries Branch of the Texas Parks and Wildlife Department (TPWD) to include data from TPWD's comprehensive monitoring program. Months were spent compiling, verifying and revising data collected by TPWD since the mid-seventies in an effort to accurately characterize Texas estuaries. Later discussions in this section about the distribution by salinity zone of adult finfish and shellfish in Texas estuaries is based on these data, unless otherwise noted.

Three main selection criteria were used when considering species for inclusion in the ELMR database (Monaco et al. 1989): commercial value (species such as Gulf menhaden and shrimp), recreational value (species such as spotted seatrout and red drum), and stress indicator (species such as oysters, which bioaccumulate contaminants).

Information about relative abundance of adult (reproductively mature) finfish and shellfish for three salinity zones—tidal fresh, mixing, and seawater—in seven Texas estuaries has been included in the ELMR database. The tidal fresh zone salinities range from 0.0 to 0.5‰, the mixing zone salinities range from 0.5 to 25‰, and the seawater zone salinities are greater than 25‰. Estuaries from Mission-Aransas southward to Laguna Madre do not normally have a tidal fresh zone, and Sabine Lake does not normally have a seawater zone. A species was considered rare if it was present but not frequently encountered; common if it was generally encountered but not in large numbers; abundant if it was often encountered in substantial numbers relative to other species; and highly abundant if it numerically dominated other species (Monaco et al. 1989).

Adult Shellfish and Finfish Environmental Requirements

Eastern oyster. Reefs of the eastern oyster (*Crassostrea virginica*) are normally found in shallow bays on hard surfaces, mud flats, or sand bars (Butler 1954; Copeland and Hoese 1966; Menzel et al. 1966). In Galveston Bay, oysters have been associated with bottom types having sand contents ranging from 17 to 100% sand (Harry 1976). Adult oysters are sessile (fixed) organisms, so reef location is largely determined by distribution and survival of spat, unless seed oysters are moved to oyster leases for depuration (purging of contaminants) or to increase future harvests on the leases.

Oysters are reportedly able to withstand a broad range of salinities, but thrive at intermediate salinities. In coastal waters of the Gulf of Mexico, they can be found where salinities commonly range from 10 to 30‰ (Gunter and Geyer 1955), and in the Laguna Madre they have been observed to grow and spawn in salinities higher than 40‰ (Breuer 1962). Chanley (1957) reported that 15 to 22.5‰ was an optimum range, while Galstoff (1964) stated that 5 to 30‰ was ideal. Galstoff also said that extended time periods of salinities at less than 10‰ or greater than 34‰ were detrimental. Ray (in Mueller and Matthews 1986) theorizes that the most likely location for oyster reefs is in mid-bay regions because optimal growth and survival conditions tend to exist in the middle region. These areas of optimal growth and survival conditions are largely determined by freshwater inflows and are subject to change in size or location depending on inflow levels.

Oyster predators and parasites thrive in salinities higher than 20‰. However, oyster mortality is increased if combinations of high salinity and temperature occur, with the extent of damage directly related to the amount of time elevated salinity and temperature levels persist (Ray in Mueller and Matthews 1986).

In Louisiana, Chatry et al. (1983) reported that previous-year salinities were correlated with seed oyster production. Based on 11 years of data, they found that seed oyster production was highest when May salinities were 6 to 8‰, June-July salinities averaged 13‰, August salinities were less than 15‰, and September salinities did not average more than 20‰.

Adult eastern oysters are rarely encountered in the tidal fresh zones or in the seawater zones of Texas estuaries. Sexually mature oysters are common in the mixing zones of Sabine Lake, Matagorda Bay, Aransas Bay, and Corpus Christi Bay, and abundant in the mixing zones of Galveston Bay and San Antonio Bay.

Brown shrimp. Adult brown shrimp (*Penaeus aztecus*) are usually found in offshore waters 14 to 110 m (46 to 361 ft) deep and have an annual life cycle (Renfro and Brusher 1982). In the Gulf of Mexico, adults occupy nearshore waters where spawning and early life stages take place (Monaco et al. 1989; Perez-Farfante 1969; Williams 1984), and seem to prefer silt and muddy sand substrates (Hildebrand 1954; Williams 1984). Juvenile and adult brown shrimp feed at night on polychaetes, amphipods, chironomid larvae, detritus, and algae. Adults are selective omnivorous predators (Gulf of Mexico Fishery Management Council 1981; Zein-Eldin and Renaud 1986).

Adult brown shrimp can endure a wide range of salinities and temperatures, but actually prefer a more defined range. Large brown shrimp (greater than 7.6 cm or 3 inches) can withstand temperatures between 4 and 36 °C (39 and 97 °F), but their preferred range is between 14.9 and 31.0 °C (59 and 88 °F) (Ward et al. 1980). Adults tolerate salinities from 0.8 to 45‰, with a preferred range of 24 to 38.9‰. The combined effects of low or high salinity and temperature appear to be more deleterious than extremes of either single factor (Zein-Eldin and Renaud 1986).

Adult brown shrimp are typically not found in the tidal fresh zone of Texas estuaries except on rare occasions during May through September in Sabine Lake and during April through May in Galveston Bay. In the mixing zone of Matagorda Bay, they are common or abundant during the spring and fall, while in the mixing zone of San Antonio Bay, they are common only during May, June and July, and are rare during January through April. Adult brown shrimp are not normally present in the seawater zones of Galveston Bay, Aransas Bay, Corpus Christi Bay, or Baffin Bay. They are common in the seawater zone of Matagorda Bay during March through June and during September and October, common to abundant in San Antonio Bay during May through August, and common in Laguna Madre during March through May.

White shrimp. Adult white shrimp (*Penaeus setiferus*) are usually found in nearshore Gulf of Mexico waters less than 27 m (89 ft) deep and are often most numerous at a depth of about 14 m (46 ft) (Renfro and Brusher 1982). In offshore waters, adults have been associated with soft mud or silt substrates (Perez-Farfante 1969; Williams 1984). White shrimp are omnivorous, feeding on detritus, parts of annelids and gastropods, fish parts and bryozoans, sponges, corals, filamentous algae, and vascular plant stems and roots (Christmas and Eztold 1977).

Like brown shrimp, white shrimp can tolerate a wide range of salinities. Adults are usually found offshore in waters with salinities greater than 27‰ (Muncy 1984). Unlike brown shrimp, white shrimp are more tolerant of high temperatures and less tolerant of low temperatures (Christmas and Eztold 1977). Pullen and Trent (1969) found that emigration of juvenile and subadult white shrimp was related to sharp drops in water temperature that occurred from October to December.

White shrimp adults are not normally found in the tidal fresh zones of Texas estuaries, with the exception of Sabine Lake where they are present all year. In Sabine Lake, they are common during December through March and again during July through August, rare during April through June, and abundant during September through November.

In the mixing zones of upper coast Texas estuaries, adult white shrimp are common during January and February and again from July through December, but rare from April through June. In the mixing zone of Sabine Lake, they are highly abundant from September through November. In the mixing zone of Matagorda Bay, they are common in April, May, June, and July, but rare the rest of the year. In the mixing zone of San Antonio Bay, they are common in May through June and during November and December, and abundant to highly abundant from July through October. In the mixing zones of Aransas and Corpus Christi bays, they are common during April and May and October and November, and not present during the rest of the year.

In the seawater zone of Galveston Bay, adult white shrimp are common from October through March, but are not present the rest of the year. The opposite pattern is exhibited in the Matagorda Bay seawater zone: they are rare from August through March, but abundant to highly abundant during April through July. A similar pattern also exists for San Antonio Bay, just shifted later in the year. White shrimp are not present until May, common during May and June and November and December, and abundant from July through September. Farther down the coast at Aransas Bay, the seasonal distribution shifts back to a pattern similar to Galveston Bay: common in the fall but not present the rest of the year. In Corpus Christi Bay, adult white shrimp are common from April through June and abundant during September through November, but are not present the rest of the year. In Laguna Madre, white shrimp follow the same pattern as brown shrimp: adults are common during March through May, but are rare from December through February and are not present during the rest of the year. Adult white shrimp are rarely found in Baffin Bay.

Blue crabs. Adult blue crabs (*Callinectes sapidus*) are associated with muddy and sandy estuarine bottom habitats. Juveniles and adults have been described as scavengers, carnivores, detritivores, and omnivores, subsisting mainly on mollusks, crustaceans, and fish (Tagatz 1968; Jaworski 1972).

Adult males are found in low salinity waters, where salinities are less than 10‰, while females move to these lower salinity areas to mate and return to higher salinity areas afterward. Adult and juvenile blue crabs can survive temperatures from 3 to 37 °C (37 to 99 °F), but only at the extremes of this range if they are acclimated and salinities are favorable (Tagatz 1969). Blue crabs are less tolerant of low salinities if temperatures are high and less tolerant of high salinities if temperatures are low (McKenzie 1970). Adults do not feed as temperatures approach 10 °C (50 °F), and burrow in mud or seek deeper waters when temperatures reach 5 °C (41 °F). Adult males are more tolerant than females of extreme high or low temperatures. Adult blue crabs are found in all salinity zones of all Texas estuaries.

Gulf menhaden. Gulf menhaden (*Brevoortia patronus*) seem to prefer areas near soft, muddy substrates because of the associated benthic populations and organic content (Christmas et al. 1982). Juvenile and adult Gulf menhaden are omnivorous filter feeders, sustained by phytoplankton, zooplankton, and detritus (Govoni et al. 1983; Guillory et al. 1983; Deegan 1986).

Gulf menhaden prefer water temperatures between 12 and 30 °C (54 and 86 °F), but they can tolerate temperatures between 2.5 and 35.5 °C (4.5 and 95 °F), even at lower salinity levels. Temperatures higher than 30 °C (102 °F) are reportedly avoided, and fish kills occurring at 39 °C (102 °F) have been documented (Holcomb 1970; Copeland and Bechtel 1971; Gallaway and Strawn 1974; Christmas and

Waller 1975). Gulf menhaden have been collected in fresh to hypersaline waters. Gravid adults are normally associated with the higher salinities of the open Gulf of Mexico ranging from 30‰ and higher. Non-gravid and developing adults are associated with mid-range salinities in the deeper part of estuaries. Fish kills have been reported at salinities of 80‰ or greater (Springer and Woodburn 1960; Holcomb 1970; Tagatz and Wilkens 1973; Gallaway and Strawn 1974; Shaw et al. 1985).

Adults leave the estuaries for the Gulf to overwinter and spawn from October through January; they return again starting in February (Holcomb 1970; Roithmayr and Waller 1963; Tagatz and Wilkens 1973). A correlation between lipid content and time of migration has been documented: fish with high lipid content migrate earlier, in response to temperature changes, than do fish with low lipid content (Deegan 1986).

Gulf menhaden are not found in any salinity zones in Galveston Bay or Aransas Bay or in the tidal fresh zone of Matagorda Bay. They are common only during September through November in Sabine Lake and Corpus Christi Bay, and during July through October in Laguna Madre. They are common in the mixing and seawater zones of San Antonio Bay from March through October. In Matagorda Bay, they are abundant in the mixing zone during April through October and highly abundant in the seawater zone during March through October.

Pinfish. Pinfish (*Lagodon rhomboides*) are found in a wide range of habitats, but are most abundant over vegetated shallow areas. Adult pinfish are thought to be omnivorous, feeding partially on seagrasses (Stoner 1980), and are an important forage fish for other species such as red drum and spotted seatrout.

Pinfish have been reported from waters having salinities ranging from 2.1 to 37.2‰ along the Texas coast (Gunter 1945). Cameron (1969a) found pinfish at salinities higher than 70‰ in Baffin Bay and reported that at low salinities following heavy rains, pinfish abundance in seagrass beds declined. Other studies (Gunter 1945; Kilby 1955; Weinstein 1979) have shown that salinity is not as strong a factor as vegetation in determining pinfish distribution. Pinfish can tolerate water temperatures between 10 and 35 °C (50 and 95 °F).

Most of the pinfish life cycle is carried out offshore. Adults spawn offshore and larvae are then transported to areas near passes. Juvenile pinfish enter the estuaries where they remain until the fall spawning period, at which time subadults and adults migrate back to offshore spawning areas.

Pinfish are not normally found in Sabine Lake or Galveston Bay, and are rare in the mixing zone of San Antonio Bay. However, they are common in the seawater zone of San Antonio Bay all year and in Matagorda Bay during all months except December and January. They are highly abundant in both the mixing and seawater zones of Aransas Bay, Corpus Christi Bay, and Laguna Madre.

Spotted seatrout. Adult spotted seatrout (*Cynoscion nebulosus*) are most abundant near seagrass beds but can also be found over shallow muddy areas, man-made structures, and shell reefs (Benson 1982). Spotted seatrout are opportunistic carnivores. Young adults feed on a combination of invertebrates and fish, while large adults feed almost exclusively on fish (Darnell 1958; Seagle 1969). In warmer months, young adults tend to consume more shrimp than in colder months, due to shrimp availability (Pearson 1929; Gunter 1945).

Spotted seatrout can tolerate salinities between 0.2 and 77‰ (Simmons 1957) but seem to prefer salinities between 20 and 35‰ (Loman 1978). Wakeman and Wohlschlag (1977) found that maximum sustained swimming speeds for individuals between 174 and 438 mm (6.8 and 17.2 inches) occurred between 20 and 25‰, and that speeds were reduced above 25‰ or below 20‰. Adult spotted seatrout prefer temperatures between 15 and 27 °C (59 and 81 °F). Temperatures for spawning range between 20 and 30 °C (68 and 86 °F) (Benson 1982).

Spotted seatrout also leave the estuaries for offshore waters to spawn. Larvae are carried back to areas near passes and reenter the bays when they become juveniles. They remain in the bay until sexually mature and then migrate to spawning areas as subadults.

Adult spotted seatrout are rarely found in the tidal fresh zone of Texas estuaries or in the mixing zone of Sabine Lake, but are common in the mixing zone of the other estuaries and in the seawater zone of all bays.

Red drum. Within bays, adult red drum (*Sciaenops ocellatus*) are found over muddy to sandy bottoms or oyster reefs devoid of seagrass (Yokel 1966; Perret et al. 1980). After their first spawning, adults are usually found in shallow Gulf of Mexico waters off the coast (Pearson 1929; Simmons and Hoese 1959; Perret et al. 1980; Ward et al. 1980; Pafford 1981; Benson 1982; Overstreet 1983; Ross et al. 1983), or in waters 40 to 70 m (131 to 230 ft) deep (Overstreet 1983) as far as 19 km (11.8 mi) offshore (Simmons and Breuer 1962). Adult red drum are predators, feeding for the most part on fish, shrimp, and crabs (Simmons and Breuer 1962; Boothby and Avault 1971). During the winter and spring, fish—especially menhaden and anchovies—are the most common food choice among adults. In the late spring, summer, and fall, this choice shifts to crabs and shrimp.

Juvenile and adult red drum can tolerate salinities in the range of 0 to 45‰, but are rarely found at 50‰ or above. (Gunter 1945; Simmons 1957; Simmons and Breuer 1962; Yokel 1966; Franks 1970; Perret et al. 1971; Juneau 1975; Tarver and Savoie 1976; Bonin 1977; Barret et al. 1978; Perret et al. 1980; Ward et al. 1980; Crocker et al. 1981; Holt et al. 1981; Daniels and Robinson 1986; Peters and McMichael 1987). Salinities in the range of 20 to 40‰ are preferred by large juveniles and adults, with maximum growth for juveniles occurring at 35‰ (Bonin 1977; Perret et al. 1980; Ward et al. 1980; Crocker et al. 1981; Holt et al. 1981a; Benson 1982). Red drum are able to withstand sudden salinity changes since they are efficient osmoregulators. Adults appear to be less tolerant of low salinity estuarine waters than juveniles, but are more tolerant of high salinities found offshore (Yokel 1966; Crocker et al. 1983).

Adult red drum have been collected over a temperature range of 2 to 33 °C (36 to 91 °F) (Simmons and Breuer 1962; Yokel 1966; Juneau 1975; Perret et al. 1980; Ward et al. 1980; Daniels and Robinson 1986). Adults are considered more prone to the effects of sudden cold spells than smaller fish (Yokel 1966), and are known to seek refuge in warmer, deeper waters (Simmons and Breuer 1962).

Red drum have life cycles very similar to spotted seatrout. They also leave the estuaries for offshore waters to spawn. Larvae are carried back to areas near passes, and juveniles reenter the bays where they remain until sexually mature. At this stage, the sexually mature subadults and adults migrate to spawning areas in the gulf.

Adult red drum, like spotted seatrout, are also rarely found in the tidal fresh zone of any Texas estuaries. Unlike spotted seatrout, adult red drum are also rare in both the mixing zone and the seawater zone of any of the estuaries except for Galveston and Matagorda bays. Then they are common during September through November in the mixing and seawater zones of Galveston Bay and also during August through November in the seawater zone of Matagorda Bay.

Striped mullet. Striped mullet (*Mugil cephalus*) are found in shallow, nearshore waters such as open beaches, flats, lagoons, bays, salt marshes, and grass beds, especially turtlegrass (*Thalassia testudinum*) and widgeongrass (*Ruppia maritima*) (Gunter 1945; Kilby 1949; Breuer 1957; Renfro 1960; Hellier 1962; Zimmerman 1969; Franks 1970; Perret et al. 1971; Swingle 1971; Christmas and Waller 1973; Moore 1974; Bishop and Miglarese 1978; Henley and Rauschuber 1981; Cech and Wohlschlag 1982). Striped mullet are associated with softer, organic-rich substrates such as mud and sand which contain detritus, but are also found over finely ground shell, clay, mud and sand mixtures, silt, and silt-clay mixtures (Kilby 1949; Breuer 1957; Tabb and Manning 1961; Franks 1970; Swingle 1971; Ward et al. 1980; Cornelius 1984).

Juvenile and adult striped mullet can acclimate to a wide range of salinities and temperatures. They have been collected in waters with temperatures ranging from 5.9 to 37.0 °C (43 to 99 °F), but their ability to withstand short periods in 40 °C (104 °F) waters is limited (Gunter 1945; Kilby 1949; Hellier 1962; Franks 1970; Perret et al. 1971; Swingle 1971; Dunham 1972; Moore 1974; Tarver and Savoie 1976; Ward et al. 1980). Adult striped mullet prefer temperatures of 16 to 30 °C (61 to 86 °F) (Ward and Armstrong 1980). Striped mullet are able to tolerate salinities from 0.0 to 75‰, but prefer salinities of approximately 26‰ for adults and 20 to 28‰ for juveniles (Gunter 1945; Kilby 1949; Simmons 1957; Hoese 1960; Renfro 1960; Hellier 1962; Perret et al. 1971; Dunham 1972; Christmas and Waller 1973; Swingle and Bland 1974; Tarver and Savoie 1976; Finucane et al. 1978; Ward et al. 1980; Cornelius 1984).

Striped mullet have life cycles similar to red drum, leaving estuarine areas during the fall and winter to spawning areas in the Gulf. Juveniles migrate back to estuarine areas to mature.

Unlike red drum, adult striped mullet are commonly found in all salinity zones of all bay systems year-round. In both the mixing and seawater zones of Matagorda Bay, striped mullet are abundant year-round.

Freshwater Inflow Effects

Oysters. Oysters are affected by freshwater inflows in various ways. Changes in salinity patterns due to severe droughts, chronic inflow reductions, and channelization within bays causes relocation of oyster reefs to upper bay areas (Ray in Mueller and Matthews 1987). In these regions, oyster reefs become much more susceptible to freshwater kills, runoff pollution, and siltation due to their proximity to the input source. Mid- and lower-bay reefs are somewhat less susceptible to these threats since the additional volume of bay water between the reefs and the input source acts to dilute the detrimental effects associated with high freshwater inflows.

Childress et al. (1975) reported that during the 1962 to 1964 drought, the gaged freshwater inflow to San Antonio Bay dropped to as low as 0.6 million acre-ft/yr, and the oyster harvest was approximately 250,000 pounds/year. In contrast, during 1971 to 1973, the mean annual gaged inflow to the bay was more than 2.8 million acre-ft/yr and the annual oyster production averaged 309,000 pounds, even though in May 1972, a killing flood of 1.13 million acre-ft destroyed most of the oysters north of the Intracoastal Waterway.

One method proposed for minimizing freshwater oyster kills called for impoundment of freshwater inflows in upstream reservoirs during the late spring or fall when most floods occur, with gradual releases over the summer months. However, maintaining lowered salinity levels is the only effective method for controlling the spread of the southern oyster drill (*Thais haemastoma*) (Butler 1954; Galtsoff 1964; Breithaupt and Dugas 1979) and *Perkinsus marinus*, a protozoan pathogen. Fresh water would have to be released to limit the maximum salinity to the 10 to 15‰ range in the summer and early fall months to control the southern oyster drill and *Perkinsus* populations.

However, due to competition for fresh water that may be in limited supply during dry summer months or periods of drought, this may be easier said than done (Ray in Mueller and Matthews 1987). In addition, low salinity alone will only control the spread, but probably not eradicate, oyster pests. Hofstetter (1977) theorized that floods severe enough to kill oyster populations in Galveston Bay may be necessary to provide a natural *Perkinsus* control since the parasite's host is periodically reduced in number. Childress et al. (1975) reported that following floods that occurred during 1960 to 1961 and 1965, San Antonio Bay commercial oyster harvests increased for several years.

Perhaps some sort of dynamic freshwater inflow equilibrium must be reached to optimize oyster production. Boynton (1975) proposed that fluctuating salinities rather than some optimum, but constant, salinity would provide the best conditions for oysters. Menzel et al. (1966) pointed out that oyster predators and parasites thrive at both high and low salinities (for example, oyster drills at high salinities and blue crabs at low salinities). Browder and Moore (1981) feel that oscillating salinity levels should inhibit oyster predation and parasitism at either extreme. The gaged freshwater inflow to San Antonio Bay averaged 2.1 million acre-ft/yr, ranging from 1.3 to 3.1 million acre-ft/yr, for the four best years of oyster production—1967, 1970, 1971, and 1972 (Childress et al. 1975).

Brown shrimp. In Louisiana, Barret and Gillespie (1973) found that from 1967 to 1972, brown shrimp yields were inversely proportional to spring freshwater inflows and rainfall. Later, Barrett and Ralph (1976) found that brown shrimp yields were directly related to the number of acres of estuarine surface waters with more than 10‰ salinity during the spring.

Two of the best brown shrimp production years in San Antonio Bay, 1967 and 1973, coincided with two years of high freshwater inflow (Childress et al. 1975). The brown shrimp harvest for 1967 was 205,000 kg (453,000 lb) and the gaged freshwater inflow was 3.15 million acre-ft. The brown shrimp harvest for 1973 was 299,000 kg (659,000 lb) and the gaged freshwater inflow was 4.85 million acre-ft. Childress et al. (1975) theorized that although brown shrimp catches decrease with salinity, total species production appeared to increase with freshwater inflow. Perhaps elevated freshwater inflows cause the shrimp to move into areas where they are more likely to be caught by shrimpers. They also noted that annual landings data from 1965 to 1973 showed that brown shrimp yields were at least 90,703 kg (200,000 lb) when annual freshwater inflows exceeded 2 million acre-ft.

White shrimp. White shrimp abundance has been directly correlated with freshwater inflow and rainfall in Texas (Hildebrand and Gunter 1953; Gunter and Hildebrand 1954). Childress et al. (1975) reported that white shrimp abundance in San Antonio Bay was positively related to river flow. Years of low white shrimp production coincided with years of low gaged inflow; years in which white shrimp production exceeded 355,556 kg (800,000 lb) coincided with years of gaged inflows in excess of 1.6 million acre-ft. They also noted that during the three best years of high white shrimp production during their study, the May-June inflow accounted for more than 33% of the annual gaged freshwater inflow.

Blue crab. In Apalachicola Bay, Louisiana, Meeter et al. (1979) reported that annual blue crab harvests from 1957 to 1977 were positively correlated with annual river flow. In San Antonio Bay, Childress et al. (1975) found that young blue crabs were most prevalent in trawl samples from December through March, but that adults were scarce due to their ability to evade the trawls. They also noted that during the three years of highest commercial blue crab catch (1969, 1972, and 1973), the average gaged freshwater inflow to San Antonio Bay was just more than 3 million acre-ft/yr.

Gulf menhaden. Childress et al. (1975) reported that gulf menhaden were more plentiful in wetter years, but during the last two years of the study, fluctuations in catch appeared to be negatively related to inflow. Gulf menhaden catch was usually inversely proportional to the monthly inflows, indicating that gulf menhaden were able to leave the bay when salinity levels dropped, and return when salinity levels increased in order to feed on plankton.

Wetzel and Armstrong (1987) showed that adult gulf menhaden, relative to other species they studied, had a strong negative response to increased freshwater inflow in Tres Palacios Bay. The curve relating population to freshwater inflow in the riverward section of Tres Palacios Bay peaked at 35 cfs.

Pinfish. Cameron (1969b) found that pinfish abundance dropped off in the shallow flats of Redfish Bay, Texas after heavy rainfalls that lowered salinities to 4‰. Wang and Raney (1971) asserted that juvenile pinfish were more common in low salinity areas in Charlotte Harbor, Florida, than adult pinfish. However, in Apalachee Bay, Florida, Stoner and Livingston (1984) noted that pinfish were most abundant near sources of freshwater inflows. This does not appear to be the case in the Guadalupe Estuary.

Spotted seatrout. In Matagorda Bay, adult spotted seatrout did not exhibit a strong response to freshwater inflow (Wetzel and Armstrong 1987). Pearson (1929) suggested that spotted seatrout were attracted to less turbid areas. Tabb and Manning (1961) hypothesized that high turbidities associated with Hurricane Donna caused spotted seatrout to suffocate because their gills were clogged with suspended solids.

Red drum. Little is known about the effects of freshwater inflow on adult redfish abundance. Childress et al. (1975) felt that commercial finfish landings were insufficient for recommending freshwater inflows. A positive relationship was noted between 1970 red drum commercial landings and September 1967 floods, but a relationship on which to base yearly recommendations could not be supported. In Matagorda Bay, red drum adults showed a slight preference for low flow in riverward sections of the bay and high flow in bayward sections of the bay, but again, data was spotty (Wetzel and Armstrong 1987).

Striped mullet. In Georgia, young striped mullet were the only species that responded actively to high freshwater inflows. During periods of high inflow, mullet nearly disappeared from upper- and middle-bay stations, but then reappeared at upriver locations by summer (Rogers et al. 1984). However, in Tres Palacios Bay, Texas, adult striped mullet showed a strong preference for freshwater inflow (Wetzel and Armstrong, 1987).

Conclusions

The adult finfish and shellfish species discussed in this section do not appear to be as sensitive to salinity levels or freshwater inflows as their juvenile counterparts. This is particularly true of those organisms that spend their adult phases in the Gulf of Mexico. Two exceptions are oysters and blue crabs.

Even though oysters can be found in salinities ranging from 10 to 30‰, the optimum range is probably between 15 and 22.5‰. In Texas estuaries, adult oysters are most often found in 0.5 to 25‰ waters. Mature oysters are particularly sensitive to freshwater inflows since they are sessile organisms and cannot migrate in response to environmental conditions such as extreme high or low salinity. In addition, oysters depend on freshwater inflows to offset predation and parasitism.

Adult male blue crabs prefer salinities of less than 10‰, while adult female blue crabs prefer higher salinities but will migrate to lower salinity waters to mate. Adult blue crabs are found in all salinity zones of Texas estuaries. In Louisiana and Texas, blue crab harvests have been positively linked to freshwater inflows.

Adult brown and white shrimp, normally found in offshore Gulf waters but present in estuarine waters prior to migration, are not especially dependent on freshwater inflows for their salinity requirements. Both species can withstand a wide range of salinities but prefer higher salinities, i.e., higher than 24‰ for brown shrimp and higher than 27‰ for white shrimp. In Texas estuaries, adult brown and white shrimp are most often found in either the mixing zone (0.5 to 25‰) or in the seawater zone (greater than 25‰). Both brown and white shrimp harvests appear to benefit from freshwater inflows, especially in the spring. Like oysters, however, too much freshwater inflow can be detrimental.

Adult pinfish, spotted seatrout, and red drum are also normally found in offshore Gulf waters, but are present in estuarine areas prior to migration. Salinity does not appear to be as important an environmental factor for influencing the distribution of these species as vegetation. Adult pinfish and spotted seatrout are found in the mixing and seawater zones of all estuaries, but adult red drum appear to be restricted to the mixing and seawater zones of Galveston and Matagorda Bays. In Florida, pinfish have been associated with freshwater inflow sources, but this has not been documented for Texas.

Adult mullet can acclimate to a wide range of salinities, but prefer approximately 26‰ waters. They are found throughout all bay systems in Texas year-round, where they appear to respond positively to freshwater inflows.

6.7 DIFFERENCES IN THE RELATIVE ABUNDANCE OF SELECTED FINFISH AND SHELLFISH AMONG TEXAS ESTUARIES.

Introduction

In earlier sections of this report, it has been demonstrated that Texas estuaries differ greatly with respect to the total quantity of fresh water received from tributaries emptying into their basins. Sabine Lake is the least saline and the upper Laguna Madre is the most saline (Figure 4.1.16 and Figure 4.1.17; Table 6.7.1). These differences in salinity, freshwater inflows, basin volumes (Figure 4.1.5), evaporation rates, and unique topography result in different habitat types or different proportions of similar habitats among Texas estuaries. Much of the variation in finfish and shellfish densities among estuaries may be attributed to differences in salinity regimes and habitat characteristics.

The present study completed a series of statistical analyses to determine the relative densities (catch rates) of several economically or ecologically important animal species among the estuaries and assess the correspondence between densities and salinity regimes in each estuary. The species selected for these analyses included blue crab (*Callinectes sapidus*), brown shrimp (*Penaeus aztecus*), white shrimp (*Penaeus setiferus*), southern flounder (*Paralichthys lethostigma*), Atlantic croaker (*Micropogonias undulatus*), black drum (*Pogonias cromis*), Gulf menhaden (*Brevoortia patronus*), pinfish (*Lagodon rhomboides*), red drum (*Sciaenops ocellatus*), spotted seatrout (*Cynoscion nebulosus*), and striped mullet (*Mugil cephalus*). In addition, eastern oyster (*Crassostrea virginica*) needs for lower salinities are summarized. No statistical analyses were completed for this species, however, because the availability of oyster dredge data was limited to a two-year period and the salinity preferences and tolerances of eastern oysters are well documented (Quast et al. 1988a).

Sampling Methods

Texas Parks and Wildlife Department Coastal Fishery Monitoring Program monitoring data (TPWD-CF)was used to determine differences in catch rates (i.e., relative densities) for selected species among the estuarine systems. This data was used because the sampling procedures are standardized and well documented (Meador et al. 1988; TPWD 1990a). It also provides more reliable density estimates than landings or harvest data which are influenced by market activity and unreliable self reporting (Green and Thompson 1981). Trawl (1983 to 1988) and gill net (1977 to 1989) data used in this study came from seven major Texas estuaries extending from the Trinity-San Jacinto Estuary southward to the lower Laguna Madre.

Table 6.7.1. Mean salinities from the Texas Parks and Wildlife Department Coastal Fisheries Monitoring Program taken with gill net (1977-1989) and trawl net (1983-1988) samples. Means within the same column having the same letter were not statistically different based on an a posteriori test (Hochberg's GT2 method in SAS); the minimum significant difference in trawls and gill nets was 1.04 and 0.876, respectively.

	Gear Type	
Estuary	Gill net	Trawl
Trinity-San Jacinto	17.5 D	14.8 G
Lavaca-Colorado	20.2 C	24.2 D
Guadalupe	20.8 C	17.4 F
Mission-Aransas	19.9 C	21.3 E
Nueces	30.4 B	31.3 C
Upper Laguna Madre	38.5 A	38.8 A
Lower Laguna Madre	31.9 B	33.5 B

Trawls

Trawl sampling was stratified. Each estuary was divided into two zones, one close to the river mouth, the other adjacent to the Gulf, except for the Laguna Madre which was divided into an upper (North) and lower (South) zone. The Laguna Madre was divided into these two regions because the upper zone has a different salinity regime from the lower zone (since it does not have a direct connection with the Gulf of Mexico), and the upper zone does not have significant riverine freshwater inflow. Sampling sites in each zone were randomly chosen from bay grids which were demarcated into areas approximately 1 minute of latitude by 1 minute of longitude (approximately 1 mi. sq.). During each month, 10 samples from each zone were collected in each bay. Trawls at each station were pulled in a circular pattern near the center of the grid for 10 or 15 minutes. Trawls were 6.1 m wide (20 ft) at the mouth and had 3.8 cm (1.5 inch) mesh throughout.

Gill Nets

Prior to the fall of 1981, gill net samples were collected each month at randomly selected locations from a set of about 100 fixed sites in each bay system. During the fall of 1981, the gill net program was changed to incorporate random selection from the same one-mile grids used for trawls. Random selections were made with the condition that the shoreline had to occur in the selected grid. The program was also changed so sampling occurred only during fall (mid-September through mid-November) and spring (mid-April through mid-June) each year. Sample sizes were increased from 6 samples per month to 45 per season per estuary. All gill net samples were accomplished with a net having four 48-m (150-ft) sections of 76-, 102-, 127-, and 152-mm (3-, 4-, 5-, and 6- inch) mesh, tied together in this sequence. The nets were always set with the 76-mm mesh nearest shore and the 152-mm mesh farthest offshore. The net was always set perpendicular to the shore at least a half-hour before sunset and picked up as close to sunrise as possible on the following day (generally within four hours).

Analyses

Differences in catch rates (densities) for selected species among the seven estuarine systems were determined using an analysis of variance (ANOVA) based on the following model:

$$Y_{ijk} = a + b_i x_i + b_j x_j + b_k x_k + e_{ijk}$$

where Y_{ijk} = mean catch

a = y intercept for $x_i = 0$, $x_j = 0$, and $x_k = 0$

b = regression coefficient

x_i = estuary (1 to 7)

x_j = year, 1983 to 1988 (trawls), or 1977 to 1989 (gill nets)

x_k = month (1 to 12 for trawls), or season (spring or fall for gill nets)

e_{ijk} = error

Procedure GLM from the SAS statistical analysis system (SAS Inst. 1988) was used to perform ANOVA on trawl and gill net catch data. All gill net analyses were accomplished using both spring and fall data without exception. Trawl analyses, however, were accomplished in a stepwise, interactive manner. Data was manipulated on a species-by-species basis to achieve the best fit of the model, and the best groupings of the mean catches among estuaries. If the model revealed statistically significant month effects, post hoc comparisons among mean catches were made on the effects of months. Only months in which mean catches were significantly higher than the lowest group of non-significantly different means were retained for the final analysis. This was done to keep from diluting the comparisons with time periods in which the species were not commonly found in the estuary, or were of a size not effectively caught by the gear. Overall goodness-of-fit was evaluated by the portion of the variability attributed to the model (R^2), while estuary main effects were tested by the Hochberg's GT2 method in SAS ($P < 0.05$).

All data was converted to a catch per/effort unit (catch/0.25 h for trawls, and catch/10 h for gill nets). All data was screened for outliers before subjecting them to analysis. Analyses with log-transformed and untransformed data were performed. If log-transformed data resulted in a significant increase in R^2 or a decrease in the coefficient of variation, transformed data were used. Otherwise, the original data was used. Since sample sizes were not equal, the partial sum of squares (Type III SS in SAS) was used to interpret results. All factors were assumed to be treatments, and all tests for significance were performed with the error mean square.

Sample size for all gill net analyses was 6,375. Sample sizes for the trawl analyses varied from 3,604 to 8,648 (Table 6.7.2). All trawl analyses were completed with log-transformed data and all gill net analyses except spotted seatrout were accomplished with log-transformed data. Mean salinities computed for each estuary in Table 6.7.1 used data collected at the same time a biological sample was taken. These means were calculated using gill net spring and fall data and trawl data from all months. Mean salinities (untransformed) were compared using the same GLM model used for catch rates.

Relationships between mean catch rates by species and gear and mean salinities among all estuaries were evaluated with simple linear regression based on the following model:

$$Y_i = a + b_1 s_i + b_2 s_i^2 + b_3 s_i^3 + e_i$$

where Y = mean catch rate for estuary 1 to 7

a = y intercept

b = regression coefficient

s = mean salinity for estuary 1 to 7

e = error

Procedure REG from the SAS statistical analysis system was employed to perform the regressions. Since there was a very limited amount of data (i.e., seven estuaries) to judge each relationship, the probability was fairly high that a significant relationship could go unreported (the power of the test was weak due to the small sample size). Therefore, any relationship that showed significance at $P < 0.15$ was called near-significant and was discussed. Coefficients of determination (R^2) and probability level are reported for the simplest statistically significant or near-significant model that was calculated (note that a simple linear model that was significant at $P < 0.15$ would be reported even if a more complex quadratic relationship was significant at $P < 0.05$). All regressions were run with and without data from the Lavaca-Colorado Estuary because catch rates for several species from Lavaca-Colorado appeared to be outliers. Mean salinities from trawl data were used with trawl mean catch rates, and mean salinities from gill nets were used with gill net mean catch rates.

A literature search was performed to locate and summarize available information on salinity tolerances for estuarine animals (Table 6.7.3, based on Table 9-1 in TDWR 1983 by Gary L. Powell, with additional references added by the same compiler). This information was compiled to facilitate qualitative comparisons and discussion.

Trawl Results

Blue crab. Four different catch rate groups were evident from the results of the blue crab analysis ($P < 0.05$; Figure 6.7.1). The Mission-Aransas Estuary had the highest catch rate among the seven estuaries. The group of estuaries having the second highest mean catch rate was comprised of the Trinity-San Jacinto, Guadalupe, and lower Laguna Madre estuaries. The third highest catch rate was in the upper Laguna Madre, and the lowest catch rate was in the Lavaca-Colorado and the Nueces estuaries.

None of the regressions relating mean catch rates to mean salinity for blue crabs were significant. A simple linear regression with data from the Lavaca-Colorado Estuary removed was very near the $P < 0.15$ level ($P < 0.17$; $R^2 = 0.39$). This regression suggests a simple linear inverse relationship between blue crab catch rates and mean salinity within an estuary.

Table 6.7.2. Sample sizes and months used to analyze other trawl data.

Species	Sample Sizes	Months
Blue crab	8,648	January - December
Brown shrimp	6,484	April - December
White shrimp	3,604	August - December
Atlantic croaker	7,204	February - November
Gulf menhaden	8,648	January - December
Pinfish	8,648	January - December
Southern flounder	8,648	January - December

Brown shrimp. Brown shrimp catch rates were divided into six non-overlapping groups ($P < 0.05$; Figure 6.7.2). The highest catch rate was found in the Mission-Aransas Estuary followed by the Guadalupe, Nueces, Lavaca-Colorado and Trinity-San Jacinto, upper Laguna Madre, and the lower Laguna Madre.

A near-significant cubic relationship between brown shrimp mean catch rates and mean salinity within an estuary was found ($P < 0.09$; $R^2 = 0.83$) when the data for the Lavaca-Colorado Estuary was removed. The need for a cubic relationship to explain the data was evident from the asymmetrical figure formed about the point having the highest catch rate (Mission-Aransas). A cubic curve also was a better fit because it helped to match the higher catch rate in the upper Laguna Madre, which had a higher salinity than the lower Laguna Madre. This relationship suggests that brown shrimp in Texas estuaries reached their highest densities (highest mean catch rates) when mean estuarine salinities were between 15 and 25‰.

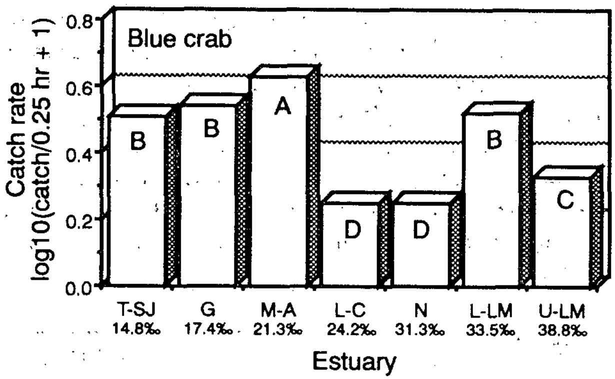


Figure 6.7.1. Mean catch rates (Log_{10} [catch/0.25 h + 1]) for blue crab by estuary using TPWD-CF monitoring program trawl samples collected during the period January through December, 1983 through 1988 inclusive.

White shrimp. Five distinct white shrimp catch rate groups were identified among the seven Texas estuaries ($P < 0.05$; Figure 6.7.3). The Trinity-San Jacinto Estuary had the highest catch rate, followed by the next highest group comprised of the Mission-Aransas and Guadalupe estuaries. The Lavaca-Colorado Estuary group overlapped the previous group and the Nueces Estuary group which had a slightly lower catch rate. The upper Laguna Madre had the next-to-lowest, and the lower Laguna Madre had the lowest catch rates.

A significant simple linear inverse relationship was found between white shrimp mean trawl catch rates and mean estuarine salinity ($P < 0.01$; $R^2 = 0.74$). This relationship showed that within Texas estuaries, the density of white shrimp decreases as mean estuarine salinities increase from 15 to 39‰.

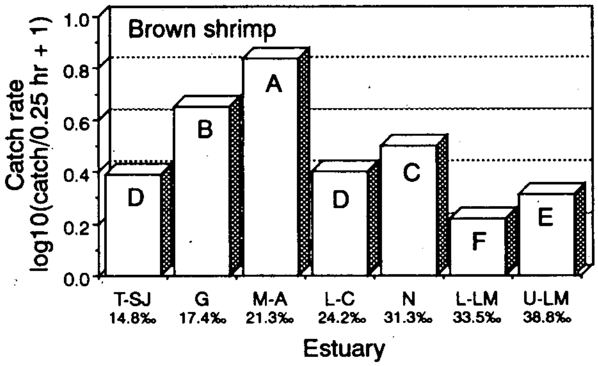


Figure 6.7.2. Mean catch rates (Log_{10} [catch/0.25 h + 1]) for brown shrimp by estuary using TPWD-CF monitoring program trawl samples collected during the period April through December, 1983 through 1988 inclusive.

Atlantic croaker. Five distinct groups having different catch rates resulted from the Atlantic croaker analysis ($P < 0.05$; Figure 6.7.4). The three highest catch-rate groups were comprised of single estuaries. They were, from highest to lowest, the Nueces, Mission-Aransas, and Lavaca-Colorado. The fourth group was comprised of the Trinity-San Jacinto, Guadalupe, and lower Laguna Madre. The fifth group had a single member, the upper Laguna Madre.

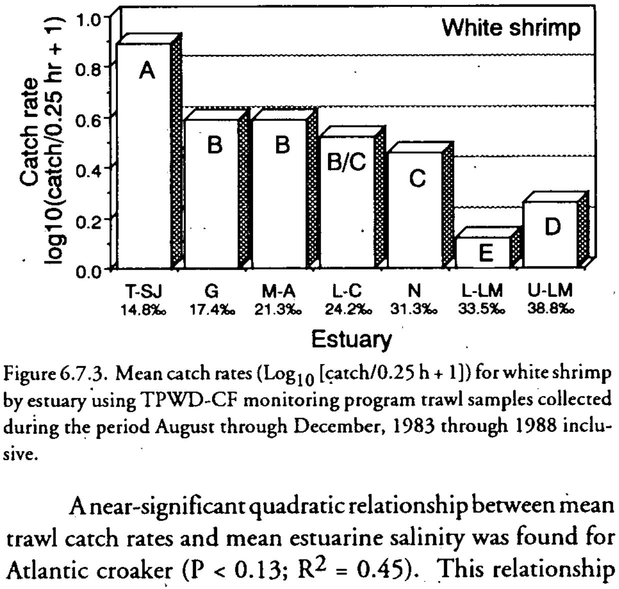


Figure 6.7.3. Mean catch rates (Log_{10} [catch/0.25 h + 1]) for white shrimp by estuary using TPWD-CF monitoring program trawl samples collected during the period August through December, 1983 through 1988 inclusive.

A near-significant quadratic relationship between mean trawl catch rates and mean estuarine salinity was found for Atlantic croaker ($P < 0.13$; $R^2 = 0.45$). This relationship suggests that Atlantic croaker had their highest densities in estuaries having mean salinities from 20 to 30‰.

Gulf menhaden. Four distinct catch rate groups resulted from the Gulf menhaden analysis ($P < 0.05$; Figure 6.7.5). The Guadalupe and Trinity-San Jacinto estuaries had the highest mean catch rates. The next highest group included the Mission-Aransas and Lavaca-Colorado. The Nueces and upper Laguna Madre had the next-to-lowest mean catch rates, and the lower Laguna Madre had the lowest.

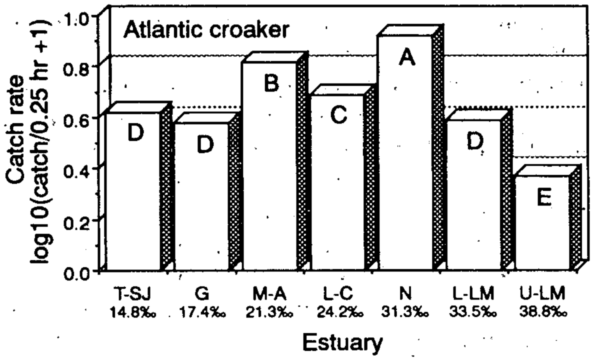


Figure 6.7.4. Mean catch rates (Log_{10} [catch/0.25 h + 1]) for Atlantic croaker by estuary using TPWD-CF monitoring program trawl samples collected during the period February through November, 1983 through 1988 inclusive.

A significant simple linear inverse regression was found for Gulf menhaden caught in trawls ($P < 0.01$; $R^2 = 0.84$). The mean density of Gulf menhaden decreased as mean estuarine salinities increased from 15 to 39‰.

Pinfish. Six unique catch rate groups were identified by the analysis ($P < 0.05$; Figure 6.7.6). The highest catch rates were observed in two estuaries—Nueces and the lower Laguna Madre. The remaining groups were comprised of single members, with the next highest catch rate being in the Mission-Aransas followed by the upper Laguna Madre, Guadalupe, Lavaca-Colorado, and Trinity-San Jacinto.

A near significant simple linear relationship between mean catch rate and mean estuarine salinity was found for pinfish ($P < 0.11$; $R^2 = 0.30$). However, a highly significant quadratic relationship was found for trawl-caught pinfish once data from the Lavaca-Colorado Estuary was removed ($P < 0.01$; $R^2 = 0.97$). The simple linear relationship suggests an increase in pinfish density with increasing mean salinity. The equation for the quadratic relationship showed the highest catch rates (densities) occurred between 30 and 35‰.

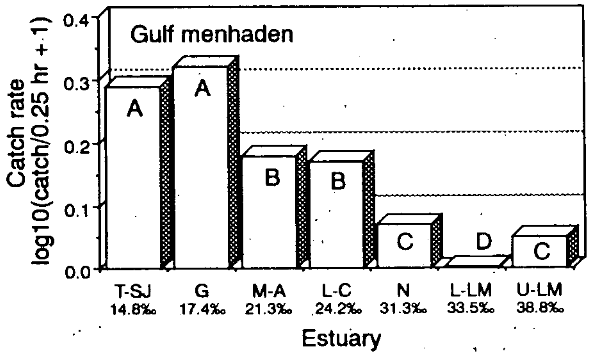


Figure 6.7.5. Mean catch rates (Log_{10} [catch/0.25 h + 1]) for Gulf menhaden by estuary using TPWD-CF monitoring program trawl samples collected during the period January through December, 1983 through 1988 inclusive.

Southern flounder. Flounder catch rates among estuaries formed three distinct groups ($P < 0.05$; Figure 6.7.7). The Mission-Aransas had the highest mean catch rate, followed by the Guadalupe Estuary group. Catch rates for the Nueces Estuary and upper Laguna Madre were not significantly different from either the Guadalupe group or the group comprising the lowest catch rates, the Trinity-San Jacinto, Lavaca-Colorado, and lower Laguna Madre.

A near-significant cubic relationship was found for trawl-caught flounder once data from Lavaca-Colorado was removed, suggesting that the highest densities of flounder were found in estuaries having mean salinities around 20‰ ($P < 0.09$; $R^2 = 0.84$). Inspection of the data with predicted values shows the cubic relationship was required because the catch rates were asymmetrically distributed about the optimum.

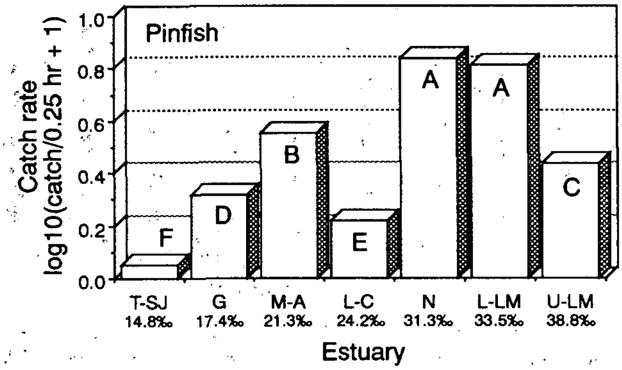


Figure 6.7.6. Mean catch rates (Log$_{10}$ [catch/0.25 h + 1]) for pinfish by estuary using TPWD-CF monitoring program trawl samples collected during the period January through December, 1983 through 1988.

Gill Net Results

All the a posteriori comparisons of gill net data showed significant differences in mean catch rates among estuaries. However, no significant or near-significant regressions were found for any finfish caught in gill nets. There does not appear to be any simple linear relationship between mean catch rate (density) and mean estuarine salinity for larger fish.

Black drum. The upper and lower Laguna Madre, Trinity-San Jacinto, and Mission-Aransas had the highest black drum catch rates ($P < 0.05$; Figure 6.7.8). The lowest mean catch rate was observed in the Nueces Estuary. The mean catch rates from the Trinity-San Jacinto and Mission-Aransas estuaries could not be statistically separated from either the high means from the upper and lower Laguna Madre estuaries or the lower means found in the Lavaca-Colorado and Guadalupe estuaries. Also, the relatively low sample means from the Lavaca-Colorado and the Guadalupe estuaries could not be separated from the low mean found in the Nueces Estuary.

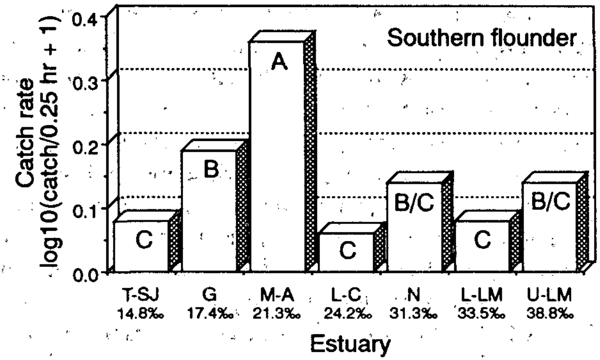


Figure 6.7.7. Mean catch rates (Log$_{10}$ [catch/0.25 h + 1]) for southern flounder by estuary using TPWD-CF monitoring program trawl samples collected during the period January through December, 1983 through 1988 inclusive.

Gulf menhaden. Four statistically distinct catch rates were found for Gulf menhaden ($P < 0.05$; Figure 6.7.9). The highest and the next-to-highest mean catch rates occurred in the Trinity-San Jacinto Estuary and in the Nueces Estuary, respectively. The group having the third highest mean catch rate was comprised of the Lavaca-Colorado, Guadalupe, Mission-Aransas, and upper Laguna Madre estuaries. However, the upper Laguna Madre Estuary mean was not statistically different from the estuary having the lowest mean catch rate, the lower Laguna Madre.

Red drum. Red drum catch rates separated into four statistically different groups ($P < 0.05$; Figure 6.7.10). The highest catch rate was observed in the lower Laguna Madre. A group having the next highest red drum catch rate included the Trinity-San Jacinto, Guadalupe, and Mission-Aransas estuaries. An intermediate set of lowest mean catch rates included Mission-Aransas (also a member of the preceding group), Lavaca-Colorado, and Nueces estuaries. The lowest red drum mean catch rate was observed in the upper Laguna Madre.

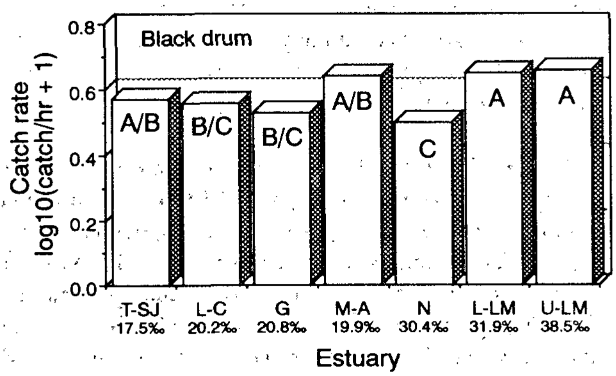


Figure 6.7.8. Mean catch rates (Log$_{10}$ [catch/10 h + 1]) for black drum by estuary using TPWD-CF monitoring program spring and fall gill net samples collected during the period 1977 through 1989 inclusive.

Southern flounder. Southern flounder catch rates also grouped into several statistically similar groups, but none of the groups, were entirely unique ($P < 0.05$). The highest mean catch rates were in the lower Laguna Madre followed by the Guadalupe which was lower but not significantly lower (Figure 6.7.11). The catch rate of the Guadalupe group was slightly higher, though not significantly higher, than the catch rates of the Trinity-San Jacinto, Nueces, and upper Laguna Madre estuaries. The latter three had slightly higher, though not significantly higher, catch rates than the Lavaca-Colorado Estuary. The Lavaca-Colorado catch rate was also slightly higher, though not significantly higher, than the Mission-Aransas catch rate.

Spotted seatrout. Spotted seatrout catch rates in gill nets was significantly higher in the lower Laguna Madre than in any other estuary ($P < 0.05$). There was considerable overlap in mean catch rates for the remaining six estuaries (Figure 6.7.12). The Nueces catch rate was the second highest, but was not significantly higher than the Guadalupe rate. The Guadalupe catch rate was slightly, though not significantly, higher than the Trinity-San Jacinto, Lavaca-Colorado, and Mission-Aransas catch rates, and the latter three catch rates were slightly, though not significantly, higher than the upper Laguna Madre catch rate, the lowest among the estuaries.

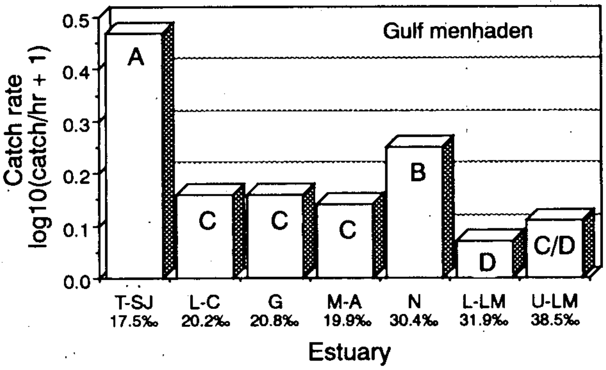


Figure 6.7.9. Mean catch rates (Log_{10} [catch/10 h + 1]) for Gulf menhaden by estuary using TPWD-CF monitoring program spring and fall gill net samples collected during the period 1977 through 1989 inclusive.

Striped mullet. The comparison of mean catch rates among the seven estuaries (Figure 6.7.13) revealed only two groups for striped mullet ($P < 0.05$). The group having the higher mean catch rate was comprised of the Mission-Aransas, lower Laguna Madre, and Guadalupe estuaries. The Trinity-San Jacinto, Lavaca-Colorado, Nueces, and upper Laguna Madre estuaries comprised the group with lower mean catch rates.

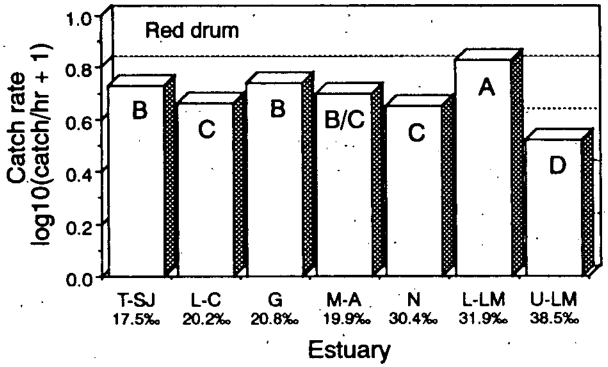


Figure 6.7.10. Mean catch rates (Log_{10} [catch/10 h + 1]) for red drum by estuary using TPWD-CF monitoring program spring and fall gill net samples collected during the period 1977 through 1989 inclusive.

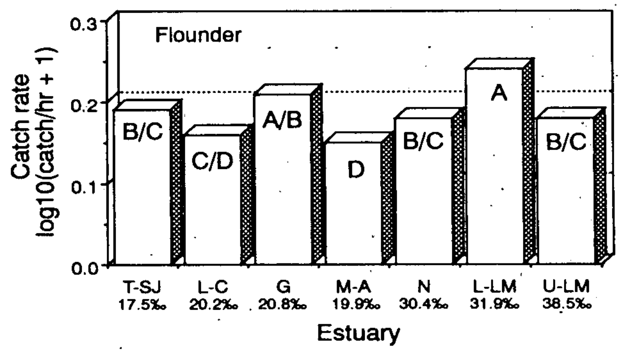


Figure 6.7.11. Mean catch rates (Log_{10} [catch/10 h + 1]) for southern flounder by estuary using TPWD-CF monitoring program spring and fall gill net samples collected during the period 1977 through 1989 inclusive.

Shellfish Discussion

Blue crab. Laboratory investigations have reported blue crabs to have a wide salinity tolerance range, 2 to 30‰ (Holland et al. 1971; Laird and Haefner 1976). Mahood et al. (1970) reported blue crabs had the greatest temperature tolerance at a salinity of 24‰. Other field observations have reported conflicting observations about blue crab salinity preferences (Gunter 1950; Hawley 1963; More 1969; Table 6.7.3). Current TPWD data used in this analysis generally showed higher catch rates in estuaries with lower salinities and lower catch rates in estuaries with higher salinities. There were two exceptions to this generalization: the Lavaca-Colorado had intermediate salinities and a low blue crab catch rate, and the lower Laguna Madre had high salinities and a fairly high catch rate. The TPWD analysis in the Guadalupe Estuary (Section 6.2, Figure 6.2.1) showed blue crabs were more abundant in areas having salinities from 10 to 25‰.

In the lower Laguna Madre, it is possible the low salinities which blue crab require for mating (Hines et al. 1987) have been provided by the Arroyo Colorado or freshenings within South Bay. South Bay has a small volume

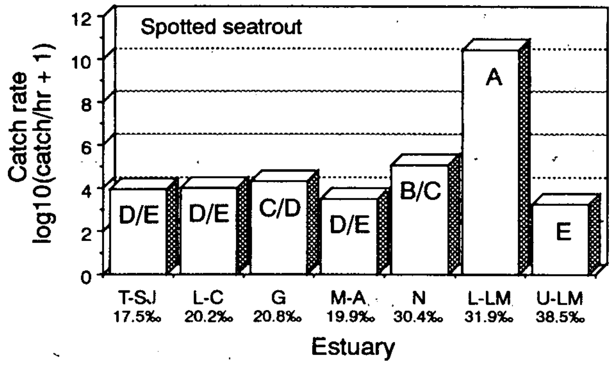


Figure 6.7.12. Mean catch rates (Log$_{10}$ [catch/10 h + 1]) for spotted seatrout by estuary using TPWD-CF monitoring program spring and fall gill net samples collected during the period 1977 through 1989 inclusive.

(high tide = 32 x 10^6 cu m or 22 x 10^3 acre-ft; Diener 1975). Therefore, local rainfall events could produce low salinities. In addition, seagrass meadows have been reported as prime habitat for blue crab (Zimmerman and Minello 1984). Therefore, if blue crabs can find the salinities they require to mate, a preferred habitat of seagrass meadows is very abundant in the lower Laguna Madre. In addition, the relatively warm year-round temperatures in the lower Laguna Madre would permit blue crabs to mate whenever low salinities occurred.

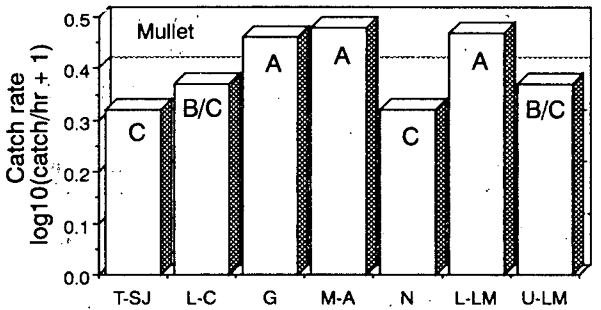


Figure 6.7.13. Mean catch rates (Log$_{10}$ [catch/10 h + 1]) for striped mullet by estuary using TPWD-CF monitoring program spring and fall gill net samples collected during the period 1977 through 1989 inclusive.

In the upper Laguna Madre Estuary, where there were also large areas of seagrass meadows, a lower blue crab catch rate was found. This may be a result of the much higher salinities experienced throughout this estuary combined with an absence of low salinity refuges within the estuary that blue crabs can use for mating. The crabs in the upper Laguna Madre may be migrants from nearby estuaries or the result of matings during infrequent freshenings.

Low blue crab catch rates in the Lavaca-Colorado Estuary were unexpected. This estuary had salinities low enough for blue crab to mate, and extensive habitat was available. Although the amount of seagrass meadows in this estuary was relatively small, there were extensive *Spartina* marshes. *Spartina* marshes have been shown to provide adequate habitat for blue crab in Galveston Bay (Thomas 1989). It is possible that what is referred to as a "drowning marsh" (either from subsidence or sea level rise) provides better habitat than a stable *Spartina* marsh (Zimmerman et al. 1991). Therefore, the drowning marshes associated with high subsidence in the Trinity-San Jacinto Estuary may be better habitat than marshes in the Lavaca-Colorado Estuary that do not have a high subsidence rate.

Eastern oyster. Historical documentation on freshwater needs and salinity preferences for oysters is extensive (Quast et al. 1988a, Table 6.7.3). Oyster reefs are reported to do best when salinities and temperatures are in the ranges of 10 to 30‰ and 20 to 30 °C. They also benefit from frequent freshenings which lower salinities to less than 20‰, temporarily reducing predator and disease organisms (Kinne 1971). These conditions do not commonly occur in Texas estuaries south of Corpus Christi, and no large commercial oyster reefs are currently located south of the Mission-Aransas Estuary, with the exception of South Bay (in the lower Laguna Madre). Historical records do show, however, that significant reefs and commercial harvests occurred in the Nueces Estuary (Collins and Smith 1893). The demise of the Nueces Bay reefs and harvests may have been caused by the combination of extensive shell dredging, increased salinities from reduction or alteration of freshwater inflows, and increased salinities from brine discharge from oil production.

The South Bay oyster reefs have been well documented. Speculation as to how and why oysters do well in an area which commonly has salinities higher than 30‰ includes suggestions of species adaptation and acclimation. Another explanation might be that South Bay has a small enough volume that it easily freshens with local rainfall, and, in its warm waters, oysters may be able to spawn during any season of the year. This phenomenon, however, still needs more study to determine whether these oysters have genetically adapted to the environment in South Bay (J.D. Grey, TPWD; pers. comm. 1989), or the salinity varies just enough to permit oysters to complete their normal life cycle.

Shrimp. The mean catch rates and the general salinity conditions in each estuary correspond with other field and laboratory findings about salinity preferences for brown and white shrimp (Table 6.7.3). White shrimp have been reported to prefer laboratory salinities in the range of 5 to

Table 6.7.3. Salinity limits, preferences, and optima for selected Texas estuarine-dependent species.

Group/species	Limits		Preference or optimum (‰)	Remarks
	min. (‰)	max. (‰)		
INVERTEBRATES				
Eastern oyster *Crassostrea virginica*	<5	>22	9 to 20	Maximum adult range beyond limits for reproduction; best conditions of salinity for eastern oyster (Moore 1898).
	<6			Gametogenesis inhibited by prolonged low salinity exposure; up to three to four months required to regain normal gonadal activity after salinity increases toward the optimum (Butler 1949).
	5 to 7.5			Normal gonadal development near 7.5‰; however, oysters with previously ripe gonads spawn when subjected to low (5‰) salinities (Loosanoff 1953).
			20 to 21	Larval spat setting requirement in Galveston Bay, Texas (Hopkins 1930).
	5 to 8		12.5 to 25	Minimum tolerance of larvae 5 to 8‰; less than 12.5‰, adult reproduction is impaired while more than 25‰, predation and disease increase greatly, especially with high temperatures (Kinne 1971).
			19 to 30	Maximum survival (80% contour plot) in lab of 2-day larvae in laboratory at 19 to 30.5 °C temperatures (Lough 1975).
			8 to 30.5	Maximum survival (60% contour plot) of 8-day larvae in laboratory at temperatures >21 °C (Lough 1975).
			>33	Maximum growth (100% contour plot) of 8-day larvae in laboratory at temperatures >19 °C (Lough 1975).
			18 to 35	Optimum (80% contour plot) for both larval survival and growth in laboratory at temperatures >30 °C (Lough 1975).
			15 to 22.5	Optimum for juvenile growth and development (Chanley 1957).
	0.5	27		Field distribution in Mesquite Bay, Texas (Hoese 1960).
	1.5	39.0	25 to 29	Early experimentally derived salinity limits; developmental stages best from 25 to 29‰ (Amemiya 1926).
	0 to 2			Oysters can survive fresh water for several days, increasing to about a month at 2‰ salinity (Gunter 1950).
	≤2			Oysters can survive up to four weeks in low salinity at 20 to 27°C temperatures; mortality increases severely at higher temperatures in Galveston Bay, Texas (Hofstetter 1977).
			15 to 30	Optimum range of salt content (Hofstetter 1959).
	5	40	5 to 15	Tolerance limits and optimum range for growth and survival; higher optimum (10 to 28‰) in cooler waters of northern latitudes, such as Chesapeake Bay (Galtsoff 1964).
		43.5 to 45		Distribution limit in Redfish and Corpus Christi bays, Texas (Copeland and Hoese 1966).
			15 to 20	Ideal salinity conditions with lowest seasonal salinities in late summer and fall (Abbott et al. 1971).
	2 to 4	18 to 22	10.0 to 16.0	Most productive reefs of Mississippi Sound subject to 10.0 to 16.0‰ average conditions (Eleuterius 1977).

Table 6.7.3. (continued)

Group/species	Limits		Preference or optimum (‰)	Remarks
	min. (‰)	max. (‰)		
			15 to 30	Best growth in reasonably stable salinity (Overstreet 1978).
	3			Lower tolerance limit about 3‰ (Gunter et al. 1974).
	8 to 10			Lower limit of predator *Thais haemastoma*, a gastropod oyster drill or conch (Gunter 1950; Gunter et al. 1974).
	<10			Low incidence of infection with fungus, *Dermocystidium marinum* (presently known as the protozoan, *Perkinsus marinus*); infection increases at more than 10‰ and mortality increases severely at both high salinities and high temperatures (Mackin 1961).
	0	34		Range of infection of Gulf coast populations with *Perkinsus marinus*; prevalence and intensity of infection correlated with salinity, temperature, and condition index; however, salinity alone explains only about 20% of infections, while other factors (polluted runoff from agricultural and industrial land use) may control variability of infection from site to site (Craig et al. 1989).
Blue crab *Callinectes sapidus*	22.9	32.4	>30.0	Range for capture of egg-bearing females near Aransas Pass, Texas (Gunter 1950).
			23 to 28	Optimum range for hatching of eggs in Virginia estuaries (Sandoz and Rogers 1944).
			>20	Occurrence of spawning and early development in Texas bays (More 1969).
			<1.9	Peak abundance of juvenile blue crabs in Texas bays (1965) (More 1969).
	<1.0		2 to 21	Lethal limit at optimum (29 °C) temperature; range of little effect on juvenile growth and survival in laboratory (Holland et al. 1971).
	0			Observed freshwater populations in Louisiana (Gunter 1938).
	2.8	40.6		Field distribution in Mesquite Bay, Texas (Hoese 1960).
	2.0	37.2	10.0 to 20.0	Field distribution in Copano and Aransas bays, Texas; range of greatest abundance (Gunter 1950).
		117		Field collection in Laguna Madre de Tamaulipas, Mexico; high salinity briefly tolerated (Hildebrand 1958).
		45		Blue crabs observed leaving upper Laguna Madre, Texas, as salinity increased (Hawley 1963).
	2	60		Field distribution in Texas bays and lagoons of northwestern Gulf of Mexico (Hedgpeth 1967).
			24.2	Salinity for widest thermal tolerance zone in adult blue crab (Mahood et al. 1970).
	<5	>18		Duration of megalops stage (6 to 20 days) with highest densities at polyhaline (>18‰) sites, while juveniles most abundant at oligohaline (<5‰) in South Carolina (Mense and Wenner 1989).

Table 6.7.3. (continued)

Group/species	Limits min. (‰)	Limits max. (‰)	Preference or optimum (‰)	Remarks
	0	40	0 to 27	Optimum catch range with 10 to 35 °C temperatures (Copeland and Bechtel 1974).
			10 to 30	Range of no effect on metabolic consumption of oxygen (respiration) (Laird and Haefner 1976).
Brown shrimp *Penaeus aztecus*	2	40		Range of equal postlarval growth over 23 to 25 °C; survival 90 to 100% in laboratory (Zein-Eldin 1963).
	<10			Marked reduction in postlarval tolerance at low (7 to 15 °C) temperatures to low (5‰) salinity (Zein-Eldin and Aldrich 1965).
			15 to 35	Range of increased postlarval growth at temperatures >25 °C; decreased growth at less than 15‰ in laboratory (Zein-Eldin and Griffith 1969).
	<5	>40		Range at which 80% of 10 to 15 mm postlarvae survive; 12-hour acclimation in laboratory (Zein-Eldin and Griffith 1969).
			>15	Appeared to enhance survival and growth of postlarvae in Barataria Bay, Louisiana (St. Amant et al. 1966).
			>15	Commercial catches poor in years when postlarvae were present in Louisiana bays with <15‰ (Gaidry and White 1973).
			29.9	Median salinity average of postlarval distribution during March to April in laboratory gradient tank (Keiser and Aldrich 1976).
			20.6	Median salinity average of postlarval distribution during May to July in laboratory gradient tank (Keiser and Aldrich 1976).
			10.0 to 19.9	Range at which juveniles were more abundant based on field population distributions (Gunter et al. 1964).
	0.2	30	10 to 30	Field distribution in Caminada Bay, Louisiana; and range for 91.8% of juveniles collected (Crowe 1975).
	0.5	45.3		Field distribution in Mesquite Bay, Texas (Hoese 1960).
			<17	Preference of juvenile (70 mm) shrimp in laboratory at >26 °C temperature (Venkataramiah et al. 1977a).
			15 to 25	Optimal range for subadult (95 mm) shrimp in laboratory at <25 °C temperature (Venkataramiah et al. 1977a).
			8.5 to 17	Optimal range for juvenile growth on low (40%) protein diet in laboratory at 21 to 31 °C temperatures; low salinity essential for fast postlarval growth from age 16 days and older (Venkataramiah et al. 1977b).
			27.6 to 28.3	Isosmotic salinity conditions for shrimp >100 mm length; osmoregulation above 28.3‰ better than white shrimp (McFarland and Lee 1963).
	2.1	36.6	15.0 to 19.9	Field distribution in Copano and Aransas bays, Texas; range of greatest abundance (Gunter 1950).

Table 6.7.3. (continued)

Group/species	Limits		Preference or optimum (‰)	Remarks
	min. (‰)	max. (‰)		
		69.0		Field collection in Laguna Madre, Texas (Simmons 1957).
	0.8			Lower distribution limit in Grand and White Lakes, Louisiana (Gunter and Shell 1958).
	0.22			Field collection in St. Lucie Estuary, Florida (Gunter and Hall 1963).
	5	70		Field distribution in Texas bays and lagoons of northwestern Gulf of Mexico (Hedgpeth 1967).
	0.1			Field collection in North Carolina estuaries (Williams and Deubler 1968).
			5	Acclimation at low (5‰) salinity provides near to optimum resistance to high temperatures and 5 to 25‰ salinities in laboratory tests (Wiesepape et al. 1972).
	9	40		No optimum salinity established with 20 to 35 °C temperatures, based on field catch rates (Copeland and Bechtel 1974).
White shrimp *Penaeus setiferus*	<2	>40		Range at which 80% of 8 to 50 mm (postlarvae to juvenile) shrimp survive; 48-hour acclimation in laboratory (Zein-Eldin and Griffith 1969).
			5 to 15	Increased growth at this range (and >25 °C) more than two times tissue production of postlarvae at 25 to 35‰ in laboratory (Zein-Eldin and Griffith 1969).
			28.0	Median salinity average of postlarval distribution during May to July in laboratory gradient tanks (Keiser and Aldrich 1976).
			21.0	Median salinity average of postlarval distribution during August to November in laboratory gradient tanks (Keiser and Aldrich 1976).
		47.96		Field collection of small white shrimp (23 to 76 mm) in Laguna Madre de Tamaulipas, Mexico (Hildebrand 1958).
	0.42			Lower distribution limit in Grand and White lakes, Louisiana; young shrimp 140 times more abundant at 0.7 to 0.8‰ (Gunter and Shell 1958).
			27.6 to 28.3	Isosmotic salinity conditions for shrimp >100 mm length; osmoregulation below 27.6‰ better than brown shrimp (McFarland and Lee 1963).
	1	34	1 to 20	Field distribution in Caminada Bay, Louisiana; range for 91.1% of juveniles collected (Crowe 1975).
	2.9	45.3		Field distribution in Mesquite Bay, Texas (Hoese 1960).
	2.1	36.6	10.0 to 14.9	Field distribution in Copano and Aransas bays, Texas; range of greatest abundance, although still common at ≤4.9‰ (Gunter 1950).
	2.9	45.3		Field distribution in Mesquite Bay, Texas (Hoese 1960).

Table 6.7.3. (continued)

Group/species	Limits min. (‰)	Limits max. (‰)	Preference or optimum (‰)	Remarks
	2	45		Field distribution in Texas bays and lagoons of northwestern Gulf of Mexico (Hedgpeth 1967).
			<10	Preference based on population distributions (Gunter et al. 1964).
	0	38		Optimum catch over entire salinity range with 20 to 38 °C temperatures (Copeland and Bechtel 1974).
FISH				
Atlantic croaker *Micropogonias undulatus*	>5	<45	25 to 35	Limits for fertilization of eggs and range of successful hatching in laboratory; the eggs become nonbuoyant, sink, and die at salinities <25‰ (Thomas and Boyd 1989).
			15 to 20	Optimum salinity range for Day 1 larvae in laboratory under optimum temperatures (Thomas and Boyd 1989).
			15 to 35	Range of no salinity-related mortality during the pelagic larval stage in laboratory under optimum temperatures (Holt and Banks 1989).
	0.1 to 0.9			Lowest range of abundant catches in Grand and White lakes, Louisiana (Gunter and Shell 1958).
			0 to 15	Range of common occurrence in Barataria Bay, Louisiana (Rogers 1979).
	0	37	6 to 15	Field distribution in Mississippi Sound and range of common occurrence of juveniles (Overstreet and Heard 1978).
			<15	More abundant in Texas waters of less than 15‰ (Gunter 1945).
	0.4	34.4	5 to 15	Most abundant in mesohaline region of South Carolina estuaries; strong correlation noted between size and salinity, with second peak abundance of larger fish found at >18‰ (Miglarese and Shealy 1982).
	2	>60		Recorded occurrence in northwestern Gulf and Laguna Madre, Texas (Hedgpeth 1967).
Black drum *Pogonias cromis*	2.6	34.9	<5	Field distribution in Copano and Aransas bays, Texas; most abundant range 10.0 to 15.0‰ (Gunter 1945).
	<5	77		Field distribution in Texas bays and lagoons of northwestern Gulf of Mexico (Hedgpeth 1967).
	0	80		Field distribution in Laguna Madre, Texas; usual range is 25 to 50‰ (Simmons and Breuer 1962).
	5	40 to 45	20 to 30	Operational limits; range of optimum metabolic condition at 20 to 28 °C temperatures; maximum swimming performance at 28 °C and 20‰ (Wohlschlag 1977).

Table 6.7.3. (continued)

Group/species	Limits		Preference	Remarks
	min. (‰)	max. (‰)	or optimum (‰)	
Red drum				
Sciaenops ocellatus		<25		Eggs fertilized from fish acclimated to 26 to 32‰ in laboratory became nonbuoyant (sank into conditions unfavorable for survival) at salinities of less than 25‰ (Holt et al. 1981a).
			15 to 35	Range of no salinity-related mortality during the pelagic larval stage in laboratory under optimum temperatures (Holt and Banks 1989).
	2.1	32.4	<15	Field distribution in Copano and Aransas bays, Texas; greatest abundance at less than 15‰ (Gunter 1945).
	0	>50	20 to 40	Field distribution in Texas bays; range of preference (most abundant in 30 to 35‰); young mature in three to five years (Simmons and Breuer 1962).
			<50	Populations in Laguna Madre, Texas, severely limited by >50‰ (Simmons 1957).
	5 to 10	40 to 45	20 to 25	Operational limits; range of optimum metabolic condition at 20 to 28 °C temperatures in laboratory (Wohlschlag 1977).
Spotted seatrout				
Cynoscion nebulosus	<5		30 to 35	Lower limit especially important when temperature is low (<10 °C); peak spawning in Florida estuaries and lagoons at 30 to 35‰; larval survival reduced if salinity low (Tabb 1966).
			>30	Spawning occurs in Louisiana estuarine areas of higher salinity (Sabins and Truesdale 1974).
	5	50	20 to 45	No egg hatching success at extreme salinities; range of successful reproduction in laboratory under optimum temperatures (Thomas and Boyd 1989).
	5	50	10 to 35	No survival of Day 1 larvae at extreme salinities; survival range of larvae at early stage in laboratory under optimum temperatures (Thomas and Boyd 1989).
			10 to 40	Range of no salinity-related mortality during the pelagic larval stage in laboratory under optimum temperatures (Holt and Banks 1989).
		≤60	<45	"Young" collected up to about 60‰ in Laguna Madre, Texas; no spawning if salinity >45‰ (Simmons 1957).
		<55	15 to 35	Absent at more than 55‰ in Baffin and Alazan bays, Texas; most abundant range 15 to 35‰ (Breuer 1957).
	2.3	34.9	5 to 20	Field distribution in Copano and Aransas bays, Texas; more than 80% collected in 5 to 20‰ (Gunter 1945).
	<5	77		Field distribution in Texas bays and lagoons of northwestern Gulf of Mexico (Hedgpeth 1967).
	1.5	45.3		Field distribution in Mesquite Bay, Texas (Hoese 1960).
	10	45	20	Operational limits; optimum metabolic condition at 20 to 28 °C temperatures in laboratory (Wohlschlag and Wakeman 1978).

20‰ (Zein-Eldin and Griffith 1969; Keiser and Aldrich 1976). This study also showed white shrimp were most abundant in the fresher estuaries (less than 30‰) and least abundant in the more saline estuaries (greater than 30‰). Brown shrimp have been reported to prefer laboratory salinities in the range of 15 to 30‰ (Zein-Eldin and Griffith 1969; Venkataramiah et al. 1977a). This study also reports the highest mean catch rates in estuaries having salinities closest to this range (i.e., Mission-Aransas, Guadalupe, and Nueces estuaries) and lower catch rates in estuaries having lower or higher salinities. The distribution of these two species within the Guadalupe Estuary as shown in trawl data also reflected these salinity preferences by being more abundant in that portion of the estuary that consistently had salinities in their preferred ranges (figures 6.2.4 and 6.2.7).

Finfish Discussion

All mean catch rates for trawl-caught finfish showed either significant relationships with mean estuarine salinity ($P < 0.05$) or near-significant relationships ($P < 0.15$). This result suggests that the smaller fish caught in trawls (Meador et al. 1988) may be more sensitive to salinity regimes than larger fish of the same species caught in gill nets (Mambretti et al. 1990). Or, it could be that larger fish caught in gill nets were a reflection of differential fishing pressure, other differential natural predation rates among the estuaries, or the ability of the estuary to provide habitat needs as the individual of a species grows and matures.

For southern flounder and Gulf menhaden, there was a lack of correlation between trawl catch rates and gill net catch rates. This could be the result of a number of different phenomena. Possibly, the catch efficiency of different gears varies among estuaries. Different harvest rates on larger fish may occur among the estuaries and reduce densities below what would normally occur.

Only smaller finfish caught in trawls indicated any relationships between mean catch rate and mean estuarine salinity. No finfish species sampled by gill net showed any significant relationship between mean catch rates and mean estuarine salinity. However, all the finfish species investigated (regardless of capture gear) showed statistically significant differences in mean catch rates among the estuaries. Most of the differences in gill net data were relatively small and may be indicative of young survival, habitat availability, and harvesting of older (larger) members.

Conclusion

The analyses in this study, other published field studies, and laboratory experiments (Table 6.7.3) provide evidence that reductions in freshwater inflows that would significantly increase the mean salinity above 25‰ within a Texas estuary could reduce the densities of eastern oysters, white shrimp, and Gulf menhaden. Altering inflows such that salinities decrease to below 15‰ or increase to above 30‰ could reduce densities of brown shrimp, small Atlantic croaker, and small southern flounder. Pinfish densities appeared to have highest densities when near estuarine salinities were around 30‰.

No relationships between mean estuarine salinity and gill net catch rates were found for black drum, southern flounder, Gulf menhaden, striped mullet, red drum, or spotted seatrout. Therefore, no immediate effects on these species would probably be evident from reduced inflows. Delayed effects, however, may show up later if substantial vegetation or sediment changes occur as the result of inflow changes, and these species are dependent on these habitat types.

6.8 INFLOW EFFECTS ON FISHERIES HARVESTS

Introduction

Although living organisms with short life cycles are often used as "indicators" in evaluating an ecosystem's ambient quality, those with longer life spans (one year or greater) can be viewed as "integrators" of the environment's fluctuations over extended periods of time. In particular, fish production may be considered a measure of an aquatic ecosystem's overall health (Price 1979). Indeed, fishery species have aptly served as an early warning of impending trouble in a number of environmental crises (Johnson 1976). In this section, Texas seafood harvests are used as a surrogate for total fisheries production from the state's bays and estuaries. Analysis of the relationships among fisheries and key environmental factors, like freshwater inflows, can be used to determine their influence on the most economically important part of the biological production from our coastal waters.

Texas coastal fisheries. Most (up to 97.5%) of the coastal fishery species harvested along the Gulf of Mexico are considered estuarine-dependent; that is, they are dependent on the bays and estuaries for food or protection during at least some portion of their life cycles (Gunter 1967, 1980; Boesch and Turner 1984). The coastal fisheries are also recognized as one of the state's most valuable natural resources. For example, commercial and sport fishermen harvested more than 115 million pounds of coastal fish and shellfish in 1986, and the total annual impact of commercial fishing, sport fishing, and other recreational bay activities on the state's economy that year was estimated at more than $2.6 billion (Fensenmaier et al. 1987). The landings estimate does not include the 40 to 140 million pounds of menhaden which are harvested annually from Texas gulf waters, but landed and processed in Louisiana.

Shellfish. Texas bays and estuaries are not all equal producers of coastal fisheries (figures 6.8.1 and 6.8.2). During the recent interval from 1980 through 1987, 35% of the "shellfish" (bay shrimp, crabs, and oysters) commercially harvested in the five major estuarine systems between Galveston and Corpus Christi, Texas, were taken in the Trinity-San Jacinto Estuary. On the basis of pounds harvested per unit area each year (using acres of water surface at mean low tide from Diener 1975), this estuary produced an average of 32.6 kg/ha (29.1 lb/acre) annually (Table 6.8.1). However, the highest annual harvest rates are associated with the Mission-Aransas Estuary (56 kg/ha or about 50 lb/acre) and the Guadalupe Estuary (44.2 kg/ha or 39.4 lb/acre), even though they accounted for only 19% and 18%, respectively, of the combined shellfish harvest from the inshore bay systems. It should also be noted that the total commercial harvest, which is dependent on each unit of bay area, is substantially larger than the weights given above because much of the estuaries' shellfish production (up to five times more shrimp) is harvested offshore in the Gulf of Mexico.

Finfish. The 1980 through 1987 commercial harvests of "finfish" (marine fishes) were likewise greatest in the Trinity-San Jacinto Estuary and constituted 39% of the combined harvest taken from the five estuaries. Nevertheless, on the basis of areal production, this estuary's commercial harvest rate of 1.1 kg/ha (about 1 lb/acre, Table 6.8.1) is exceeded by that of the Nueces estuary (2.9 kg/ha or 2.5 lb/acre) and the Mission-Aransas Estuary (1.7 kg/ha or 1.5 lb/acre). Again, the total finfish harvest per unit of bay area is actually much larger, primarily because the recreational harvests from sportfishing are not included in this comparison.

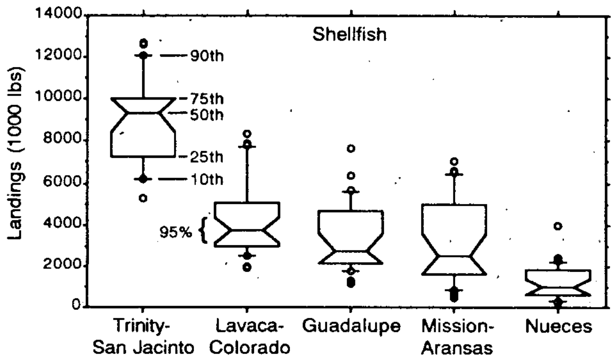


Figure 6.8.1. Percentile plot of commercial landings of shellfish species in Texas estuaries (1962 to 1987). The 10th, 25th, 50th (median), 75th, and 90th percentiles are shown for each estuary as well as data points below the 10th and the 90th percentiles. The "notch" shows the 95% confidence interval around the median.

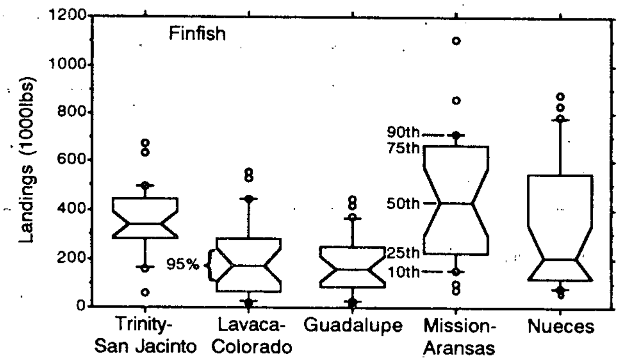


Figure 6.8.2. Percentile plot of commercial landings of finfish species in Texas estuaries (1962 to 1987). The 10th, 25th, 50th (median), 75th, and 90th percentiles are shown for each estuary as well as data points below the 10th and above the 90th percentiles. The "notch" shows the 95% confidence interval around the median.

Importance of Freshwater Inflows to Fisheries

The relationships between an organism and its environment are intricate and absolutely decisive in survival, growth, and ultimately, reproduction of the species. Few fluctuations can produce so many profound changes in an ecosystem as does change in the climatic regime (Stone et al. 1978; Kennedy 1990). Indeed, biologists often find connections between climate and fisheries, but the exact causative mechanisms of the biological effects are not always known (Knauss 1979; Austin et al. 1982). Nevertheless, significant correlations among the empirical data can lead to a better understanding of causative factors, as well as provide a simple tool for forecasting fisheries performance, especially when lack of knowledge about the causative mechanisms prevents development of an adequate deterministic model (Ulanowicz et al. 1982).

Variations in precipitation, rainfall runoff, or river discharge have been associated with changes in the catches of estuarine-dependent fisheries and have been used as an indicator of potential fish and shellfish production (Pearson 1948; Turner and Chadwick 1972; Sutcliffe 1972, 1973; Stevens 1977, 1979). In Texas, Hildebrand and Gunter (1953) were the first to demonstrate a significant relationship between rainfall and the catch of white shrimp (*Penaeus setiferus*). Other freshwater inflow relationships to Texas coastal fisheries were reported by Gunter and Hildebrand (1954), Gunter and Edwards (1969), Hackney (1978), Powell (1979), Texas Department of Water Resources (1980a, 1980b, 1981c, 1981d, 1981e, 1982c, and 1983), Armstrong (1982), Saila et al. (1982), and Mueller and Matthews (1987). This section reports on new relationships which have been developed from updated freshwater inflow and fisheries harvest data.

Table 6.8.1. Distribution and trends of commercial fisheries harvests in five Texas estuaries.

		Estuary				
Group and measure	Years	Trinity-San Jacinto	Lavaca-Colorado	Guadalupe	Mission-Aransas	Nueces
Shellfish—percent (%) of Texas total harvest						
	1962-1969	54.1	20.3	14.3	8.1	3.3
	1970-1979	45.7	18.8	15.3	14.9	5.2
	1980-1987	34.9	21.7	18.4	19.2	5.8
Shellfish—harvest in pounds per acre of water[a]						
	1962-1969	22.4	11.8	15.1	10.4	4.4
	1970-1979	25.6	14.8	22.0	26.1	9.5
	1980-1987	29.1	25.5	39.4	49.9	15.7
Finfish—percent (%) of Texas total harvest						
	1962-1969	22.5	21.8	12.4	36.4	6.8
	1970-1979	19.2	11.4	12.4	30.3	26.6
	1980-1987	38.7	5.0	5.9	19.3	31.0
Finfish—harvest in pounds per acre of water[a]						
	1962-1969	0.9	1.3	1.3	4.8	0.9
	1970-1979	1.1	0.9	1.9	5.5	5.1
	1980-1987	1.0	0.2	0.4	1.5	2.5

[a] Based on estuary surface area at mean low tide.

Assessing the Relationship Between Inflow and Harvest

Analytical data. Harvest data on seven species, red drum (*Sciaenops ocellatus*), black drum (*Pogonias cromis*), spotted seatrout (*Cynoscion nebulosus*), blue crab (*Callinectes sapidus*), eastern oyster (*Crassostrea virginica*), brown shrimp (*Penaeus aztecus*), and white shrimp, were used in the analysis. Annual catch data from the 1962 through 1976 commercial harvests of coastal fish and shellfish were obtained from *Texas Landings*, a cooperative publication of the U.S. Department of the Interior and the TPWD (Farley 1963 to 1969), and the U.S. Department of Commerce and the TPWD (Farley 1970 to 1978). Commercial landings data from 1977 through 1987 were obtained from the TPWD (Quast et al. 1988b). Similarly, the 1959 through 1976 annual catch and effort data from the Texas shrimp fishery was found in *Gulf Coast Shrimp Data*, a publication of the U.S. Department of the Interior (1960 to 1969) and the U.S. Department of Commerce (1970 to 1978), which was prepared in cooperation with the state fishery agencies of Alabama, Florida, Louisiana, Mississippi, and Texas. Additional shrimp fishery data from 1977 through 1987 was provided on computer tape to the TPWD by the U.S. Department of Commerce (Dr. John Poffenberger, Economics and Statistics Office, National Marine Fisheries Service, Miami, Florida; pers. comm. 1989). All harvest weights are given as whole animal weights except for shrimp and oysters. Heads-on shrimp weights were calculated by multiplying the heads-off weights by 1.54 (white shrimp) or 1.61 (brown shrimp). Oyster weights are given only as shucked meat weights.

Environmental variables which were tested for relationships to the coastal fisheries harvests included seasonal (bimonthly) freshwater inflows and average minimum air temperatures. The hydrological data was compiled into two categories: (1) USGS gaged flows measured at the last nontidally affected streamgaging station in each contributing river basin, and (2) combined inflows from all gaged and ungaged river and coastal drainage basins that contribute fresh water to each estuarine system. The ungaged inflows were estimated using a rainfall runoff modeling procedure which is described in Section 4.1 of this report. Air temperatures from the National Weather Service station nearest to each estuary were obtained from *Climatological Data, Texas (1959-1988)*, a publication of the National Climatic Data Center, U.S. Department of Commerce, Ashville, North Carolina.

Statistical methods. Analysis of the time series records of the dependent variable (fishery harvest) and the independent predictor variables (freshwater inflow, temperature, and fishing effort where available) was accomplished using the "All Possible Subsets Regression" (9R) computer program contained in the BMDP statistical package (Dixon et al. 1988). The statistical procedure allows the ten best subsets of predictor variables to be identified using the Mallows' C_p criterion, which is defined as

$$C_p = RSS / s^2 - (n - kp')$$

where RSS is the residual sum of squares, s^2 is the residual mean square, n is the number of cases in the analysis, k is 2, and p′ is the number of variables (including the intercept) in the regression equation. The "best" regression equation is the one that has the lowest C_p value. Since Mallows' C_p has a built-in penalty to guard against overfitting the model, as often happens when the selection criteria is based on maximization of R^2 (the coefficient of determination), it encourages selection of regression equations with fewer variables. Nevertheless, it is still sometimes difficult to say which of the subsets is really best based on Mallows' C_p alone. Obviously, from a scientific standpoint, one should generally select the equation which is most meaningful to the research problem of interest. This is the intent of the procedure used here.

In practice, the alignment of the time series data files between dependent and independent variables is not the same among species in the fishery analyses (Figure 6.8.3). This is a standard procedure which recognizes that environmental effects on the growth and survival of the young may not be expressed until the affected age-class matures and enters the commercially exploited adult population sometime later (Hjort 1914; Gulland 1965; May 1974). Consideration of the life histories of the fishery species requires that the probable lagged effect in time of the environmental factors on harvest must be appropriately taken into account. The dependent variable (harvest) must be lagged after the independent variables (freshwater inflow and temperature); therefore, the independent variables are chronologically antecedent to the harvest in the time series analyses.

Nonlinearities and curvilinear relationships among the variables were transformed to linearity most successfully with natural logarithms (ln) in this analysis. Regression analyses were performed with both transformed and nontransformed data, and the "best" equation was selected on the basis of which produced the best statistical fit. The tolerance level in the BMDP-9R program was set below the default level (0.00001 instead of 0.0001), to decrease the chance of obtaining regression equations with collinear variables that may have little relationship to the dependent variable. The unexplained variation (residuals) from the regression analyses were also statistically examined using the Runs test for serial correlation and the Durbin-Watson test for autocorrelation to guard against acceptance of equations with this problem.

Data with nonnormal distributions were indicated by large values of skewness and kurtosis in the BMDP-9R program. This can become a problem because nonnormal distributions with heavy tails (high kurtosis) may contain several "outliers" produced by extreme events in the historical record. Detection and removal of outliers were accomplished where necessary by identifying the case(s) with maximum values for Cook's Distance, Mahalanobis Distance, and the standardized residual. The best indicator of candidate outliers for deletion is considered to be the Cook's Distance, because it measures the change in regression coefficients if an outlier case is omitted from the computation (Dr. David Hinkley, University of Texas at Austin; pers. comm. June 21, 1989). On the other hand, the Mahalanobis Distance is a measure of the outlier's distance from the mean centroid of all cases included in the analysis. The standardized residual simply indicates which case prediction deviates greatest from the observed case in the regression analysis. Candidate outliers were chosen using all three methods available in an effort to produce the best equation possible. After selection, the regression analysis was repeated with the outlier case(s) removed sequentially, usually producing an improved harvest equation. Although there is no established statistical rule on the number of deletions allowed, they never exceeded 10% of the cases in any of the fishery records analyzed here.

Fisheries analysis results. Statistically significant ($P < 0.05$) regression equations were developed for each of the seven coastal fishery species harvested in the Guadalupe Estuary, both as a function of gaged streamflows (Table 6.8.2) and as total combined freshwater inflows to the estuary (Table 6.8.3). For each equation, the number of cases (n = harvest years) in the regression analysis, the significance (P = probability) level, the correlation coefficient (R), the adjusted coefficient of determination (Adj. R^2), and the standard error of the harvest estimate (S.E. Est.) are given. In addition, the maximum value, minimum

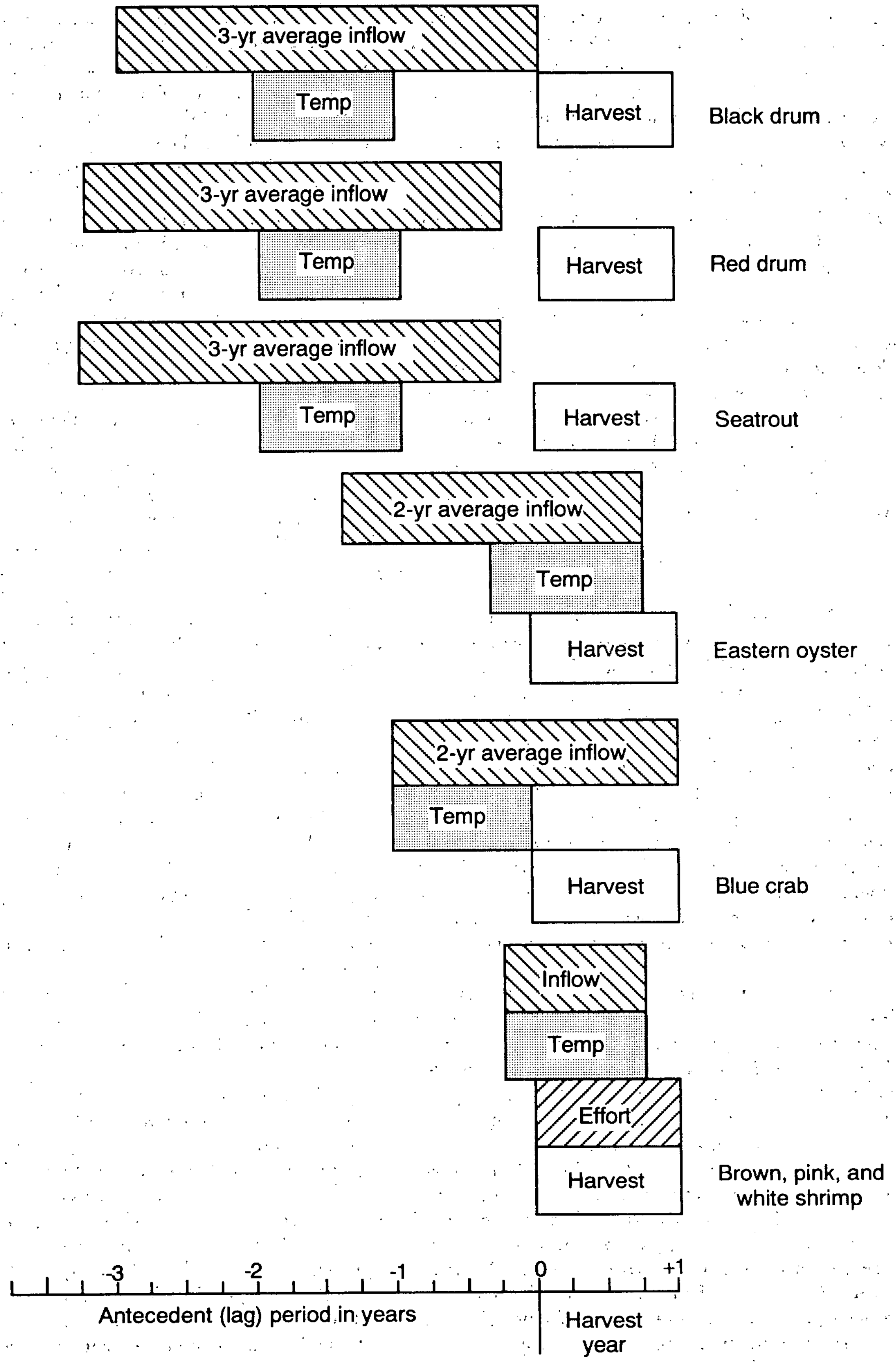


Figure 6.8.3. Alignment of independent and dependent variables in the fisheries analyses.

Table 6.8.2. Equations with statistics for the Guadalupe Estuary relating fishery harvests to seasonal freshwater inflows from only the gaged drainage basins. See table footnote for definitions of variables. The number of years of data (n), significance level (P), correlation coefficient (R), adjusted R^2, and standard error of the estimate (S.E. Est.) are given for each equation; the standard error of the coefficient (S.E. coeff.), and minimum, maximum, and mean values are presented for the dependent and independent variables.

Species or equation	Variable	S.E. coeff.	Maximum value	Minimum value	Mean value
White shrimp harvest = f (seasonal gaged flow, effort): ($n = 27$, $P < 0.0001$, $R = 0.90$, Adj. $R^2 = 77\%$, S.E. Est. = ± 125.4026)					
$H_{ws} = -134.384 - 205.012 (\ln Q_{ND}) + 196.837 (\ln Q_{JF}) + 73.0483 (\ln Q_{MJ}) + 0.1267 (E_i)$					
	H_{ws}	–	1415.0	241.8	737.5
	$\ln Q_{ND}$	57.3595	6.7182	4.5992	5.5104
	$\ln Q_{JF}$	67.5024	6.7987	4.4819	5.5444
	$\ln Q_{MJ}$	35.1373	7.8683	4.0146	5.9623
	E_i	0.0213	7208.4	1041.1	3748.0
Brown shrimp harvest = f (seasonal gaged flow, effort, temperature): ($n = 27$, $P < 0.0001$, $R = 0.95$, Adj. $R^2 = 89\%$, S.E. Est. = ± 0.3354)					
$\ln H_{bs} = 6.2116 + 0.1806 (\ln Q_{MJ}) + 0.000761 (E_i) - 0.0616 (T_{JF})$					
	$\ln H_{bs}$	–	7.5122	4.2062	5.9209
	$\ln Q_{MJ}$	0.0643	7.8683	4.0146	5.9003
	E_i	0.000054	4351.0	223.2	1916.9
	T_{JF}	0.0212	50.2	39.3	45.7
Blue crab harvest = f (seasonal gaged flow, temperature): ($n = 24$, $P < 0.0001$, $R = 0.92$, Adj. $R^2 = 78\%$, S.E. Est. = ± 0.3607)					
$\ln H_{bc} = 20.4508 - 1.1509 (\ln Q_{JF}) + 1.8123 (\ln Q_{JA}) - 1.0202 (\ln Q_{SO}) - 0.0746 (T_{JF}) + 0.2193 (T_{MJ}) - 0.2312 (T_{JA}) - 0.0988 (T_{ND})$					
	$\ln H_{bc}$	–	8.4713	5.6208	7.1854
	$\ln Q_{JF}$	0.2589	6.4102	4.8426	5.6291
	$\ln Q_{JA}$	0.2068	6.2277	3.9741	5.2651
	$\ln Q_{SO}$	0.1644	6.8078	4.4079	5.6261
	T_{JF}	0.0286	50.2	39.3	45.2
	T_{MJ}	0.0672	76.0	70.0	72.9
	T_{JA}	0.0612	80.5	74.8	77.8
	T_{ND}	0.0367	57.7	46.2	52.9

Table 6.8.2. (continued)

Species or equation	Variable	S.E. coeff.	Maximum value	Minimum value	Mean value
Bay oyster harvest = f (seasonal gaged flow, temperature): (n = 24, $P < 0.0001$, R = 0.95, Adj. R^2 = 88%, S.E. Est. = ± 0.3388)					
$\ln H_{bo} = 40.4359 - 1.4864 (\ln Q_{JF}) + 1.8920 (\ln Q_{MA}) - 1.2550 (\ln Q_{MJ}) + 0.0589 (T_{JF}) - 0.4108 (T_{JA})$					
	$\ln H_{bo}$	–	7.5690	3.9871	5.6230
	$\ln Q_{JF}$	0.2441	6.4102	4.9402	5.6938
	$\ln Q_{MA}$	0.2267	6.6678	4.8203	5.6822
	$\ln Q_{MJ}$	0.1492	7.3466	4.7194	6.2267
	T_{JF}	0.0241	50.2	39.3	45.6
	T_{JA}	0.0497	80.5	75.5	77.8
Black drum harvest = f (seasonal gaged flow, temperature): (n = 24, P = 0.0001, R = 0.86, Adj. R^2 = 66%, S.E. Est. = ± 20.2631)					
$H_{bd} = -1756.67 + 134.397 (\ln Q_{JF}) - 96.3520 (\ln Q_{MA}) - 19.9919 (\ln Q_{SO}) + 62.8709 (\ln Q_{ND}) + 17.1953 (T_{JA})$					
	H_{bd}	–	131.0	0.4	37.4
	$\ln Q_{JF}$	23.1272	6.2360	4.8873	5.6623
	$\ln Q_{MA}$	16.3997	6.5514	4.8956	5.6270
	$\ln Q_{SO}$	9.2753	6.5152	4.5706	5.7374
	$\ln Q_{ND}$	13.8773	6.3547	4.8363	5.6062
	T_{JA}	3.6559	80.5	74.8	77.8
Red drum harvest = f (seasonal gaged flow): (n = 19, P = 0.0001, R = 0.86, Adj. R^2 = 69%, S.E. Est. = ± 0.3379)					
$\ln H_{rd} = 2.1282 - 1.4714 (\ln Q_{ND}) + 0.7495 (\ln Q_{JF}) + 1.1481 (Q_{JA})$					
	$\ln H_{rd}$	–	5.1885	3.1946	4.1952
	$\ln Q_{ND}$	0.2868	6.3547	4.8363	5.6487
	$\ln Q_{JF}$	0.2859	6.2360	4.8873	5.7335
	$\ln Q_{JA}$	0.1866	6.1620	4.0792	5.2970

Table 6.8.2. (Concluded)

Species or equation	Variable	S.E. coeff.	Maximum value	Minimum value	Mean value
Spotted seatrout harvest = f (seasonal gaged flow): (n = 19, P = 0.0002, R = 0.81, Adj. R^2 = 61%, S.E. Est. = ± 0.4603)					
$\ln H_{ss} = 3.3498 - 1.2239 (\ln Q_{ND}) + 1.2079 (\ln Q_{MJ})$					
	$\ln H_{ss}$	–	4.7432	2.5014	3.8435
	$\ln Q_{ND}$	0.2756	6.3547	4.8363	5.6487
	$\ln Q_{MJ}$	0.2379	6.7992	4.6425	6.1321

H = commercial harvest of fishery species in thousands of pounds per year

ln H = natural logarithm of commercial fishery harvest

E_i = inshore harvest effort in number of fishing trips per year

Q = bimonthly gaged freshwater inflow in thousands of acre-feet per year
- $\ln Q_{JF}$ = natural logarithm of gaged inflow in January + February
- $\ln Q_{MA}$ = natural logarithm of gaged inflow in March + April
- $\ln Q_{MJ}$ = natural logarithm of gaged inflow in May + June
- $\ln Q_{JA}$ = natural logarithm of gaged inflow in July + August
- $\ln Q_{SO}$ = natural logarithm of gaged inflow in September + October
- $\ln Q_{ND}$ = natural logarithm of gaged inflow in November + December

T = bimonthly average minimum air temperature (°F)
- T_{JF} = average minimum temperature in January and February
- T_{MA} = average minimum temperature in March and April
- T_{MJ} = average minimum temperature in May and June
- T_{JA} = average minimum temperature in July and August
- T_{SO} = average minimum temperature in September and October
- T_{ND} = average minimum temperature in November and December

Table 6.8.3. Equations with statistics for the Guadalupe Estuary relating fishery harvests to seasonal combined freshwater inflows from all contributing river and coastal drainage basins. See table footnote for definitions of variables. The number of years of data (n), significance level (P), correlation coefficient (R), adjusted R^2, and standard error of the estimate (S.E. Est.) are given for each equation; the standard error of the coefficient (S.E. coeff.), and minimum, maximum, and mean values are presented for the dependent and independent variables.

Species or equation	Variable	S.E. coeff.	Maximum value	Minimum value	Mean value
White shrimp harvest = f (seasonal combined flow, effort): ($n = 27$, $P < 0.0001$, $R = 0.88$, Adj. $R^2 = 73\%$, S.E. Est. = ± 136.295) $H_{ws} = -233.556 - 180.422 (\ln Q_{ND}) + 174.333 (\ln Q_{JF}) + 83.5266 (\ln Q_{MJ}) + 0.1314 (E_i)$					
	H_{ws}	–	1415.0	241.8	737.5
	$\ln Q_{ND}$	62.3706	7.0137	4.7605	5.7277
	$\ln Q_{JF}$	70.3643	6.9224	4.6191	5.7229
	$\ln Q_{MJ}$	38.3592	7.9003	4.1043	6.1565
	E_i	0.0228	7208.4	1041.1	3748.0
Brown shrimp harvest = f (seasonal combined flow, effort, temperature): ($n = 26$, $P < 0.0001$, $R = 0.95$, Adj. $R^2 = 88\%$, S.E. Est. = ± 0.3372) $\ln H_{bs} = 5.9168 + 0.2126 (\ln Q_{MJ}) + 0.000793 (E_i) - 0.0613 (T_{JF})$					
	$\ln H_{bs}$	–	7.3262	4.2062	5.8597
	$\ln Q_{MJ}$	0.0736	7.2522	4.1043	6.0259
	E_i	0.000058	4351.0	223.2	1829.8
	T_{JF}	0.0216	50.2	39.3	45.6
Blue crab harvest = f (seasonal combined flow, temperature): ($n = 24$, $P = 0.0001$, $R = 0.83$, Adj. $R^2 = 62\%$, S.E. Est. = ± 0.4659) $\ln H_{bc} = 14.4130 - 0.9076 (\ln Q_{MA}) + 1.8840 (\ln Q_{JA}) - 1.1669 (\ln Q_{SO}) - 0.1182 (T_{JF})$					
	$\ln H_{bc}$	–	8.4713	5.6208	7.1977
	$\ln Q_{MA}$	0.3188	6.7554	4.8767	5.7914
	$\ln Q_{JA}$	0.3185	6.3852	4.0570	5.5324
	$\ln Q_{SO}$	0.2127	7.2007	4.6318	5.9907
	T_{JF}	0.0417	50.2	39.8	45.6

Table 6.8.3. (Concluded)

Bay oyster harvest = f (seasonal combined flow, temperature):

(n = 24, $P < 0.0001$, R = 0.93, Adj. R^2 = 83%, S.E. Est. = ± 0.3972)

$$\ln H_{bo} = 38.1475 - 1.2008 (\ln Q_{JF}) + 1.3544 (\ln Q_{MA}) - 1.0726 (\ln Q_{MJ}) - 0.3407 (T_{JA})$$

Species or equation	Variable	S.E. coeff.	Maximum value	Minimum value	Mean value
	$\ln H_{bo}$	-	7.5690	3.9871	5.6230
	$\ln Q_{JF}$	0.2799	6.5734	4.9945	5.8355
	$\ln Q_{MA}$	0.2189	6.7554	4.8767	5.7922
	$\ln Q_{MJ}$	0.1648	7.3872	4.9126	6.3830
	T_{JA}	0.0569	80.5	75.5	77.8

Black drum harvest = f (seasonal combined flow, temperature):

(n = 26, P = 0.0024, R = 0.73, Adj. R^2 = 44%, S.E. Est. = ± 27.1509)

$$H_{bd} = -947.813 + 58.9644 (\ln Q_{JF}) - 52.1974 (\ln Q_{MA}) + 60.5614 (\ln Q_{ND}) + 8.4824 (T_{SO})$$

Species or equation	Variable	S.E. coeff.	Maximum value	Minimum value	Mean value
	H_{bd}	-	131.0	0.4	40.5
	$\ln Q_{JF}$	20.2532	6.3704	5.0350	5.8390
	$\ln Q_{MA}$	17.1459	6.5891	4.9925	5.7603
	$\ln Q_{ND}$	15.5307	6.6502	5.0670	5.8410
	T_{SO}	2.8277	73.4	64.2	69.7

Red drum harvest = f (seasonal combined flow):

(n = 19, P = 0.0011, R = 0.76, Adj. R^2 = 52%, S.E. Est. = ± 0.4206)

$$\ln H_{rd} = 4.1138 - 1.2491 (\ln Q_{ND}) + 1.3577 (\ln Q_{JA})$$

Species or equation	Variable	S.E. coeff.	Maximum value	Minimum value	Mean value
	$\ln H_{rd}$	-	5.1885	3.1946	4.1952
	$\ln Q_{ND}$	0.3583	6.6502	5.0670	5.9462
	$\ln Q_{JA}$	0.2920	6.2800	4.3027	5.5307

Spotted seatrout harvest = f (seasonal combined flow):

(n = 19, P = 0.0003, R = 0.80, Adj. R^2 = 60%, S.E. Est. = ± 0.4687)

$$\ln H_{ss} = 2.5900 - 1.3655 (\ln Q_{ND}) + 1.4753 (\ln Q_{MJ})$$

Species or equation	Variable	S.E. coeff.	Maximum value	Minimum value	Mean value
	$\ln H_{ss}$	-	4.7432	2.5014	3.8435
	$\ln Q_{ND}$	0.3054	6.6502	5.0670	5.9462
	$\ln Q_{MJ}$	0.2910	6.9711	4.9537	6.3535

Table 6.8.3. (Concluded)

Species or equation	Variable	S.E. coeff.	Maximum value	Minimum value	Mean value

H = commercial harvest of fishery species in thousands of pounds per year
$\ln H$ = natural logarithm of commercial fishery harvest

E_i = inshore harvest effort in number of fishing trips per year

Q = bimonthly combined freshwater inflow in thousands of acre-feet per year
$\ln Q_{JF}$ = natural logarithm of combined inflow in January + February
$\ln Q_{MA}$ = natural logarithm of combined inflow in March + April
$\ln Q_{MJ}$ = natural logarithm of combined inflow in May + June
$\ln Q_{JA}$ = natural logarithm of combined inflow in July + August
$\ln Q_{SO}$ = natural logarithm of combined inflow in September + October
$\ln Q_{ND}$ = natural logarithm of combined inflow in November + December

T = bimonthly average minimum air temperature (°F)
T_{JF} = average minimum temperature in January and February
T_{MA} = average minimum temperature in March and April
T_{MJ} = average minimum temperature in May and June
T_{JA} = average minimum temperature in July and August
T_{SO} = average minimum temperature in September and October
T_{ND} = average minimum temperature in November and December

value, mean (average) value, and standard error of each regression coefficient in the resulting fishery equations are also presented in the tables.

To illustrate the performance of the equations in predicting fishery harvests over the historical period, observed harvests for each species can be plotted against the predicted harvests for both the gaged inflow equations (Figure 6.8.4) and the combined inflow equations (Figure 6.8.5). For each observation, the vertical distance from the line of perfect fit is a measure of the case's residual (unexplained) variation. When considered together, the results indicate that the regression models are adequate predictors of past fishery harvests in the Guadalupe Estuary.

Seafood harvests in other Texas bays and estuaries were similarly analyzed by species. For example, statistically significant spotted seatrout equations were additionally developed for the Trinity-San Jacinto, Lavaca-Colorado, Mission-Aransas, and Nueces estuaries, explaining from 64% to 90% of the year-to-year variations in harvest (Table 6.8.4). The predictive performances of the harvest equations from all five of the major estuarine systems between Galveston and Corpus Christi, Texas, are also shown in Figure 6.8.6 for comparison.

The regression analyses also produced statistically significant equations for the harvest of blue crabs (Table 6.8.5) and white shrimp (Table 6.8.6) from each of the other estuaries analyzed. The predictive performances of the harvest equations from all five of the estuaries analyzed are reasonable, as indicated respectively for these species by figures 6.8.7 and 6.8.8, and they explain 48% to 82% of the year-to-year harvest fluctuations in these inshore areas.

In a further effort to investigate the effects of freshwater inflows, the offshore fisheries harvests of brown shrimp and white shrimp in the designated federal fishing zones were additionally analyzed. The analysis of brown shrimp harvests from Zone 18 offshore of the Trinity-San Jacinto Estuary, and the combined harvests from Zones 18 through 20 offshore between Galveston and Corpus Christi, Texas, produced statistically significant equations that explain 57% and 55% of the respective harvest variations (Table 6.8.7). Likewise, the white shrimp equation for Zone 18 accounts for 53% of historical harvest fluctuations (Table 6.8.7). The predictive performances of the offshore shrimp equations are acceptable and are graphically depicted in Figure 6.8.9.

The purpose of these analyses was to investigate the relationships between freshwater inflows to an estuary and its fishery production. The addition of seasonal temperature and fishing effort (shrimp fishery only) to improve the regressions shows that other variables not related to inflow can influence the amount of fish and shellfish harvested. In general, solutions to the harvest equations are rather sensitive to annual variations of fishing effort (where the data was available). The solutions are much less sensitive to normal variations in the seasonal average minimum temperatures. This does not mean that extreme temperature events, such as the relatively rare freezing over of the bays during the passage of strong Arctic fronts, cannot substantially affect fisheries mortality, and subsequently the harvest of most fisheries species. The one exception may be the eastern oyster, which is considered to have evolved as a cold-water mollusk, and seems to relate negatively to high temperatures in the spring and summer, rather than cold temperatures in the fall and winter.

Summary

The historical commercial harvests of seven coastal fishery species (red drum, black drum, spotted seatrout, blue crab, bay oyster, and white and brown shrimp) were analyzed as a function of fishing effort, air temperature, and freshwater inflow to the Guadalupe Estuary using the "All Possible Subsets Regression" statistical program BMDP-9R. Interestuary comparisons of spotted seatrout, blue crab, and white shrimp harvest equations from the five major bay and estuary systems between Galveston and Corpus Christi, Texas, (Trinity-San Jacinto, Lavaca-Colorado, Guadalupe, Mission-Aransas, and Nueces estuaries) show similarities, unique differences, and competitive responses to freshwater inflow among the coastal fishery species. For example, the spotted seatrout and white shrimp exhibit several positive correlations to springtime (May-June) inflows, while the blue crab often relates negatively to higher inflows during this season. The regression analyses produced 26 inshore harvest equations and three offshore harvest equations of statistical significance, which accounted for 44% to 90% (average = 69%) of the year-to-year fluctuations in seafood harvest. Furthermore, the harvest equations appear adequate for freshwater inflow management purposes, particularly if they are used together with salinity tolerance information in a balanced approach that does not depend exclusively on any one species.

6.9 CONCLUSIONS

Larval transport. Evidence presented in Section 6.1 indicated that eggs and larvae of estuarine organisms getting into estuaries from the Gulf of Mexico do not appear to be using a salinity gradient to locate the estuary. There was little evidence that these organisms used mechanisms such as swimming into areas with high landward currents or evading areas with high seaward currents to improve their chances of moving into and remaining in the estuary. This study

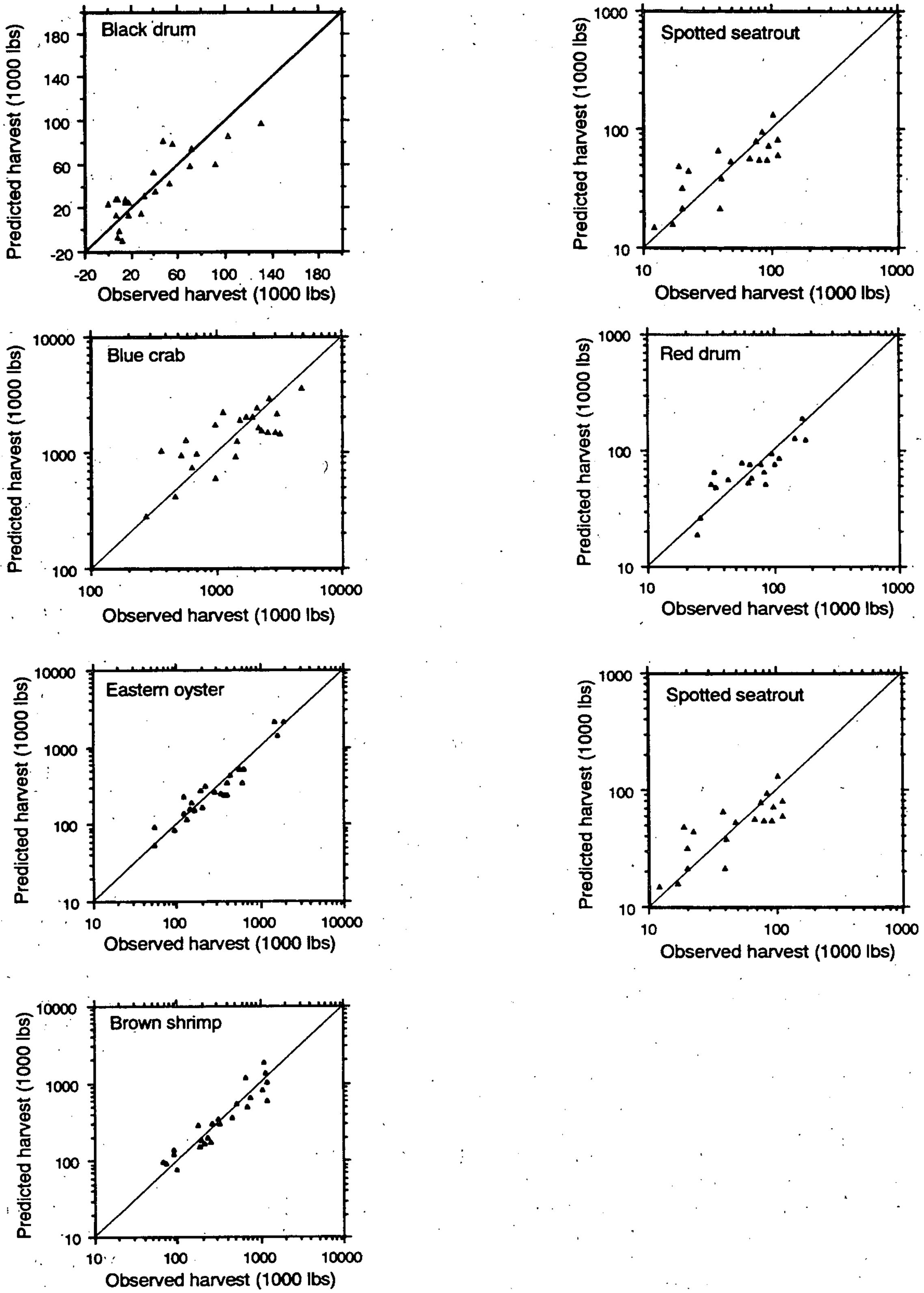


Figure 6.8.4. Predicted versus observed harvests of fishery species as a function of gaged freshwater inflows to the Guadalupe Estuary.

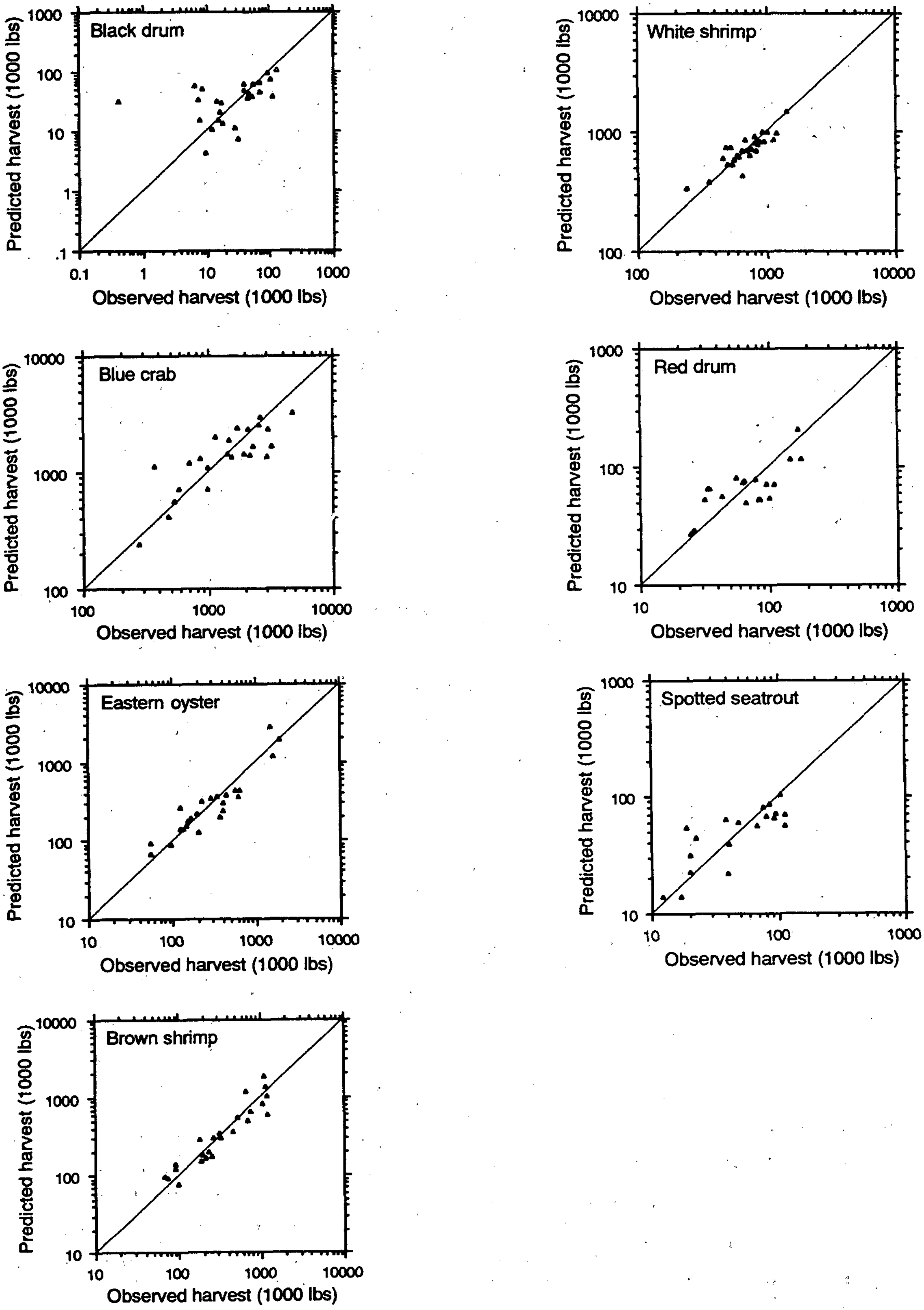


Figure 6.8.5. Predicted versus observed harvests of fishery species as a function of combined freshwater inflows to the Guadalupe Estuary.

Table 6.8.4. Equations with statistics for spotted seatrout harvest from four bays along the Texas Coast relating fishery harvests to combined freshwater inflows from all contributing river and coastal drainage basins. See table footnote for definitions of variables. The number of years of data (n), significance level (P), correlation coefficient (R), adjusted R^2, and standard error of the estimate (S.E. Est.) are given for each equation; the standard error of the coefficient (S.E. coeff.), and minimum, maximum, and mean values are presented for the dependent and independent variables.

Estuary or equation	Variable	S.E. coeff.	Maximum value	Minimum value	Mean value
Trinity-San Jacinto Estuary spotted seatrout harvest = f (seasonal combined flow): ($n = 18$, $P = 0.0002$, $R = 0.83$, Adj. $R^2 = 64\%$, S.E. Est. = ± 0.3272)					
$\ln H_{ss} = 4.9615 + 1.3774 (\ln Q_{ND}) - 1.3114 (\ln Q_{JF})$					
	$\ln H_{ss}$	—	5.8412	4.0200	5.0464
	$\ln Q_{ND}$	0.2514	7.9716	6.4218	7.1361
	$\ln Q_{JF}$	0.2621	8.0534	6.7451	7.4302
Lavaca-Colorado Estuary spotted seatrout harvest = f (seasonal combined flow): ($n = 18$, $P < 0.0001$, $R = 0.92$, Adj. $R^2 = 80\%$, S.E. Est. = ± 0.2771)					
$\ln H_{ss} = 8.8568 - 0.8683 (\ln Q_{ND}) - 0.5370 (\ln Q_{MA}) + 0.6047 (\ln Q_{MJ})$					
	$\ln H_{ss}$	—	5.6157	3.4243	4.5832
	$\ln Q_{ND}$	0.1178	6.7751	4.7519	5.9142
	$\ln Q_{MA}$	0.1791	6.8005	4.6625	5.9225
	$\ln Q_{MJ}$	0.1751	7.3698	5.0179	6.6851
Mission-Aransas Estuary spotted seatrout harvest = f (seasonal combined flow): ($n = 19$, $P = 0.0001$, $R = 0.90$, Adj. $R^2 = 75\%$, S.E. Est. = ± 0.2618)					
$\ln H_{ss} = 6.8626 + 0.0948 (\ln Q_{SO}) + 0.2173 (\ln Q_{ND}) - 0.1668 (\ln Q_{JF}) - 0.6708 (\ln Q_{JA})$					
	$\ln H_{ss}$	—	5.8880	3.7111	5.0861
	$\ln Q_{SO}$	0.0423	6.3269	1.7750	4.7977
	$\ln Q_{ND}$	0.0700	4.7423	1.6094	3.2243
	$\ln Q_{JF}$	0.0694	4.4462	0.8329	3.0266
	$\ln Q_{JA}$	0.0991	4.9822	2.6462	3.6179
Nueces Estuary spotted seatrout harvest = f (seasonal combined flow): ($n = 20$, $P = < 0.0001$, $R = 0.96$, Adj. $R^2 = 90\%$, S.E. Est. = ± 0.3017)					
$\ln H_{ss} = 1.6865 + 0.3863 (\ln Q_{SO}) - 1.1369 (\ln Q_{JF}) + 0.6850 (\ln Q_{MA}) + 0.4586 (\ln Q_{MJ})$					
	$\ln H_{ss}$	—	5.2591	2.4849	4.0682
	$\ln Q_{SO}$	0.0963	6.7049	2.2925	5.1561
	$\ln Q_{JF}$	0.1276	4.5570	1.7048	3.3788
	$\ln Q_{MA}$	0.1153	4.8024	1.8871	3.0792
	$\ln Q_{MJ}$	0.1285	5.3904	2.0412	4.6263

H = commercial harvest of fishery species in thousands of pounds per year
$\ln H$ = natural logarithm of commercial fishery harvest

E_i = inshore harvest effort in number of fishing trips per year

Q = bimonthly gaged freshwater inflow in thousands of acre-feet per year
$\ln Q_{JF}$ = natural logarithm of gaged inflow in January + February
$\ln Q_{MA}$ = natural logarithm of gaged inflow in March + April
$\ln Q_{MJ}$ = natural logarithm of gaged inflow in May + June
$\ln Q_{JA}$ = natural logarithm of gaged inflow in July + August
$\ln Q_{SO}$ = natural logarithm of gaged inflow in September + October
$\ln Q_{ND}$ = natural logarithm of gaged inflow in November + December

T = bimonthly average minimum air temperature (°F)
T_{JF} = average minimum temperature in January and February
T_{MA} = average minimum temperature in March and April
T_{MJ} = average minimum temperature in May and June
T_{JA} = average minimum temperature in July and August
T_{SO} = average minimum temperature in September and October
T_{ND} = average minimum temperature in November and December

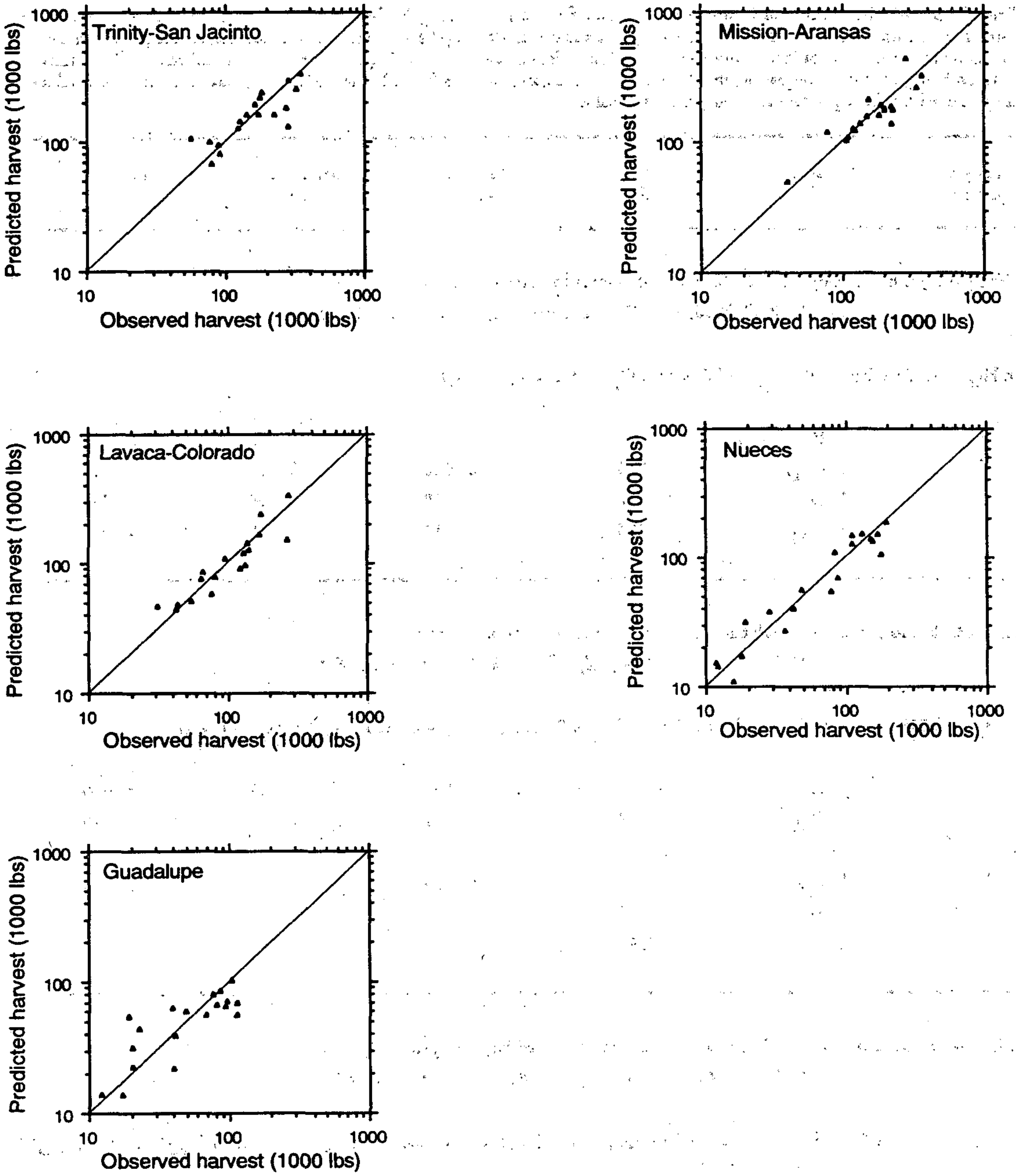


Figure 6.8.6. Predicted versus observed harvests of spotted seatrout as a function of combined freshwater inflows to each of five Texas estuaries.

Table 6.8.5. Equations with statistics for blue crab harvest from four bays along the Texas Coast relating fishery harvests to seasonal combined freshwater inflows from all contributing river and coastal drainage basins. See table footnote for definitions of variables. The number of years of data (n), significance level (P), correlation coefficient (R), adjusted R^2, and standard error of the estimate (S.E. Est.) are given for each equation; the standard error of the coefficient (S.E. coeff.), and minimum, maximum, and mean values are presented for the dependent and independent variables.

Trinity-San Jacinto Estuary blue crab harvest = f (seasonal combined flow):

($n = 24$, $P = 0.0006$, $R = 0.75$, Adj. $R^2 = 50\%$, S.E. Est. $= \pm 0.2028$)

$$\ln H_{bc} = 4.3837 + 0.3717 (\ln Q_{MA}) - 0.2605 (\ln Q_{JA}) + 0.2786 (\ln Q_{ND})$$

Estuary or equation	Variable	S.E. coeff.	Maximum value	Minimum value	Mean value
	$\ln H_{bc}$	—	8.0124	6.8850	7.4578
	$\ln Q_{MA}$	0.0878	8.3040	6.2214	7.4825
	$\ln Q_{JA}$	0.0816	7.8538	5.3404	6.6781
	$\ln Q_{ND}$	0.0774	8.4117	6.1970	7.2968

Lavaca-Colorado Estuary blue crab harvest = (seasonal combined flow, temperature):

($n = 25$, $P = 0.0001$, $R = 0.86$, Adj. $R^2 = 67\%$, S.E. Est. $= \pm 0.2968$)

$$\ln H_{bc} = -9.2196 + 0.2403 (\ln Q_{MA}) - 0.2916 (\ln Q_{MJ}) + 0.6586 (\ln Q_{JA}) + 0.1215 (T_{MA}) + 0.0784 (T_{SO})$$

Estuary or equation	Variable	S.E. coeff.	Maximum value	Minimum value	Mean value
	$\ln H_{bc}$	—	7.6043	5.4201	6.6440
	$\ln Q_{MA}$	0.0959	6.9612	4.4931	5.8279
	$\ln Q_{MJ}$	0.1121	7.4461	4.8590	6.5631
	$\ln Q_{JA}$	0.1453	6.3059	4.8211	5.6189
	T_{MA}	0.0239	64.5	50.0	59.4
	T_{SO}	0.0296	73.4	64.2	69.7

Mission-Aransas Estuary blue crab harvest = f (seasonal combined flow, temperature):

($n = 26$, $P < 0.0001$, $R = 0.93$, Adj. $R^2 = 82\%$, S.E. Est. $= \pm 0.5835$)

$$\ln H_{bc} = -8.8574 + 0.5386 (\ln Q_{JF}) - 0.2829 (\ln Q_{MJ}) + 0.3834 (\ln Q_{SO}) + 0.6079 (\ln Q_{ND}) + 0.2666 (T_{SO}) - 0.1327 (T_{ND})$$

Estuary or equation	Variable	S.E. coeff.	Maximum value	Minimum value	Mean value
	$\ln H_{bc}$	—	7.9070	2.9601	6.6390
	$\ln Q_{JF}$	0.1044	4.7140	0.1823	3.1562
	$\ln Q_{MJ}$	0.1294	6.0144	1.9021	4.4792
	$\ln Q_{SO}$	0.0894	6.6209	1.6094	4.5628
	$\ln Q_{ND}$	0.1223	4.8652	1.6292	3.2569
	T_{SO}	0.0602	71.9	64.6	68.4
	T_{ND}	0.0455	58.0	47.3	52.0

Table 6.8.5. (concluded)

Estuary or equation	Variable	S.E. coeff.	Maximum value	Minimum value	Mean value
Nueces Estuary blue crab harvest = f (seasonal combined flow, temperature): (n = 21, P = 0.0020, R = 0.88, Adj. R^2 = 66%, S.E. Est. = ± 0.6524)					
$\ln H_{bc} = -7.8143 + 0.5423 (\ln Q_{JF}) - 0.5287 (\ln Q_{MA}) - 0.6442 (\ln Q_{MJ}) + 0.5502 (\ln Q_{ND}) - 0.2451 (T_{JF}) + 0.2595 (T_{MA}) + 0.1763 (T_{ND})$					
	$\ln H_{bc}$	—	6.7767	2.0412	4.4490
	$\ln Q_{JF}$	0.2328	4.9016	1.8563	3.1804
	$\ln Q_{MA}$	0.2062	5.0304	1.8871	2.9310
	$\ln Q_{MJ}$	0.1814	6.0811	2.1400	4.7249
	$\ln Q_{ND}$	0.1477	5.2893	1.3610	3.3571
	T_{JF}	0.0697	51.1	41.1	46.4
	T_{MA}	0.0706	64.2	55.4	59.8
	T_{ND}	0.0606	56.5	47.3	51.9

H = commercial harvest of fishery species in thousands of pounds per year
$\ln H$ = natural logarithm of commercial fishery harvest

E_i = inshore harvest effort in number of fishing trips per year

Q = bimonthly gaged freshwater inflow in thousands of acre-feet per year
$\ln Q_{JF}$ = natural logarithm of gaged inflow in January + February
$\ln Q_{MA}$ = natural logarithm of gaged inflow in March + April
$\ln Q_{MJ}$ = natural logarithm of gaged inflow in May + June
$\ln Q_{JA}$ = natural logarithm of gaged inflow in July + August
$\ln Q_{SO}$ = natural logarithm of gaged inflow in September + October
$\ln Q_{ND}$ = natural logarithm of gaged inflow in November + December

T = bimonthly average minimum air temperature (°F)
T_{JF} = average minimum temperature in January and February
T_{MA} = average minimum temperature in March and April
T_{MJ} = average minimum temperature in May and June
T_{JA} = average minimum temperature in July and August
T_{SO} = average minimum temperature in September and October
T_{ND} = average minimum temperature in November and December

Table 6.8.6 Equations with statistics for white shrimp harvest from 4 bays along the Texas Coast relating fishery harvests to combined freshwater inflows from all contributing river and coastal drainage basins. See table footnote for definitions of variables. The number of years of data (n), significance level (P), correlation coefficient (R), adjusted R^2, and standard error of the estimate (S.E. Est.) are given for each equation; the standard error of the coefficient (S.E. Coeff.), and minimum, maximum, and mean values are presented for the dependent and independent variables.

Estuary or equation	Variable	S.E. Coeff.	Maximum value	Minimum value	Mean value
Trinity-San Jacinto Estuary white shrimp harvest = f (seasonal combined flow, temperature): (n = 27, P = .0001, R = 0.87, Adj. R^2 = 67%, S.E. Est. = ± 583.551)					
$H_{ws} = -54267.4 - 487.560 (\ln Q_{JF}) + 907.769 (\ln Q_{MA}) - 486.397 (\ln Q_{JA}) + 302.409 (T_{SO}) - 115.176 (T_{ND}) + 177.249 (T_{JF}) + 423.253 (T_{JA})$					
	H_{ws}	—	4700.7	982.0	2857.0
	$\ln Q_{JF}$	173.422	8.5526	4.9001	7.3243
	$\ln Q_{MA}$	224.066	8.6771	5.7242	7.2652
	$\ln Q_{JA}$	164.229	8.1936	5.3073	6.7078
	T_{SO}	77.825	74.0	67.3	71.7
	T_{ND}	46.310	61.1	48.6	54.7
	T_{JF}	47.778	53.0	43.4	48.3
	T_{JA}	122.377	80.7	77.1	79.0
Lavaca-Colorado Estuary white shrimp harvest = f (seasonal combined flow, effort, temperature): (n = 27, P < 0.0001, R = 0.92, Adj. R^2 = 81%, S.E. Est. = ± 0.1837)					
$\ln H_{ws} = 5.7174 - 0.1267 (\ln Q_{SO}) + 0.1123 (\ln Q_{MJ}) + .00014 (E_i) + 0.0172 (T_{JF})$					
	$\ln H_{ws}$	—	8.1881	6.8249	7.5080
	$\ln Q_{SO}$	0.0434	7.5784	4.4474	6.2428
	$\ln Q_{MJ}$	0.0366	7.7039	4.2767	6.3515
	E_i	0.000015	12693.1	4313.7	7667.1
T_{JF}	0.0134	50.2	39.3	45.7	
Mission-Aransas Estuary white shrimp harvest = f (seasonal combined flow, effort, temperature): (n = 27, P < 0.0001, R = 0.88, Adj. R^2 = 73%, S.E. Est. = ± 0.3065)					
$\ln H_{ws} = 8.3808 + 0.1175 (\ln Q_{SO}) + 0.1073 (\ln Q_{MA}) + 0.000208 (E_i) - 0.0651 (T_{ND})$					
	$\ln H_{ws}$	—	7.4873	5.5211	6.5615
	$\ln Q_{SO}$	0.0340	7.2890	1.0647	4.2313
	$\ln Q_{MA}$	0.0497	4.6260	0.5878	2.4141
	E_i	0.000031	8731.0	1132.8	3820.4
	T_{ND}	0.0239	58.0	47.3	51.7
Nueces Estuary white shrimp harvest = f (seasonal combined flow, temperature): (n = 26, P = 0.0006, R = 0.74, Adj. R^2 = 48%, S.E. Est. = ± 0.4349)					
$\ln H_{ws} = 24.7974 + 0.1640 (\ln Q_{ND}) + 0.1962 (\ln Q_{MJ}) - 0.2837 (T_{MJ})$					
	$\ln H_{ws}$	—	6.8874	4.4018	5.8148
	$\ln Q_{ND}$	0.0691	5.9623	1.3083	3.0304
	$\ln Q_{MJ}$	0.0592	6.5822	1.8406	4.1303
	T_{MJ}	0.0831	73.6	69.3	71.5

H = commercial harvest of fishery species in thousands of pounds per year
$\ln H$ = natural logarithm of commercial fishery harvest

E = inshore harvest effort in number of fishing trips per year

Q = bimonthly gaged freshwater inflow in thousands of acre-feet per year
$\ln Q_{JF}$ = natural logarithm of gaged inflow in January + February
$\ln Q_{MA}$ = natural logarithm of gaged inflow in March + April
$\ln Q_{MJ}$ = natural logarithm of gaged inflow in May + June
$\ln Q_{JA}$ = natural logarithm of gaged inflow in July + August
$\ln Q_{SO}$ = natural logarithm of gaged inflow in September + October
$\ln Q_{ND}$ = natural logarithm of gaged inflow in November + December

T = bimonthly average minimum air temperature (°F)
T_{JF} = average minimum temperature in January and February
T_{MA} = average minimum temperature in March and April
T_{MJ} = average minimum temperature in May and June
T_{JA} = average minimum temperature in July and August
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T_{ND} = average minimum temperature in November and December

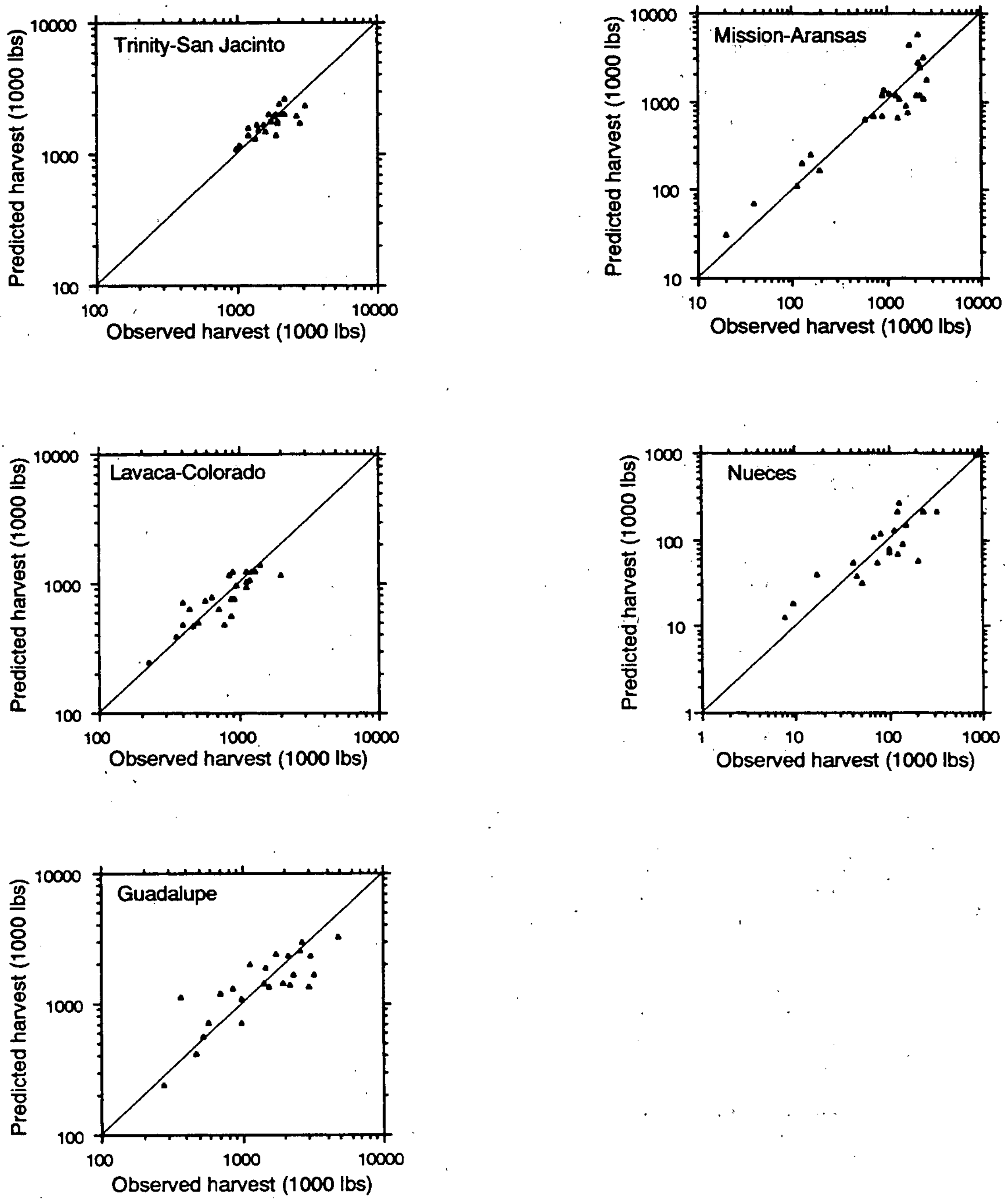


Figure 6.8.7. Predicted versus observed harvests of blue crab as a function of combined freshwater inflows to each of five Texas estuaries.

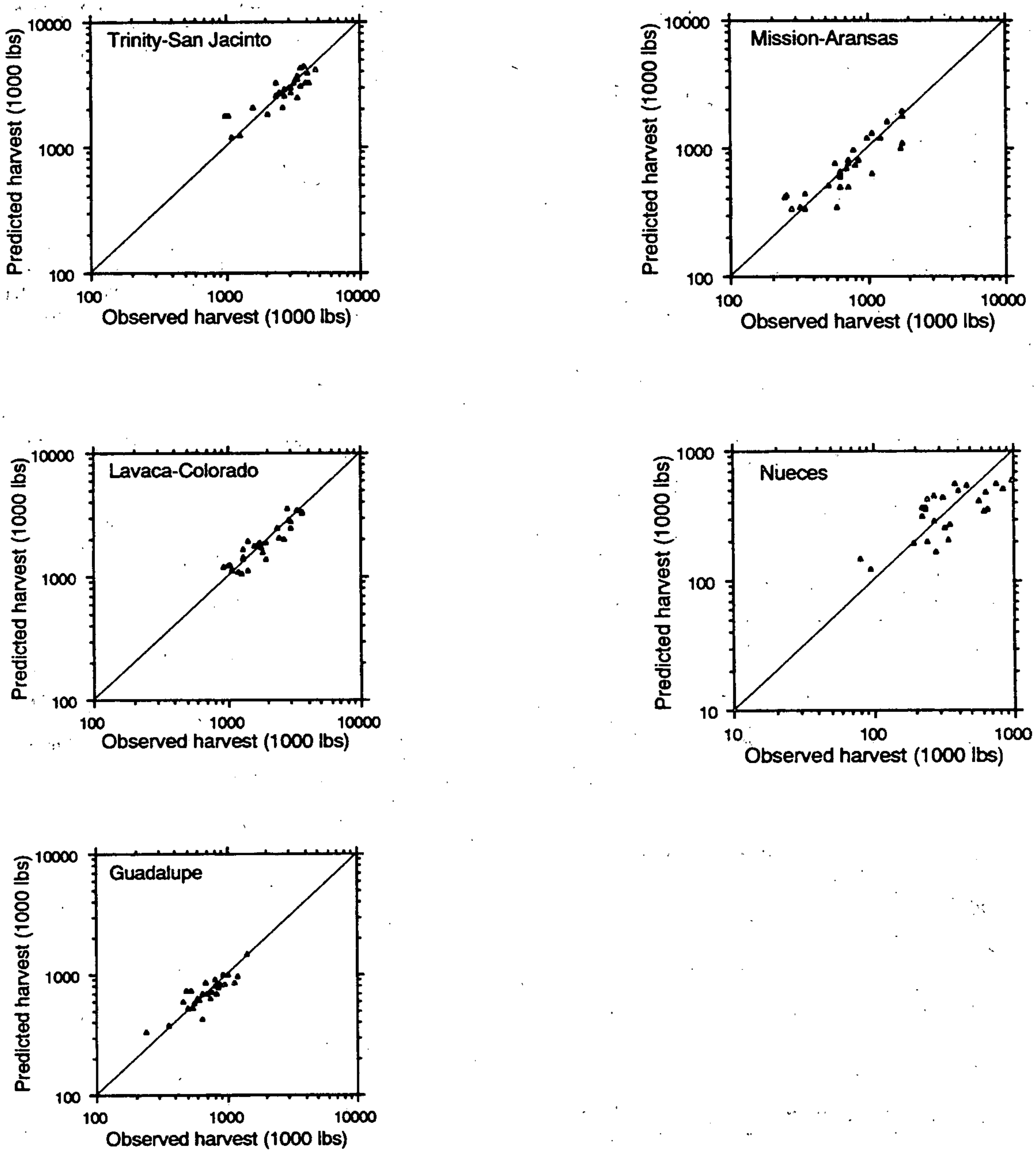


Figure 6.8.8. Predicted versus observed harvests of white shrimp as a function of combined freshwater inflows to each of five Texas estuaries.

Table 6.8.7. Equations with statistics for offshore shrimp harvest relating fishery harvests to seasonal combined freshwater inflows from all contributing river and coastal drainage basins. See table footnote for definitions of variables. The number of years of data (n), significance level (P), correlation coefficient (R), adjusted R^2, and standard error of the estimate (S.E. Est.) are given for each equation; the standard error of the coefficient (S.E. coeff.), and minimum, maximum, and mean values are presented for the dependent and independent variables.

Estuary or equation	Variable	S.E. coeff.	Maximum value	Minimum value	Mean value
Zone 18-19-20 offshore brown shrimp harvest = f (seasonal combined flow, effort): ($n = 27$, $P = 0.0001$, $R = 0.80$, Adj. $R^2 = 57\%$, S.E. Est. = ± 0.1854)					
$\ln H_{bs} = 18.6497 - 0.1618 (\ln Q_{ND}) - 0.3237 (\ln Q_{MA}) + 0.2432 (\ln Q_{MJ}) + 0.0000229 (E_o)$					
	$\ln H_{bs}$	–	17.9549	16.8220	17.3242
	$\ln Q_{ND}$	0.0534	8.8887	6.1823	7.5968
	$\ln Q_{MA}$	0.0659	8.9315	6.1711	7.5873
	$\ln Q_{MJ}$	0.0628	9.3118	6.1319	8.1631
	E_o	0.0000086	27946.0	9233.0	16353.5
Zone 18 offshore brown shrimp harvest = f (seasonal combined flow): ($n = 26$, $P < 0.0001$, $R = 0.77$, Adj. $R^2 = 55\%$, S.E. Est. = ± 0.3237)					
$\ln H_{bs} = 16.4879 + 0.2638 (\ln Q_{SO}) - 0.3454 (\ln Q_{MA})$					
	$\ln H_{bs}$	–	16.6913	14.9101	15.7702
	$\ln Q_{SO}$	0.0760	8.3481	4.8426	6.7966
	$\ln Q_{MA}$	0.0802	8.6771	5.7242	7.2687
Zone 18 offshore white shrimp harvest = f (seasonal combined flow, effort, temperature): ($n = 26$, $P = 0.0017$, $R = 0.80$, Adj. $R^2 = 53\%$, S.E. Est. = ± 705.128)					
$H_{ws} = -75189.3 - 607.299 (\ln Q_{JF}) + 652.126 (\ln Q_{MA}) + 586.796 (\ln Q_{JA}) + 0.3281 (E_o) + 335.108 (T_{MA}) + 661.112 (T_{JA})$					
	H_{ws}	–	4894.5	1372.6	3175.3
	$\ln Q_{JF}$	215.097	8.5626	4.9001	7.3894
	$\ln Q_{MA}$	245.292	8.6771	5.7242	7.3346
	$\ln Q_{JA}$	200.457	8.1936	5.3073	6.7231
	E_o	0.1073	8149.2	2159.0	4589.4
	T_{MA}	80.501	64.5	58.0	60.6
	T_{JA}	164.512	80.7	77.2	79.1

H = commercial harvest of fishery species in thousands of pounds per year
ln H = natural logarithm of commercial fishery harvest

E_o = offshore harvest effort in number of fishing trips per year

Q = bimonthly combined freshwater inflow in thousands of acre-feet per year
$\ln Q_{JF}$ = natural logarithm of combined inflow in January + February
$\ln Q_{MA}$ = natural logarithm of combined inflow in March + April
$\ln Q_{MJ}$ = natural logarithm of combined inflow in May + June
$\ln Q_{JA}$ = natural logarithm of combined inflow in July + August
$\ln Q_{SO}$ = natural logarithm of combined inflow in September + October
$\ln Q_{ND}$ = natural logarithm of combined inflow in November + December

T = bimonthly average minimum air temperature (°F)
T_{JF} = average minimum temperature in January and February
T_{MA} = average minimum temperature in March and April
T_{MJ} = average minimum temperature in May and June
T_{JA} = average minimum temperature in July and August
T_{SO} = average minimum temperature in September and October
T_{ND} = average minimum temperature in November and December

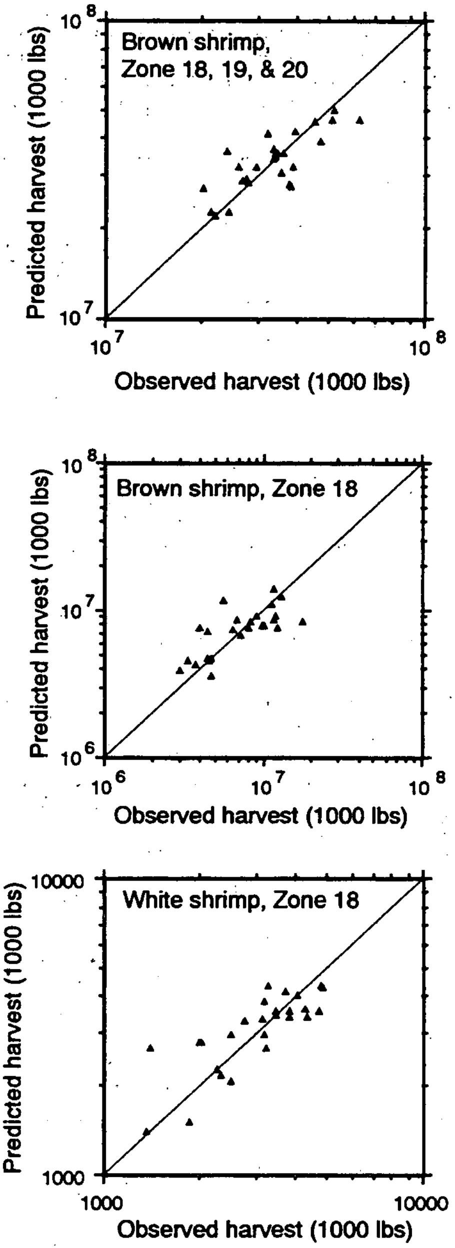


Figure 6.8.9. Predicted versus observed harvests of brown and white shrimp in federal fishing zone 18 and combined zones 18, 19, and 20 as a function of combined freshwater inflows to the associated Texas bays and estuaries.

indicated that tidal currents, on-shore winds, and tide type (spring or neap) were mainly responsible for transport of planktonic organisms into the estuaries. This would explain why many estuarine organisms spawned in the Gulf are found in all Texas estuaries regardless of the widely differing salinity conditions found in each (Mambretti et al. 1990).

Effects of freshwater inflows on juvenile fish and shellfish. Many estuarine organisms live under slightly stressed conditions to take advantage of nutrients supplied by freshwater inflows or to escape predators and disease. Section 6.2 compared the results of several field and laboratory studies and found that higher densities of juvenile shellfish animals occurred in areas that had salinities slightly below those considered optimal for growth and survival. Blue crabs were most abundant in trawl catches when mean salinities ranged from 6 to 15‰, but in laboratory studies they grew fastest when salinities ranged from 10 to 20‰ (Leffler 1972).

Available information on oysters indicated they grow best when temperatures ranged from 20 to 30 °C (68 to 86 °F) and salinities range from 10 to 30‰. In the Guadalupe Estuary, the largest reefs are found in waters where salinities commonly range from 10 to 25‰. Spat set seems to be favored by warmer waters (27 to 32 °C or 31 to 90 °F) and intermediate salinities (10 to 28‰). There is some evidence that salinity may affect whether spat settles subtidally (less than 20‰) or intertidally (greater than 20‰).

Section 6.2 also reported evidence that for some species, preferences or tolerances of different salinity conditions change with size or maturity. Small white shrimp less than 50 mm TL caught in bag seines showed no preference for a specific salinity range. However, larger white shrimp caught in bag seines showed a significantly higher catch rate for samples taken from areas having salinities less than 24 to 26‰. Larger trawl-caught white shrimp had higher catch rates from areas with salinities less than 15‰. Brown shrimp less than 50 mm TL were most abundant when mean salinities ranged from 24 to 36‰. There was no clear trend between density and salinity for larger brown shrimp (51 to 100 mm TL) caught near shore in bag seines, indicating that the shrimp became more widely distributed. However, larger brown shrimp caught in trawls were most abundant in salinities ranging from 10 to 25‰.

Habitat use. Studies reviewed in Section 6.3 showed that many species move into inundated marsh areas and probably feed directly on detritus, benthic species, and epiphytes. This means that estuarine organisms actively transport nutrients out of the marsh during any high-water event and are not totally dependent on major floods to flush materials out of the marsh for them to use. The studies also corroborated findings from Section 6.2 that many estuarine species are most abundant in areas nearer the lower to middle range of their salinity tolerances. In addition, juvenile brown and white shrimp and several other animals had higher abundances in vegetated areas (emergent and submerged) compared with unvegetated areas.

Salinity effects on metabolism. Section 6.4 summarized the results of several physiological studies that measured the metabolic scope (i.e., the difference between resting and maximum sustained swimming speed) for several fish species to determine optimum salinities for efficient metabolism. The results of these studies showed that juvenile (fingerling) fish were generally more euryhaline than adult fish and consequently had a higher proportion of their energy available for growth. Adult spotted seatrout and red drum have maximum metabolic scopes around 20‰, with a shift to 25‰ in colder temperatures. Maximum swimming speeds for these two species also occurred at these optimum salinities. Several other species (e.g., sheepshead, black drum, sand seatrout, and Atlantic croaker) also had optimum swimming speeds at salinities of less than 30‰ when the temperature was 28 °C.

Salinity effects on reproduction. Section 6.5 presented results of reproductive investigations for Atlantic croaker, red drum, and spotted seatrout. These three species spawn in different areas: spotted seatrout spawn in the estuary, red drum spawn near Gulf passes, and Atlantic croaker spawn farther offshore. Estuarine salinities affect all reproductive stages of these three species because the subadult or adult life stages occur in the estuary and their eggs or larvae appear in the estuaries just a few days after spawning.

Salinity levels may influence the gonadal development and endocrine functions of estuarine fish, as shown by studies of spotted seatrout and Atlantic croaker. Seatrout, which spawn in the estuary, were more sensitive in this regard than croaker, which spawn offshore. Seatrout gonadal development was best at 35‰ and was suppressed more by salinities of less than 20‰ than by salinities of more than 45‰.

Egg fertilization and survival through hatching was high for all three species in the salinity range 25 to 35‰. These conditions are usually met since these species either spawning in the Gulf (red drum and Atlantic croaker) or in the lagunal or near-Gulf areas of estuaries where salinities are generally near 30‰. Of these, spotted seatrout would probably encounter the most problems since they spawn in the estuary. The high variability in estuarine salinities may account for the protracted multiple spawning behavior evolving in this species. It may also account for spotted seatrout having high survival rates from spawning to hatching over a wide salinity range (10 to 35‰).

Spotted seatrout larvae had the widest salinity range in which substantial larval survival was maintained for up to nine days. The survival range was narrower for red drum and even more restricted for Atlantic croaker. All three species showed differences in daily tolerance limits associated with development age. The basic conclusion from these studies is that as long as salinities in a portion of the estuary are between 15 and 35‰, there should not be any major adverse effects on any aspect of the reproduction of these species.

Effects of inflow and salinity on adult fish and shellfish abundance. Evidence from the literature presented in Section 6.6 showed that adult or subadult finfish living in estuaries are not generally sensitive to salinity concentrations or inflows. Adult brown shrimp are not affected by inflows since they are not found in the estuaries as adults; they move to the Gulf as subadults prior to spawning. Adult white shrimp apparently move in and out of the estuary, and as adults, can move to the Gulf if conditions become unbearable. The literature reports a wide salinity tolerance range for many adult species (5 to more than 45‰). Most fish can move relatively large distances, and if adverse conditions persist, they can move to areas where conditions are more favorable. Obviously, less mobile organisms must deal with adverse conditions when they occur; adult oysters and other pelecypods are prime examples. If adverse conditions persist for too long, the organisms will be stressed and eventually die.

Abundance of fish and shellfish among estuaries. Section 6.7 compared the mean catch rates from gill net and otter trawl samples among seven estuaries and investigated the significance of regressions between mean catch rates and mean salinities among the estuaries. No significant relationships were found for species caught with gill nets, but several species caught with trawls did show significant relationships. Both of these gear types caught southern flounder and Gulf menhaden, but showed different results. Since the gear types selected different size classes (and probably different age groups), the different results may indicate that salinity effects on abundance depend on the life stage or age in the two species.

Based on trawl samples, white shrimp and juvenile Gulf menhaden had highest mean catch rates in estuaries with mean salinities of less than 20‰. Blue crab catch rates were highest in estuaries having mean salinities of less than 22‰, with the exception of the lower Laguna Madre which also had a high catch rate and a mean bottom salinity of 33.5‰. Speculation about the reason blue crabs do well in the lower Laguna Madre included the possible use of low-salinity areas in the Arroyo Colorado for mating and overcounting due to mistaken identification with the lesser blue crab. Brown shrimp had higher catch rates when mean salinities were between 17 and 30‰. Atlantic croaker had higher catch rates from estuaries that had salinities between 20 and 30‰. Juvenile southern flounder had the highest catch rates in the Mission-Aransas with a mean salinity of about 21‰.

Effects of inflow on the harvest of fish and shellfish. In Section 6.8, significant regressions were reported between inflows, at various times of the year, and commercial harvests of blue crab, oysters, white shrimp, brown shrimp, red drum, black drum, and spotted seatrout. For each estuary, a set of independent equations were reported for each species, except in a few instances where there were no significant equations or where there were problems with the data. Shellfish regressions showed that lagging inflows by a two- to ten-month period prior to the time the organism was actually in the estuary generally produced equations with the highest R^2 values. The fish equations required a running mean inflow based on the three years prior to the harvest, and oysters and crabs required a running mean based on the inflow during the year of harvest and one year prior to the harvest. The long antecedent period for fish indicates that young fish are more sensitive to inflow quantities than older animals. Another indication that inflow relationships for fish are more complicated than for shellfish is that R^2 values for the fish equations were generally lower than for shellfish.

These equations are empirical, and causal relationships can only be inferred. The form of these equations makes sense, however, considering what is known about the different life history and distribution for each species within the estuary. Section 6.8 also pointed out that development of equations based on fishery-independent data would be highly desirable because it would remove the commercial and regulatory bias from the data.

Final conclusions. Evidence from the literature and new studies presented in this chapter indicate that of the economically important species considered, populations of white shrimp, oysters, and blue crabs would most likely decline with reductions in freshwater inflows. Brown shrimp densities would probably decline if salinities in large areas of any estuary were consistently maintained at more than 30‰. If mean salinities were less than 20‰, brown shrimp densities would probably increase with increases in salinity up to 25‰.

Any changes in marine or estuarine finfish living in estuaries will probably be complicated by indirect effects (i.e., changing their habitat or food base) and lagged (caused by their relatively long life span). There is some evidence that juvenile Gulf menhaden and southern flounder may be affected by increased salinity. A negative linear regression was found between mean Gulf menhaden trawl catch rates and mean estuarine salinity. The highest mean catch rates for southern flounder caught in trawls were from estuaries having mean salinities between 17 and 22‰. Lower mean catch rates were associated with estuaries having mean salinities above or below this range.

Information presented here indicates juvenile animals with limited mobility are generally most affected by changes in freshwater inflows. Most of the larger and more mobile animals studied in this chapter tolerate a wide range of salinities including seawater (about 35‰). Animals that have invaded estuaries and maintain higher densities in low salinity areas are probably gaining an advantage by more easily obtaining food and finding protection from predators, parasites, and impairing disease organisms.

CHAPTER 7: AN ECOLOGICALLY SOUND ENVIRONMENT FOR THE GUADALUPE ESTUARY

7.0 INTRODUCTION

This chapter provides a detailed description of an ecologically sound Guadalupe Estuary in terms of the salinity regimes and nutrient and sediment loads needed for the ecological processes that maintain the plant and animal populations characteristic of the system. This information is used in Chapter 8 to estimate annual and seasonal freshwater inflow needs for the Guadalupe Estuary. Previous sections of this report have extracted information from many different sources to investigate relationships between freshwater inflows and abiotic or biotic parameters in estuaries. This chapter and Chapter 8 focus specifically on the Guadalupe Estuary.

Inflow and salinity regimes. Section 7.1 provides a description of the estuary, naming the bays, providing volumes, depths, sources of freshwater inflows, areas of upstream drainage basins, and reservoirs that already exist in the drainage system. It also provides a description of monthly and annual inflow quantities that have occurred during the period of record. Simple statistics are provided to summarize the pattern of inflows including the mean, median, and percentiles. Results of a trend analysis are also included to demonstrate inflow patterns during the period of record. The section concludes with a description of the salinity regimes and gradients that occur within the estuary.

Water quality profile. Section 7.2 provides general information on nutrient concentrations (e.g., total nitrate, phosphorus, and carbon) in different areas of the bay, and trend analyses for several nutrients. The section also discusses relationships between freshwater inflow quantities and changes in nutrient concentrations in different parts of the estuary.

Nutrient budget. Section 7.3 synthesizes a nutrient budget for the Guadalupe Estuary using total nitrogen. The budget was prepared by systematically identifying the sources, sinks, and losses of nutrients, and the relative importance each has to the whole system. It also serves as a check on the understanding of nutrient transfer within the system since large imbalances in this budget would indicate that significant components had been missed.

An estimated minimum nitrogen load constraint. Section 7.4 estimates the minimum annual freshwater inflow required to provide an annual nitrogen load to the Guadalupe Estuary that offsets losses from the system. The constraint was developed using information in sections 7.2 and 7.3. It assumes that if sufficient nitrogen is provided, adequate quantities of other nutrients like phosphorus and organic carbon will also be supplied.

Sediment load requirements. Section 7.5 estimates the minimum annual freshwater inflow required to provide sufficient sediments for maintaining bathymetry in the Mission Lake area of the upper portion of the Guadalupe Estuary. This minimum sediment requirement will provide for some marsh development to offset marsh losses due to subsidence or sea level rise. This section begins by summarizing the recent geologic sediment history for the Guadalupe Estuary and describes river-borne sediments and how they have built the Guadalupe Delta. It also briefly discusses how sediments influenced Mission Lake, Guadalupe Bay, and the upper portion of San Antonio Bay.

The biota. Section 7.6 summarizes the knowledge about the plant and animal populations comprising the communities in the Guadalupe Estuary by reviewing published scientific articles and presenting an inventory of estuarine plants and animals. The section provides a description of different plant communities, benthic populations, and finfish and shellfish populations that inhabit different areas within the Guadalupe Estuary. When information was available, the relative abundance for each animal species was estimated by calculating the mean catch rates and frequency of occurrence in different monitoring gear. The section also discusses the relationship between relative abundance and freshwater inflows or salinity for selected animal species which are ecologically or economically important to the Guadalupe Estuary.

7.1 INFLOW AND SALINITY PATTERNS IN THE GUADALUPE ESTUARY

Introduction

Inflow and salinity patterns in the Guadalupe Estuary described in this section complement the broad comparative descriptions of inflow and salinity presented in Section 4.1 for the Guadalupe and five other Texas estuaries. The hydrological database used in the descriptions and analyses covers the period 1941 to 1987. Monthly combined inflows for this period are presented in Table 7.1.1. This is the most recent (September 10, 1992) update of the combined inflow data for the Guadalupe Estuary. Fishery equations presented in Section 6.8 and salinity equations presented in Section 8.2 were based on an earlier interim data set. The salinity database covers the period 1968 to 1987.

Inflows to the Estuary

Estuary and basin description. The Guadalupe Estuary lies on the central gulf coast of Texas and covers 551 km^2 (213 mi^2) at mean low water (Diener 1975). Its volume at mean low water is roughly 787 million m^3 (638,000 acre-ft), and its average depth is 1.4 m (4.7 ft).

The estuary consists of a series of interconnected bays including San Antonio Bay, Mesquite Bay, Espiritu Santo Bay, Mission Lake, Guadalupe Bay, Hynes Bay, and Ayres Bay (Figure 7.1.1). San Antonio Bay is bisected northeast by southwest by the Intracoastal Waterway which varies in depth from 3.7 to 4.5 m (12 to 15 ft) with a bottom width of about 38.1 m (125 ft) (TDWR 1980). The Guadalupe Estuary exchanges water with the Lavaca-Colorado Estuary via Saluria Bayou and Espiritu Santo Bay to the northeast. To the southwest, water exchange occurs with the Mission-Aransas Estuary via Mesquite Bay and with the Gulf of Mexico via Cedar Bayou. Although Cedar Bayou leads directly to the Gulf of Mexico, it is often closed and generally accounts for only a small exchange volume.

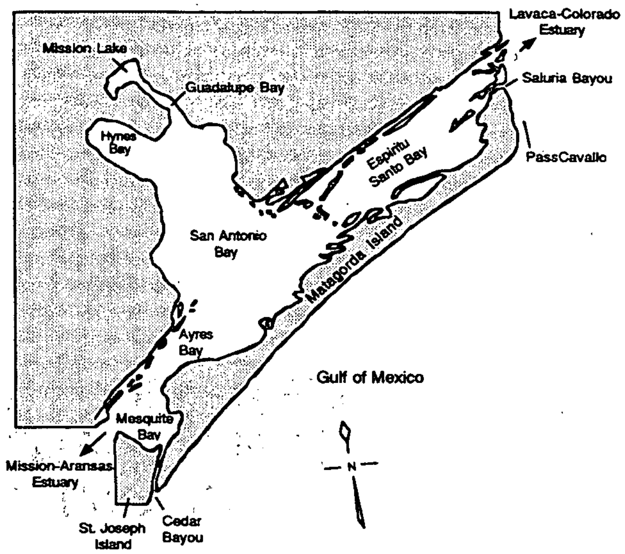


Figure 7.1.1. Guadalupe Estuary.

The San Antonio River and the Guadalupe River contribute the largest volumes of fresh water to the Guadalupe Estuary. Under normal conditions, inflows from these rivers enter through Guadalupe Bay and Mission Lake. During major flood events, however, flood waters inundate the entire delta and enter through Hynes Bay as well. These floods accelerate transport of sediments, nutrients, and fresh water into San Antonio Bay.

The total contributing drainage area for the Guadalupe Estuary covers 26,736.3 km^2 (10,322.9 mi^2). It includes two major river basins, the San Antonio and Guadalupe river basins, and portions of two smaller coastal basins, the Lavaca-Guadalupe and San Antonio-Nueces coastal basins (Figure 7.1.2). Annual precipitation over the contributing area varies from 71 cm (28 in) in the western regions of the Guadalupe and San Antonio river basins to 102 cm (40 in) inches near the Gulf coast (Figure 4.1.3). Gross evaporation from the estuary exceeds precipitation, averaging roughly 152 cm (60 in) per year.

For hydrological purposes, the entire drainage basin is divided into gaged and ungaged areas. Runoff from gaged areas is measured by USGS streamflow gages on the Guadalupe River at Victoria, on Coleto Creek near Schroeder, and on the San Antonio River at Goliad (USGS stations 08176500, 08177500, and 08188500, respectively). 93% of the total drainage area contributing to the Guadalupe Estuary (24,949.5 km^2 or 9,633 mi^2) is gaged, while 7% (1,784.5 km^2 or 689 mi^2) is ungaged. The gaged drainage area includes 14,794.1 km^2 (5,712 mi^2) of the Guadalupe River Basin (including the gaged portion of the Coleto Creek watershed) and 10,155.4 km^2 (3,921 mi^2) of the San Antonio River Basin.

Table 7.1.1. Combined[a] inflows (acre-ft) by month for the Guadalupe Estuary during the period 1941 through 1987. The 47-year average (Avg.) is presented at the end of the table.

Year	January	February	March	April	May	June	July	August	September	October	November	December	Total
1941	208,986	344,059	394,149	532,568	1,283,832	436,441	374,411	132,962	148,197	196,640	113,360	123,296	4,288,901
1942	72,453	144,825	90,952	192,928	125,743	72,053	1,008,185	177,796	637,485	324,775	149,534	125,529	3,122,258
1943	151,167	100,561	148,258	85,779	116,182	184,116	116,305	57,304	81,563	57,047	88,278	98,250	1,284,810
1944	277,063	122,603	336,347	110,428	519,148	214,995	92,000	106,869	187,966	100,225	102,057	213,523	2,383,224
1945	243,991	234,538	224,377	472,687	122,174	158,565	100,268	245,586	52,287	122,212	62,962	80,244	2,119,891
1946	103,463	182,925	254,490	142,357	263,991	318,676	82,716	201,721	640,578	876,090	300,211	174,832	3,542,050
1947	281,286	149,605	170,263	186,643	348,644	106,742	95,220	110,178	58,015	55,886	80,270	69,210	1,711,962
1948	81,223	104,132	99,192	53,123	159,356	43,912	71,793	101,516	117,449	49,481	33,801	36,393	951,371
1949	41,909	83,000	125,011	600,315	221,648	133,778	142,436	64,568	50,327	430,785	72,990	113,718	2,080,485
1950	66,046	76,536	56,298	104,632	71,209	197,897	50,852	38,281	41,047	30,007	28,588	33,327	794,720
1951	31,944	34,677	37,406	38,831	133,054	233,295	26,819	16,970	243,925	34,889	32,970	29,700	894,480
1952	29,279	47,383	41,797	101,304	223,042	115,257	46,954	17,254	542,233	53,168	118,497	147,196	1,483,364
1953	119,162	69,497	50,817	66,199	274,839	25,787	27,285	205,243	209,021	139,015	51,636	70,468	1,308,969
1954	45,954	35,069	32,404	40,940	63,494	22,424	14,121	9,708	11,817	55,255	24,742	20,188	376,116
1955	25,522	87,086	31,195	22,676	76,795	58,362	17,869	43,291	78,854	20,785	11,267	20,514	494,216
1956	18,497	20,830	17,747	20,751	44,181	5,123	6,534	7,129	15,564	43,565	16,480	58,681	275,082
1957	14,045	36,638	259,004	670,751	755,462	559,930	51,808	28,589	395,576	579,281	448,581	142,469	3,942,134
1958	483,313	892,688	290,252	146,426	419,653	133,588	108,657	57,772	315,823	328,197	239,596	191,175	3,607,140
1959	109,703	303,523	105,910	266,573	172,520	125,528	101,614	138,077	66,247	366,714	108,241	111,245	1,975,895
1960	114,348	151,373	128,436	99,752	179,988	317,858	216,297	282,204	85,206	1,207,435	666,397	413,726	3,863,020
1961	361,891	453,103	195,250	134,047	88,393	602,156	328,943	98,043	255,286	98,004	213,298	82,805	2,911,219
1962	76,454	69,500	63,508	91,362	71,036	159,030	46,221	29,417	95,191	58,018	66,115	109,880	935,732
1963	57,189	92,692	53,336	56,599	39,972	37,760	27,064	14,386	20,887	31,262	83,032	48,119	562,298
1964	55,853	108,343	127,933	52,070	40,329	70,725	21,454	90,022	133,839	72,573	93,224	58,081	924,446
1965	151,578	442,541	99,746	100,449	503,291	354,109	83,486	53,728	53,804	170,404	166,608	293,916	2,473,660
1966	155,788	177,200	117,988	218,470	386,970	139,550	125,480	78,386	100,763	102,844	74,562	65,036	1,743,037
1967	76,215	63,369	58,445	63,587	85,348	47,934	29,903	77,089	2,230,029	483,034	260,431	187,550	3,662,934
1968	726,253	300,075	188,455	242,592	566,354	571,037	286,423	110,627	184,297	159,371	111,848	186,518	3,633,850
1969	145,207	278,687	297,435	401,626	355,526	166,866	64,447	66,624	70,742	114,581	117,163	158,432	2,237,336
1970	184,894	140,864	280,008	186,705	289,822	407,126	122,390	74,169	115,624	148,342	58,834	58,524	2,067,302
1971	56,117	45,834	48,038	46,702	60,462	41,658	43,242	182,893	514,109	390,658	224,494	304,219	1,958,426
1972	153,711	175,038	104,360	90,452	1,079,100	301,176	141,436	142,284	114,454	173,891	186,843	76,342	2,739,087
1973	97,904	134,253	193,091	454,872	202,233	806,857	556,658	353,924	344,657	1,262,823	387,740	237,290	5,032,302
1974	284,288	207,120	168,645	110,929	294,726	153,030	69,280	129,936	376,212	196,873	409,492	363,835	2,764,366
1975	217,693	376,786	250,248	215,880	712,723	652,469	312,283	189,580	187,469	99,742	123,092	111,534	3,449,499
1976	110,963	71,530	79,781	537,215	587,928	328,104	304,567	141,070	154,259	399,041	585,331	765,991	4,065,780
1977	302,332	382,551	207,036	925,080	500,180	442,108	142,353	95,829	120,977	108,671	226,179	94,743	3,548,039
1978	96,599	114,037	88,193	97,840	64,627	166,559	43,181	331,043	527,294	139,278	190,042	102,580	1,961,273
1979	464,103	310,791	332,049	509,228	724,087	563,313	291,350	168,579	409,211	80,462	80,751	86,621	4,020,545
1980	137,202	85,720	67,410	61,761	276,312	72,880	37,064	110,218	144,762	87,221	75,942	76,405	1,232,897
1981	89,280	75,529	103,917	126,764	302,742	1,108,493	389,276	181,141	847,826	317,581	433,818	153,319	4,129,686
1982	111,174	240,522	102,800	84,237	477,749	96,481	55,311	43,375	37,396	72,078	171,240	70,695	1,563,058
1983	76,745	142,843	203,897	98,816	113,462	97,831	199,890	63,749	110,438	113,436	106,937	52,306	1,380,350
1984	91,229	59,226	90,903	38,214	40,210	20,423	9,354	23,308	12,326	149,189	90,538	83,339	708,259
1985	184,905	116,247	261,536	306,931	126,842	232,937	225,574	73,649	74,292	169,326	291,479	240,610	2,304,328
1986	139,539	124,181	92,018	67,172	144,822	387,685	98,479	56,961	103,393	284,364	223,892	511,303	2,233,809
1987	387,270	304,561	385,522	169,669	240,061	2,457,912	557,117	335,604	192,385	138,447	145,956	116,470	5,430,974
Avg.	159,824	176,908	152,252	200,956	296,814	296,820	156,689	114,014	244,917	227,510	169,135	148,301	2,344,140

[a] Combined inflow = gaged + ungaged; ungaged = modeled + return flows - diversions

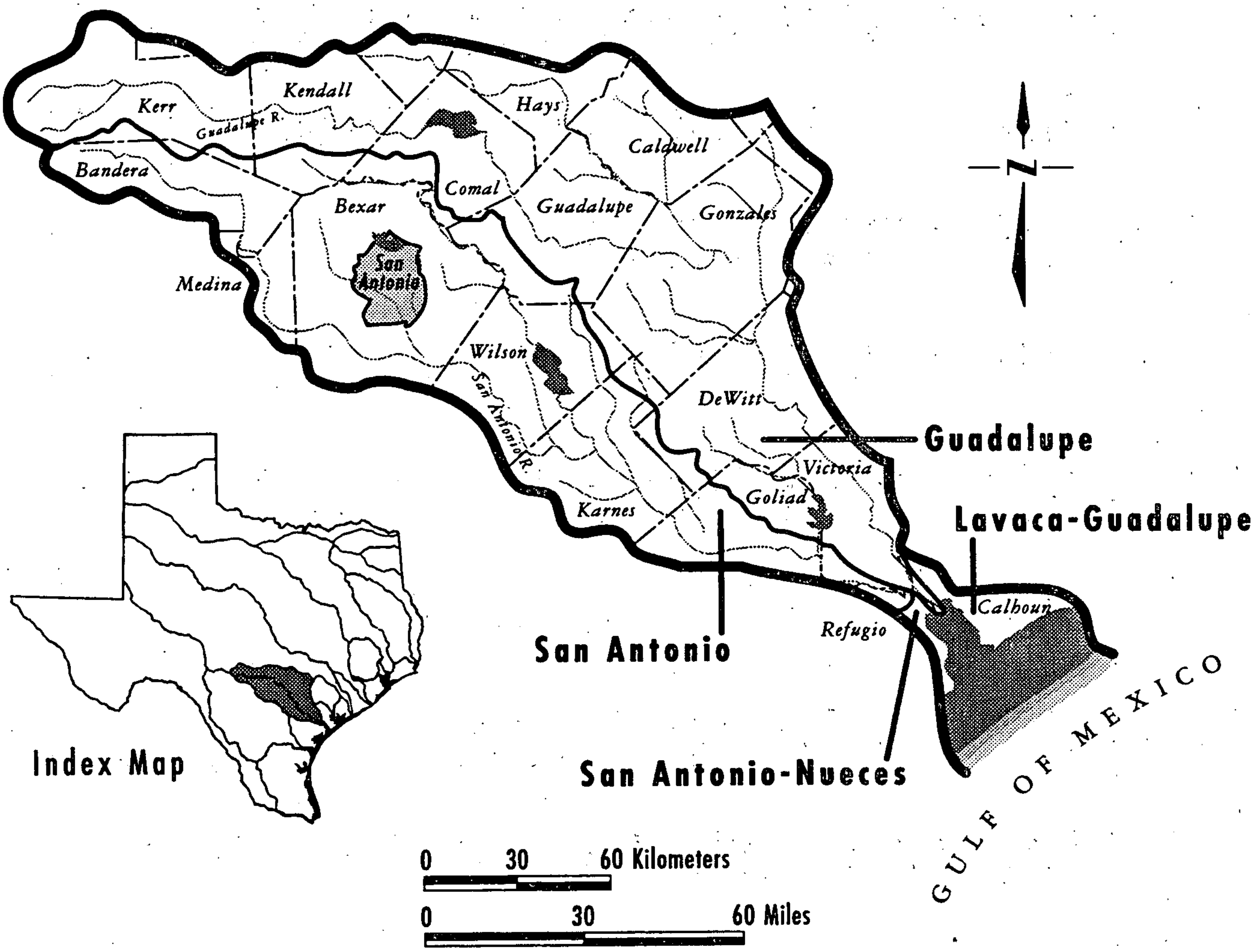


Figure 7.1.2. Basins contributing to the Guadalupe Estuary.

Reservoirs. Over the period of record, several major reservoirs were built in the Guadalupe and San Antonio basins (Table 4.1.3) primarily for water supply and flood control. Canyon Reservoir, operated as a flood control and hydroelectric generation reservoir, is presently the largest reservoir (1,392 million m^3) in the Guadalupe River Basin (TDWR 1980b). Medina Lake, built in 1913 and operated for irrigation purposes, is currently the largest reservoir (313 million m^3) in the San Antonio River Basin. Reservoirs generally tend to reduce peak flows, alter seasonal patterns of riverine flows that reach the bay, and, in some instances, provide insurance against periods of drought. Reservoirs are built to store water for municipal, agricultural, recreational, and industrial uses, and for flood control.

Inflow statistics. Average freshwater inflow to the Guadalupe Estuary over the period of record was 195,619 acre-ft/month (2.34 million acre-ft/yr); the median inflow was 120,444 acre-ft/month. Over this period, 80.8% came from gaged areas, and 19.2% came from ungaged areas. The average freshwater inflow volume entering the Guadalupe Estuary each year represents three to four times the volume of the estuary (Figure 4.1.5). Coincidentally, the average flushing time, considering only terrestrial freshwater inflows, is three to four months.

By far the largest fraction of freshwater inflows to the Guadalupe Estuary comes from gaged portions of the Guadalupe River Basin. These areas contributed 58.1% (1.34 million acre-ft/yr) of total freshwater inflows to the estuary; 56.9% came from gaged portions of the Guadalupe River alone. Gaged portions of the San Antonio River contributed 22.7% (0.532 million acre-ft/yr) of total freshwater inflows.

Ungaged watersheds lying below the lowest streamgages generated the remaining fraction of freshwater inflows entering the Guadalupe Estuary. These ungaged freshwater inflows consist of runoff from ungaged watersheds computed with a rainfall/runoff simulation model less diversions of water from streams in the watersheds plus return flows to streams in the watersheds or directly to the estuary. Over the period of record, runoff computed from the rainfall/runoff simulation model averaged 0.462 million acre-ft/yr, diversions averaged 0.029 million acre-ft/yr, and return flows averaged 0.017 million acre-ft/yr, representing 19.7%, 1.2%, and 0.7 % of total freshwater inflows to the estuary, respectively.

Monthly inflow patterns. Average and standard deviations of monthly inflows for the entire Guadalupe Estuary, gaged Guadalupe River flows at Victoria, and gaged San Antonio River flows at Goliad are presented in Figure 7.1.3. The seasonal pattern of inflows for each record is quite similar. In each case, the largest and most variable inflows occur from May to June and from September to October, and the smallest inflows occur in August. May and June inflows average 295,128 and 295,687 acre-ft/month, respectively, reflecting the occurrence of heavy springtime precipitation throughout the contributing basin. The most variable inflows occur in June and September (standard deviations of 395,115 and 319,290 acre-ft/month, respectively). The smallest inflows occur in late August and December (118,516 and 149,746 acre-ft/month, respectively), while the least variable flows are in August and March (standard deviations of 94,218 and 102,002 acre-ft/month, respectively). Additional basic inflow statistics for the Guadalupe Estuary are presented in Tables 7.1.2 and 7.1.3. (Skewness, the normalized third moment of a frequency distribution, indicates the degree of asymmetry of the distribution. A skewness of zero indicates that the distribution is symmetric about the mean value. A positive skewness, as found for Guadalupe Estuary monthly inflows and typical

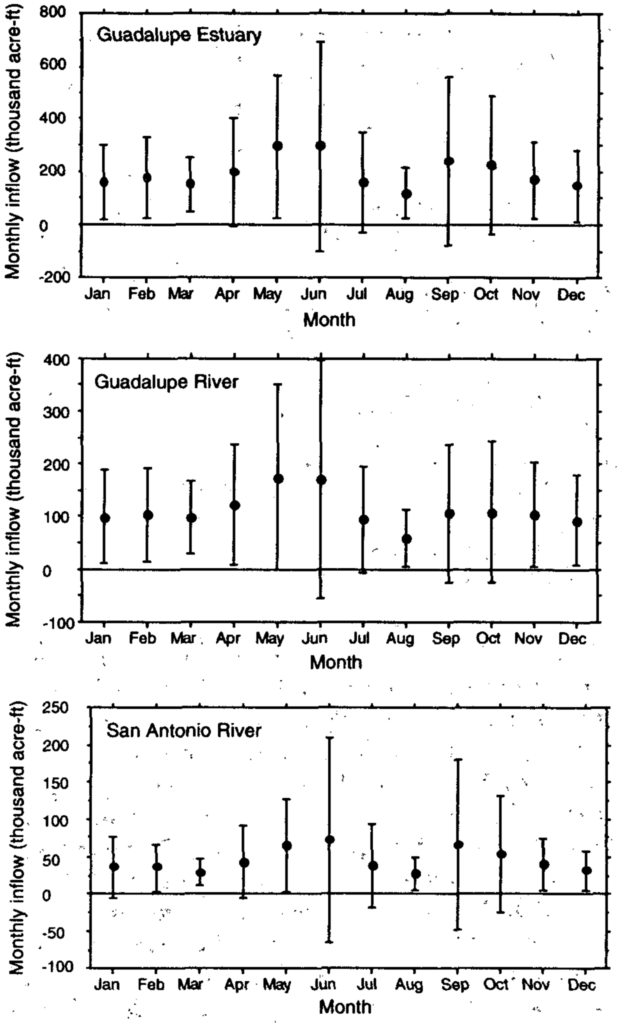


Figure 7.1.3. Mean and one standard deviation error bars for monthly freshwater inflows to the Guadalupe Estuary and for monthly gaged inflows recorded on the Guadalupe River at Victoria(USGS station 08176500) and on the San Antonio River at Goliad (USGS station 08188500).

Table 7.1.2. Guadalupe Estuary monthly freshwater inflow statistics, 1941 to 1987, in acre-feet.

Statistic	Value
Mean monthly freshwater inflow	195,345
Standard deviation	230,442
Standard error	9,703
Minimim inflow (June 1956)	5,123
Maximum inflow (June 1987)	2,457,912
Skewness	4.23
Percentile rank	
10th	37,405
25th	69,389
50th	117,719
75th	240,292
90th	431,088

Table 7.1.3. Mean, median, and standard deviation of inflows by month for the Guadalupe Estuary, 1941 to 1987.

Month	Mean	Median	Standard deviation
January	159,824	111,174	141,200
February	176,908	124,181	156,126
March	152,252	117,988	101,101
April	200,956	110,428	201,649
May	296,814	221,648	273,750
June	296,820	166,559	397,102
July	156,689	95,220	186,013
August	114,014	95,829	88,129
September	244,917	120,977	353,607
October	227,510	138,447	272,548
November	169,135	113,360	147,476
December	148,301	109,880	138,764

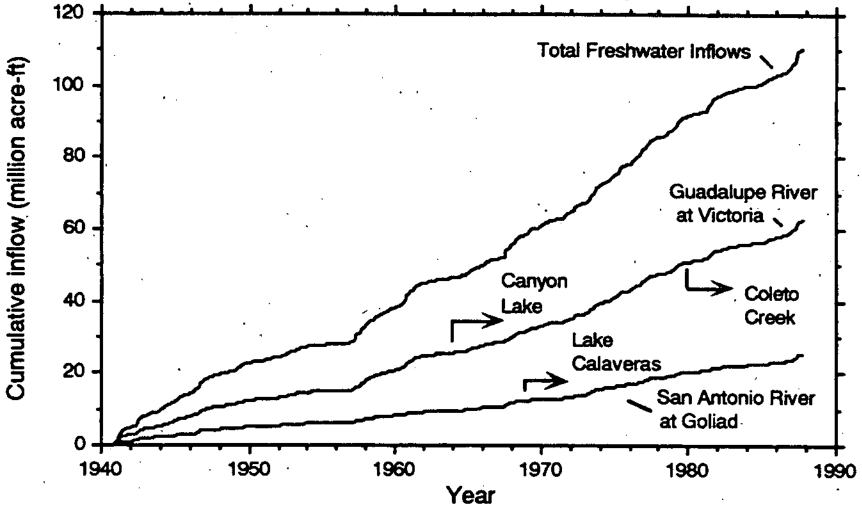


Figure 7.1.4. Cumulative freshwater inflows to the Guadalupe Estuary and cumulative gaged inflows on the Guadalupe River at Victoria and on the San Antonio River at Goliad, 1941 to1987. Data of major impoundments indicated by bent arrows.

for river inflows, indicates that the distribution is asymmetric with a tail extending to the positive side of the mean, while a negative skewness indicates a tail to the negative side of the mean. Positive skewness can be attributed to a few large inflow events.)

Trend analysis. Cumulative inflow hydrographs and statistical analyses of inflows to the entire Guadalupe Estuary presented in Section 4.1 indicated no significant reduction in mean inflow rates to the estuary due to construction of reservoirs. Similar analyses of flows recorded on the Guadalupe River at Victoria and on the San Antonio River at Goliad are presented here. Cumulative inflows (mass curves) for gaged Guadalupe River inflows, for gaged San Antonio River inflows, and for the entire Guadalupe Estuary are presented in Figure 7.1.4. Results of the Sen nonparametric trend analysis test (Van Belle and Hughes 1984) applied to the same three sets of inflows are presented in Table 7.1.4. In both the cumulative inflow hydrographs and in the nonparametric trend analysis, the only trends evident are those during drought periods. Decreasing trends in freshwater inflows are found for the periods 1941-1958 and 1958-1966, while no significant trend is found for inflows to either river for the 1966-1987 period. Drought conditions existed in Texas during the mid-1950's and early 1960's, accounting for the decreasing trends. Although anomalies such as the decrease in inflow rates from roughly 1981 through 1986 do appear (attributable to less-than-

Table 7.1.4. Trend analysis of monthly inflows to the Guadalupe Estuary, the Guadalupe River at Victoria (USGS station 08176500), and the San Antonio River at Goliad (USGS station 08188500). Sen's t-statistic, % change per year of deseasonalized inflow data, significance of result.

Period and area	t	% change/year	Significant?
1941-1958			
Guadalupe Estuary	-4.14	-3.85	Yes
Guadalupe River at Victoria	-4.24	-3.53	Yes
San Antonio River at Goliad	-3.60	-3.62	Yes
1958-1966			
Guadalupe Estuary	-3.25	-10.9	Yes
Guadalupe River at Victoria	-3.63	-11.14	Yes
San Antonio River at Goliad	-3.86	-11.20	Yes
1966-1987			
Guadalupe Estuary	-1.97	-0.52	No
Guadalupe River at Victoria	0.20	1.04	No
San Antonio River at Goliad	2.07	0.97	No

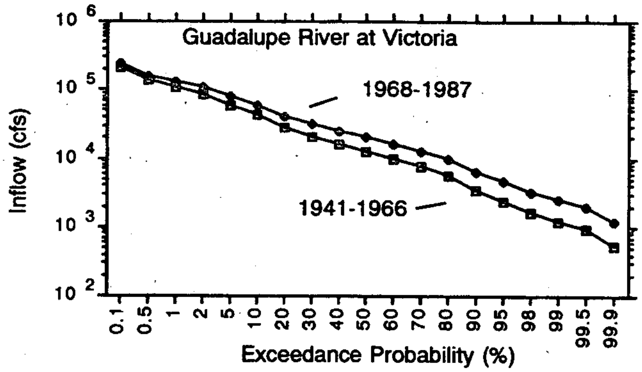


Figure 7.1.5. Exceedance probability curves for the Guadalupe River at Victoria. Statistics are based on maximum annual mean daily inflows.

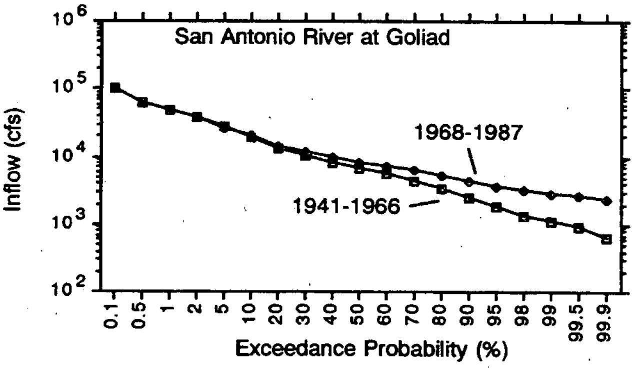


Figure 7.1.6. Frequency-duration curves for the San Antonio River at Goliad. Statistics are based on annual maximum daily inflows.

average precipitation for that period), the overall picture presented is of no significant changes in average freshwater inflow rates to the Guadalupe Estuary from 1941-1987 except during droughts.

Exceedance probability analysis. Comparison of exceedance probabilities for maximum gaged daily streamflows on the Guadalupe (at Victoria) and San Antonio (at Goliad) rivers suggests that inflows on both rivers were greater for the period 1968 to 1987 than from 1941 to 1966 (Figures 7.1.5 and 7.1.6). Comparison of mean inflows for the same periods supports the same conclusion. Mean annual inflows on the San Antonio River were 0.396 million acre-ft from 1941 to 1966, and 0.690 million acre-ft from 1968 to 1987. On the Guadalupe River, mean annual inflows increased from 1.08 million acre-ft to 1.67 million acre-ft over the same periods. The difference in mean inflows between these two periods can be attributed to increased urbanization in both watersheds, increased groundwater pumpage and return flows, and increased precipitation in the latter period. (Recall that two droughts occurred in the earlier period.) For the Guadalupe River, flows at all exceedance probabilities for the 1968 to 1987 curve lie above those in the 1941 to 1966 curve (Figure 7.1.5). However, the higher amplitude, low probability flood flows show less of an increase than the low amplitude, high frequency flows. This suggests that Canyon Reservoir may be acting to attenuate large amplitude flood peaks on the Guadalupe River. On the San Antonio River (Figure 7.1.6), low amplitude flows show a significant increase for the latter (1968 to 1987) period, while large amplitude flows show almost no increase. This may be due strictly to an increase in groundwater return flows, which tend to be frequent but relatively small in amplitude, from San Antonio into the San Antonio River.

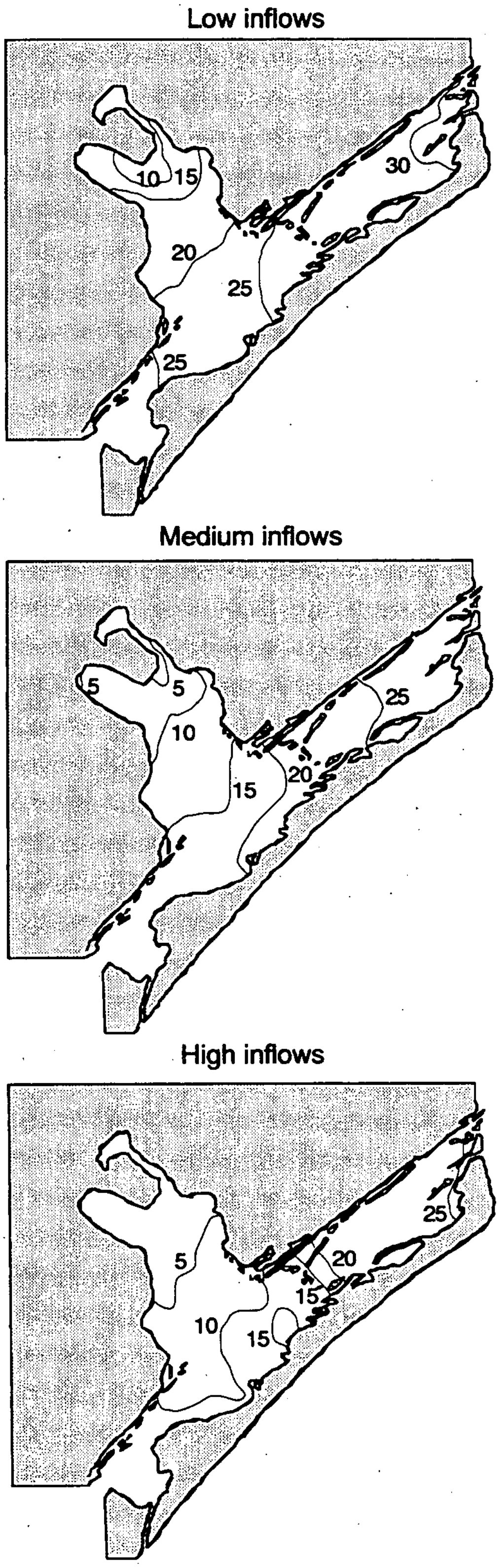
Salinity Characteristics

Salinity statistics. Basic statistics describing salinity conditions for five sub-areas in the Guadalupe Estuary were provided in Table 4.1.2. In addition, mean salinity levels, salinity frequency distributions, and seasonal salinity patterns are illustrated in Figures 4.1.6, 4.1.15, and 4.1.16, respectively.

Overall average salinities range from roughly 6‰ near Seadrift in the upper reaches of the estuary, to roughly 25‰ in Espiritu Santo Bay. Salinity from the upper to the lower reaches of San Antonio Bay generally decrease by about 10‰; salinity is generally lower in Mesquite Bay than in Espiritu Santo Bay.

Salinity distribution in the estuary. A more refined breakdown of salinity conditions in the Guadalupe Estuary is presented in Figure 7.1.7. For this figure, the salinity database was sorted to represent conditions during low, medium, and high inflows to the estuary. Low inflows were taken as monthly inflows below the 25th percentile flow (70,100 acre-ft/month, Table 4.1.1), medium inflows as being between the 25th and 75th percentiles (70,100 acre-ft/month and 241,000 acre-ft/month, respectively), and high inflows as being above the 75th percentile inflow. Salinities were sorted according to the inflow of the month the salinity measurement was taken. Isohalines for these three conditions are presented in Figure 7.1.7. The overall effect of increasing inflows is to drive fresh water down the estuary and into the lagoonal arms. Salinity in most of San Antonio Bay decreases by roughly 10‰ going from low inflow conditions to high inflow conditions. The lagoonal arms of the estuary are more buffered from inflow variations. Salinities in Espiritu Santo Bay vary by only 5‰ over extreme inflow conditions.

A trend analysis for salinity in the Guadalupe Estuary was presented in Section 4.1. The only trend indicated was that of increasing salinity in lower mid-San Antonio Bay. No trends were found for the remaining sites in the estuary.



Low inflows

Medium inflows

High inflows

Figure 7.1.7. Salinity contours for low (less than 25th percentile), medium (between 25th and 75th percentile, 70,100 and 241,000 acre-ft/month, respectively), and high (greater than 75th percentile) inflow conditions in the Guadalupe Estuary.

Summary

Primary contributors of fresh water to the Guadalupe Estuary are gaged portions of the Guadalupe and San Antonio rivers; historically, these sources have provided 79.6% of freshwater inflows to the Guadalupe Estuary. Seasonal characteristics for total freshwater inflows to the estuary and for gaged inflows are all very similar. Bimodal peaks occurring in late spring and early fall appear in each of these records. Despite development of reservoirs and urban areas in these basins, the only trends found in inflow records were associated with drought periods. Similarly, no significant trends in salinity were found in four of five sites in the Guadalupe Estuary.

7.2 PROFILE OF WATER QUALITY IN THE GUADALUPE ESTUARY

Bay Nutrient Concentrations

To complement the discussion of distribution of major nutrients within the Guadalupe Estuary presented in Section 4.2, a water quality profile of the estuary is presented here, based on data compiled for four selected areas illustrated in Figure 7.2.1. Data were compiled from databases in the TWDB Coastal Data System and from the TNRCC Statewide Monitoring Network. Approximate locations of major sampling stations are shown in Figure 7.2.1. With respect to the Guadalupe Estuary, routine monitoring programs of the TWDB and TNRCC began in 1967 and have continued through 1989. Frequency of sample collection during this period has varied from monthly, to semi-monthly, to quarterly, or yearly, depending on the site and parameter category. Parameters discussed here were usually sampled semi-monthly or quarterly. Additionally, data from several estuarine ecological studies of one to three years' duration, performed during the 1970's, are included from the TWDB Coastal Data System. Chemical analyses for most of the samples were performed by the Texas Department of Health. Table 7.2.1 presents basic summary statistics for nitrogen, phosphorus, and total organic carbon concentrations for each bay area. The dissolved inorganic nitrogen reported in the table is the sum of nitrate, nitrite, and ammonium nitrogen concentrations. Total nitrogen is defined as nitrate plus nitrite plus total Kjeldahl nitrogen.

Seasonal variation. Variation in nutrient concentrations arise from variability in the external supply and from seasonal and non-seasonal fluctuations in the uptake and recycling of nutrients in the bay. Figures 7.2.2, 7.2.3, and 7.2.4 show monthly variation of mean nutrient concentrations in the Seadrift area, mid-bay, Mesquite Bay, and Espiritu Santo Bay, grouping all years' samples by

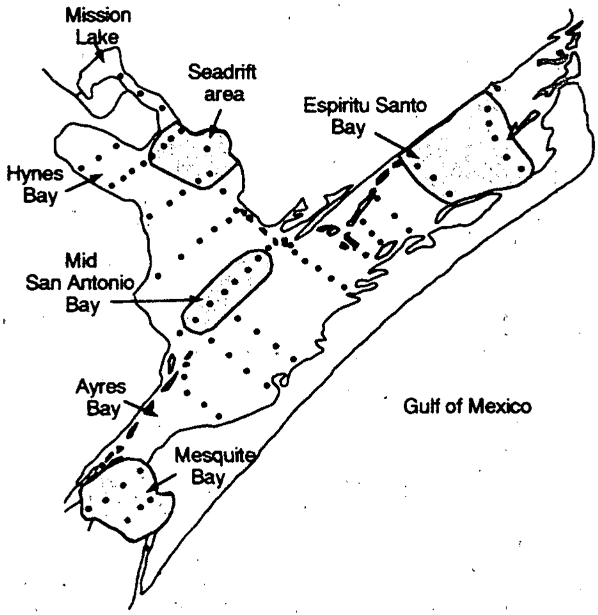


Figure 7.2.1. Areas within the Guadalupe Estuary for which descriptive water quality analyses were developed. Dots show approximate positions of historical monitoring stations.

month. The reliability of the means as indicators of seasonal variation is illustrated by the 95% confidence interval plotted about the means.

Nutrient concentrations in upper San Antonio Bay near Seadrift are more variable than in other areas of the estuary, probably as a result of periodic changes in inputs from the Guadalupe River. Dissolved inorganic nitrogen and phosphorus show decreasing variation with increasing distance from the river mouth. In contrast, total organic carbon remains variable throughout the estuary. This probably reflects the diversity of organic materials, including phytoplankton, included in this measure. Populations of planktonic microorganisms typically bloom and then die cyclically in response to varying chemical and physical conditions in the estuary (Riley, 1967).

Variation induced by the influence of floods and meteorological events makes it hard to define seasonal patterns in estuaries. Orthophosphorus and dissolved inorganic nitrogen concentrations appear to increase during fall and winter when the rates of uptake by phytoplankton would be reduced. In contrast, total organic carbon concentrations include carbon fixed by this phytoplankton and other biological activity, and are high in the spring, summer, and early fall.

Trends in nutrient concentrations. Long-term trends of change in variable systems such as estuaries are difficult to measure. The period of record for most of the water quality monitoring stations is 1968 to 1989. Nevertheless, some stations were sampled more frequently than others during this period, and during some years the bays were sampled more frequently than during other years. To make best use of the data available, long-term variation in nutrient

Table 7.2.1. Average nutrient concentrations in regions and bays of the Guadalupe Estuary; units are mg/l. Presented as mean ± standard deviation, with number of observations in parentheses.

Parameter	Seadrift	Mid-bay	Espiritu Santo	Mesquite
Nitrate-N	0.46 ± 0.47 (140)	0.19 ± 0.31 (143)	0.06 ± 0.18 (116)	0.04 ± 0.09 (115)
Ammonium-N	0.07 ± 0.09 (113)	0.08 ± 0.15 (118)	0.06 ± 0.18 (119)	0.04 ± 0.09 (117)
DIN-N[a]	0.59 ± 0.58 (83)	0.17 ± 0.20 (123)	0.05 ± 0.03 (66)	0.11 ± 0.15 (110)
Total-N	1.53 ± 0.65 (45)	0.95 ± 0.34 (36)	0.77 ± 0.42 (41)	0.94 ± 0.40 (43)
Ortho-P[b]	0.11 ± 0.09 (58)	0.09 ± 0.06 (94)	0.04 ± 0.03 (46)	0.07 ± 0.05 (90)
Total-P	0.25 ± 0.13 (119)	0.14 ± 0.07 (124)	0.06 ± 0.04 (125)	0.12 ± 0.06 (111)
Total organic carbon	7.64 ± 5.84 (84)	7.66 ± 5.98 (78)	6.35 ± 7.16 (84)	7.23 ± 5.28 (82)

[a] DIN is dissolved inorganic nitrogen.

[b] Ortho-P is orthophosphate.

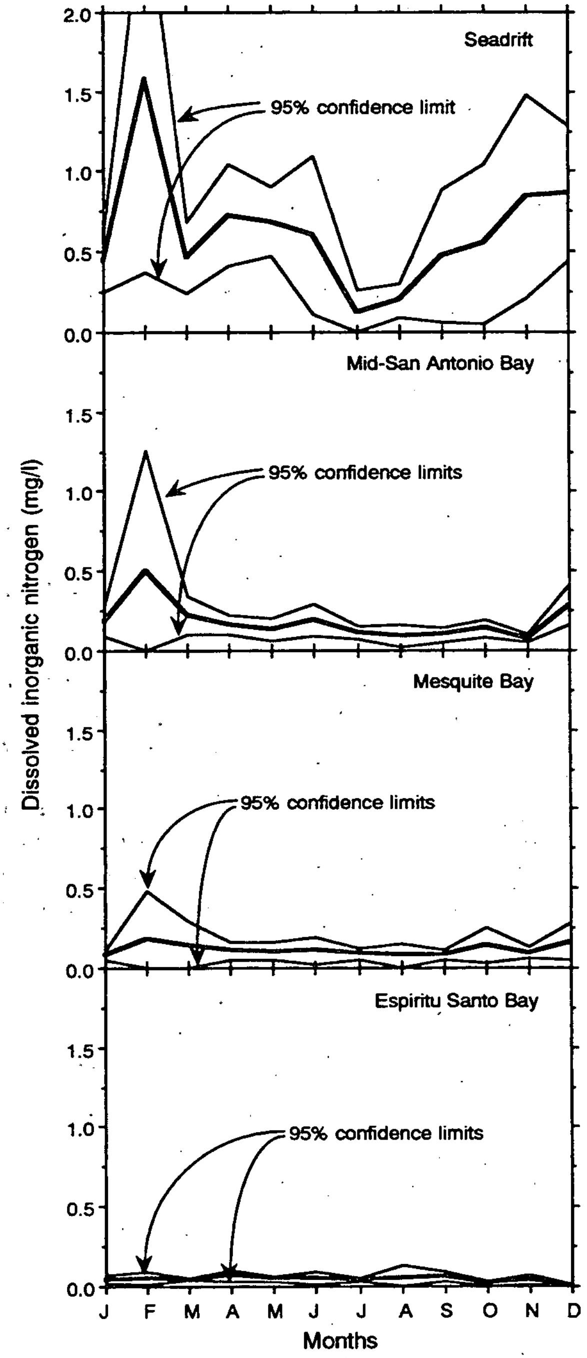


Figure 7.2.2. Seasonal variation of dissolved inorganic nitrogen (NH_3+NO_2+NO_3) in representative areas of the Guadalupe Estuary. The upper and lower 95% confidence intervals are shown flanking monthly means.

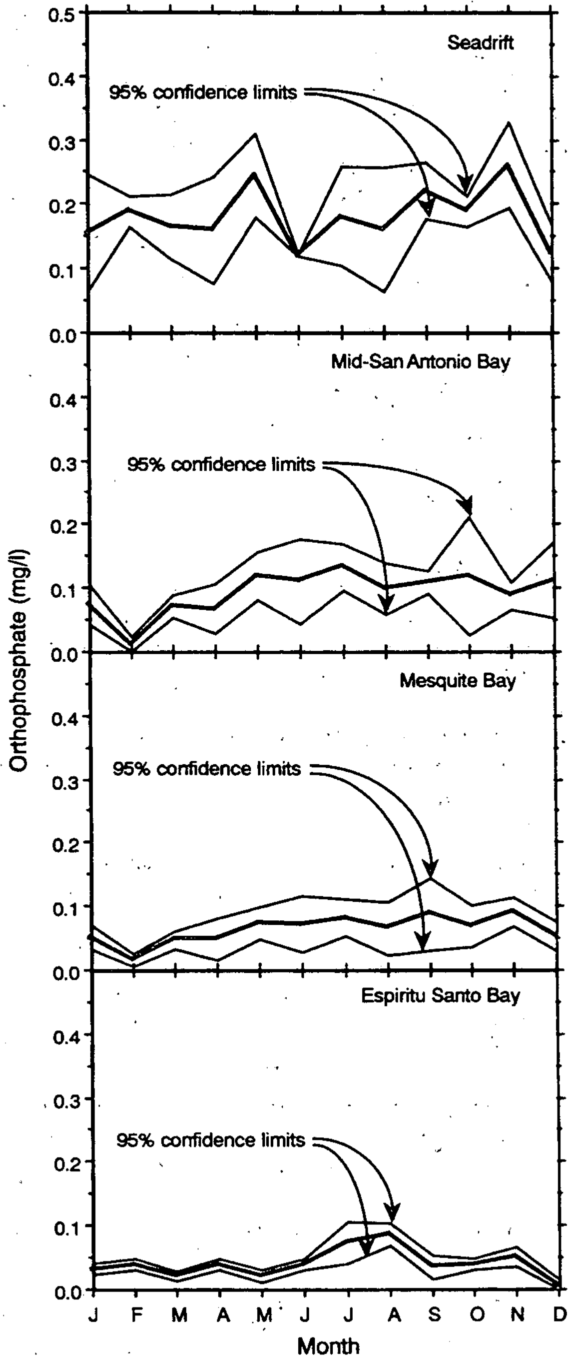


Figure 7.2.3. Seasonal variation of orthophosphate in representative areas of the Guadalupe Estuary. The upper and lower 95% confidence intervals are shown flanking monthly means.

concentrations of the Guadalupe Estuary was investigated using a nonparametric test based on Kendall's tau as proposed by Hirsch et al. (1982). This test was designed to reliably determine trends in data with seasonal or monthly variation. The seasonal component of concentration changes was factored out using simple three-month seasons (viz. January through March in the first season, etc.).

Results of the tests are presented in Table 7.2.2. The trend statistic, "z," indicates a statistically significant trend if its absolute value exceeds 1.96. Because this non-parametric test is based on ranked values and is not sensitive to the absolute values, it can indicate a significant trend when the magnitude of change is too small to be of concern. Therefore, the estimated percentage change per year in each parameter

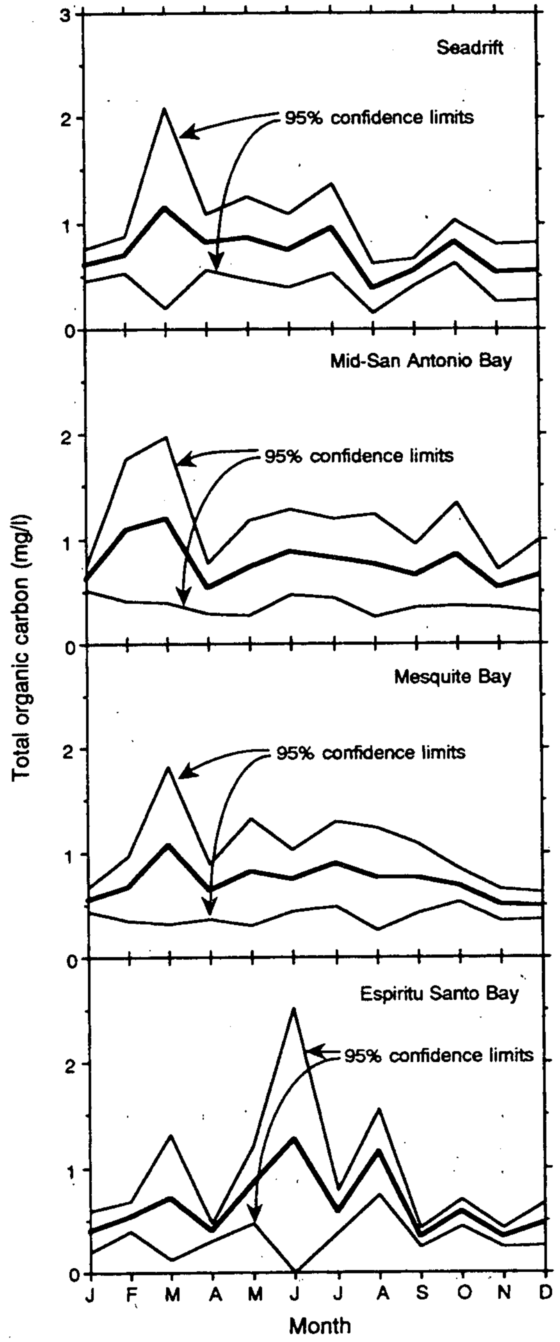


Figure 7.2.4. Seasonal variation of total organic carbon in representative areas of the Guadalupe Estuary. The upper and lower 95% confidence intervals are shown flanking monthly means.

was calculated from the slope of the best-fit line through the data, and this value is also presented in Table 7.2.2. Where this change is estimated to be less than 5%/yr, the differences may be below a level at which the measurement detection limits could reliably support the finding.

One result of the trend analysis is the contrast between apparent declines in concentration of dissolved inorganic nitrogen forms, while little change is observed in total nitrogen. Perhaps this indicates an increase in the efficiency with which the system converts and retains dissolved nutrients in the form of organic components. Two points should be noted in this regard. First, some total nitrogen values were estimated from a nitrate-based regression, which would tend to stabilize values because the constant term in the equation is fixed. Second, some suspicion should be directed at trends of decline, since the detection limits of various chemical methods have improved over the years. No attempt was made to correct for such improvement in this analysis, and values reported as "less than" were taken at the threshold level for that particular assay.

Although the values reported for concentration trends should be used with caution, the generality of some results suggests some hypotheses which justify further monitoring. First, the difference in trends between nitrogen and phosphorus indicates that nitrogen and phosphorus concentrations in the bays are independent over the long term. Since dissolved nitrogen and total organic carbon have both decreased, the trend results could indicate a linkage between dissolved nitrogen and organic matter production. Second, there is not a clear correspondence between trends in the lower bay regions and trends in the upper bay or in the Guadalupe and San Antonio rivers. This suggests the indicated trends in many areas of the estuary have more to do with internal changes which affect nutrient processing rather than with alterations in delivery of nutrients from outside to the estuary.

Relationship between Concentrations and Inflows

Floods are often assumed to enrich the estuary by bringing in a large amount of nutrients at once, increasing nutrient concentrations over much of the estuary. Of course the ability of the estuary to respond to nutrients brought in by high inflows depends on the season (due to temperature effects) and the offsetting flushing effect which disrupts activities of many species in parts of the estuary. The Guadalupe Estuary is complex hydraulically and biologically. Nutrient inputs may be filtered and partially processed in the upper bay. Some retention and storage occurs there also. An estimate of the influence of inflow variation on estuarine nutrient concentrations depends on the timing of sample collections with respect to rates of inputs, processing, and advection in the estuary. Due to these complicating factors, simple tests of the relationship between estuarine nutrient concentrations and inflows, such as are presented below, provide only a rough measure of the effect.

The first step in testing the association between inflow volume and bay nutrient concentrations required calculating the inflow volume over the 30-day period preceding each sampling. The nutrient data was then divided into four

Table 7.2.2. Water quality trends in bays of the Guadalupe Estuary and in the lower Guadalupe and San Antonio rivers for the 1968 to 1989 period of record; "z" is the trend statistic. The trend is significant when $|z| > 1.96$ ($P < 0.05$). The change is expressed as a percent per year on the next line. Positive or negative sign indicates increasing or decreasing trend.

Parameter	Guadalupe River	San Antonio River	Seadrift	Mid-bay	Mesquite Bay	Espiritu Santo Bay
Nitrate-N						
z value	-4.07	2.41	-0.08	-3.52	-2.43	-1.97
Trend (%/yr)	-3.2	1.8	2.0	-6.4	-6.4	-4.0
Ammonium-N						
z value	0.92	2.21	-1.69	-5.42	-6.16	-5.93
Trend (%/yr)	0.0	1.3	2.2	-12.0	-13.2	-13.2
DIN[a]						
z value	-0.55	3.11	1.83	-3.81	-4.39	1.95
Trend (%/yr)	-0.5	2.4	3.4	-6.0	-6.0	1.3
Total N						
z value	-2.94	2.01	1.13	-1.86	2.14	0.45
Trend (%/yr)	-1.1	1.2	0.7	-0.5	0.8	0.7
Ortho-P[b]						
z value	0.83	0.79	1.81	2.17	2.22	3.39
Trend (%/yr)	-3.6	2.0	3.2	1.6	2.8	4.4
Total-P						
z value	1.71	0.93	2.61	3.12	2.90	3.88
Trend (%/yr)	2.1	0.3	2.4	2.4	3.2	4.4
Total organic carbon						
z value	2.06	-0.49	-2.72	-3.20	-2.19	-2.67
Trend (%/yr)	4.3	-1.7	-3.6	-6.4	-4.4	-3.8

[a] DIN is dissolved inorganic nitrogen.

[b] Ortho-P is orthophosphate.

Table 7.2.3. Average dissolved inorganic and total nitrogen (mg/l) among inflow quantity categories (summed 30-day inflows) with Kruskal-Wallis test (H) for equality of means. "*" indicates that means of the various inflow categories differ significantly.

Inflow category (in acre-ft)	Seadrift	Mid-bay	Mesquite Bay	Espiritu Santo Bay
Dissolved inorganic nitrogen				
Inflow ≤ 70,140	0.40	0.10	0.06	0.052
70,141 ≤ inflow ≤ 120,040	0.79	0.12	0.12	0.045
120,041 ≤ inflow ≤ 240,560	0.66	0.17	0.08	0.035
Inflow ≥ 240,561	0.53	0.24	0.16	0.054
Kruskal-Wallis H	4.97	10.80	13.25	1.860
Probability	$P < 0.17$	$P < 0.013$ *	$P < 0.04$ *	$P < 0.60$
Total nitrogen				
Inflow ≤ 70,140	1.18	0.92	0.96	0.89
70,141 ≤ inflow ≤ 120,040	1.44	1.06	0.96	0.88
120,041 ≤ inflow ≤ 240,560	1.34	1.01	0.92	0.89
Inflow ≥ 240,561	1.37	1.25	0.99	0.97
Kruskal-Wallis H	12.56	23.62	2.87	1.15
Probability	$P < 0.006$ *	$P < 0.001$ *	$P < 0.41$	$P < 0.77$

Table 7.2.4. Average dissolved orthophosphate and total phosphate (mg/l) among inflow quantity categories (summed 30-day inflows) with Kruskal-Wallis test (H) for equality of means. “*” indicates that means of the various inflow categories differ significantly.

Inflow category (in acre-ft)	Seadrift	Mid-bay	Mesquite Bay	Espiritu Santo Bay
Dissolved orthophosphate				
Inflow ≤ 70,140	0.22	0.09	0.07	0.04
70,141 ≤ inflow ≤ 120,040	0.23	0.08	0.05	0.04
120,041 ≤ inflow ≤ 240,560	0.17	0.09	0.06	0.03
Inflow ≥ 240,561	0.15	0.12	0.08	0.05
Kruskal-Wallis H	8.52	7.82	7.57	0.93
Probability	$P < 0.04$ *	$P < 0.05$ *	$P < 0.06$	$P < 0.82$
Total phosphate				
Inflow ≤ 70,140	0.25	0.12	0.12	0.05
70,141 ≤ inflow ≤ 120,040	0.32	0.14	0.12	0.05
120,041 ≤ inflow ≤ 240,560	0.26	0.12	0.13	0.07
240,561 ≤ inflow	0.21	0.18	0.10	0.07
Kruskal-Wallis H	10.88	12.46	4.27	6.73
Probability	$P < 0.012$ *	$P < 0.006$ *	$P < 0.23$	$P < 0.08$

inflow categories based on the exceedance frequency quartiles for monthly surface inflow in Table 7.1.1. For example, the low inflow group of measurements consisted of samples taken when inflows in the preceding 30 days ranged from 0 to 70,140 acre-ft (100 to 75% exceedance frequency). The low-to-medium grouping had samples taken when 30-day inflows ranged from 70,141 to 120,040 acre-ft (75% to 50% exceedance frequency), and so forth. The Kruskal-Wallis non-parametric test was used to identify nutrient parameters for which the means among inflow groups were significantly different. Group means and test statistics are given for nitrogen in Table 7.2.3, phosphorus in Table 7.2.4, and organic carbon in Table 7.2.5. Additional tests were repeated with inflows summed over the 90 days preceding sampling. These results were essentially the same as the 30-day inflow tests and are not presented.

Statistically significant inflow associations generally indicate that higher nutrient concentrations occur with higher inflow volumes. Mid-San Antonio Bay shows a consistent positive response of nutrient concentration to inflows. Espiritu Santo Bay shows the least influence of inflows. All areas of the Guadalupe Estuary show higher levels of total organic carbon as inflows increase. This suggests high flows stimulate heterotrophic metabolism of the estuary. In the upper estuary area near Seadrift, there is some indication that high nitrogen and phosphorus levels occur most often during periods of intermediate inflows.

Perhaps the pattern of variation within the estuary is consistent with the idea that there is a balance between the nutrient enrichment from inflows and the destabilizing effect of high flows. In the mid-bay region, for example, biological processes may keep nutrients tied up when conditions are stable (with low to moderate inflow). But the biota may not be able to assimilate available nutrients during high flows when they are responding to changes in salinity. Thus available nutrient levels would be higher.

Table 7.2.5. Average total organic carbon (mg/l) among inflow quantity categories (summed 30-day inflows) with Kruskal-Wallis test (H) for equality of means. “*” indicates that means of the various inflow categories differ significantly.

Inflow category (in acre-ft)	Seadrift	Mid-bay	Mesquite Bay	Espiritu Santo Bay
Inflow ≤ 70,140	5.86	5.29	5.92	5.20
70,141 ≤ inflow ≤ 120,040	6.84	7.06	6.40	3.84
120,041 ≤ inflow ≤ 240,560	7.97	7.72	7.52	6.99
Inflow ≥ 240,561	9.68	10.70	9.19	8.93
Kruskal-Wallis H	9.01	9.47	8.27	10.28
Probability	$P < 0.03$ *	$P < 0.02$ *	$P < 0.04$ *	$P < 0.02$ *

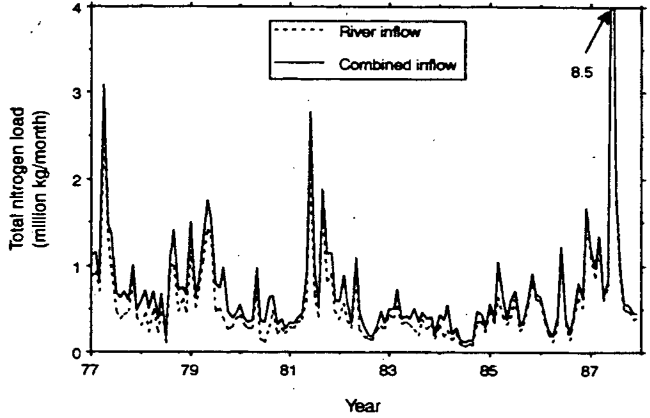


Figure 7.2.5. Total nitrogen loading to the Guadalupe Estuary from the San Antonio and Guadalupe rivers (river inflow) and all drainage basin sources (combined inflow) during the 1977 to 1987 period.

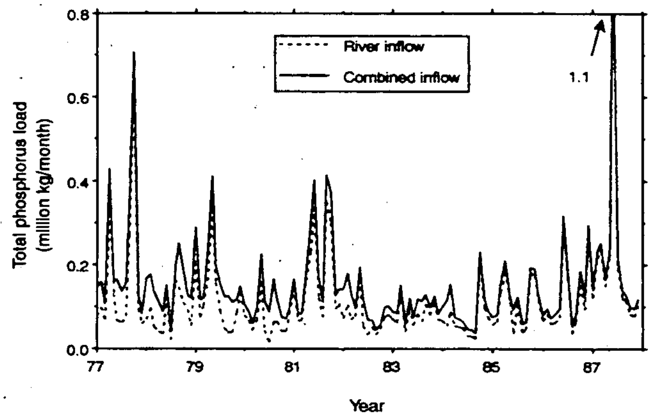


Figure 7.2.6. Total phosphorus loading to the Guadalupe Estuary from the San Antonio and Guadalupe rivers (river inflow) and all drainage basin sources (combined inflow) during the 1977 to 1987 period.

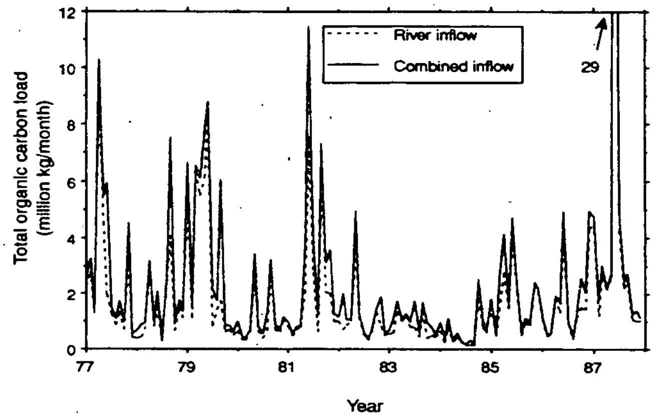


Figure 7.2.7. Total organic carbon loading to the Guadalupe Estuary from the San Antonio and Guadalupe rivers (river inflow) and all drainage basin sources (combined inflow) during the 1977 to 1987 period.

Variation in Nutrient Loading

Data on nutrient loading for the Guadalupe Estuary were presented in Section 4.3. Additional information is presented here on the variation and long-term changes in this loading. The most complete data on nutrient loading for this estuary is available for the most important sources, the Guadalupe and San Antonio rivers. Nutrient contributions from the San Antonio and Guadalupe rivers are summed and compared to the combined input from all drainage basin sources in figures 7.2.5, 7.2.6, and 7.2.7. For samples without total Kjeldahl-N measurements, total nitrogen was estimated from a regression based on nitrate-N ($R^2 = 0.68$) developed from pooled bay data. The average monthly contributions of these two rivers to the total surface loadings of total nitrogen, total phosphorus, and total organic carbon are 77%, 72%, and 78%, respectively.

Trends in nutrient loading. Trends in nutrient concentrations and loadings from the two main gaged sources, the San Antonio and Guadalupe rivers, were investigated using the nonparametric Kendall tau test described earlier, based on three-month seasons. Estimation of loading trends in combined drainage basin inputs is hampered by inconsistencies in return flow and rainfall runoff records.

Table 7.2.6 presents the results of the trend analysis. As previously stated, only "z" values greater than 1.96 or less than -1.96 are significant at $P < 0.05$. Only two categories display significant trends: a decline in total organic carbon loading from the San Antonio River and a decline in ammonia nitrogen loading from the combined river input. These trends seem problematic, since they are not reflected in similar trends in other categories, as might be expected. The indicated trends also do not correspond to trends of concentrations within the lower reaches of the two rivers. Given the variability of the data and the relatively short time period included in the analysis, these results mainly indicate that general nutrient loading to the estuary appears to have remained stable over the period of interest.

7.3. NUTRIENT BUDGET FOR THE GUADALUPE ESTUARY

Introduction

The delivery of nutrients to an estuary and losses of nutrients from the estuary can be described as a

Table 7.2.6. Trends in nutrient loading to the Guadalupe Estuary, 1968 to 1987, from gaged sources. For each parameter, the first line gives the trend statistic "z." The trend is significant when $|z| > 1.96$ ($P < 0.05$). The second line gives the % change per year estimated from least squares fit.

Parameter	San Antonio River	Guadalupe River	Combined rivers
Nitrate-N			
z value	0.92	-1.84	0.26
Trend (%/yr)	0.88	-2.20	0.43
Ammonium-N			
z value	1.72	-0.96	-2.29
Trend (%/yr)	0.31	-7.30	-8.50
Total N			
z value	0.16	-0.56	0.05
Trend (%/yr)	0.06	-0.21	0.23
Total P			
z value	0.11	-1.08	-1.91
Trend (%/yr)	0.65	-1.70	-1.80
Total organic carbon			
z value	-2.03	-0.29	-0.65
Trend (%/yr)	-8.7	-1.50	-2.10

budget in which both the sources of input and the paths of loss are identified. Changes in the internal amounts stored in the system can also be considered, since material can be stored during some seasons to fuel production during other seasons. A balance of inputs and losses demonstrates several things. The procedure tests our knowledge of the system; if the amount of materials which cannot be accounted for is large, then important sources, sinks, or processes have been overlooked. The comparative magnitudes of components can be related to sensitivity of the system to changes in those components. The relative importance of physical and biological processes within the budget may forecast the ability of the system to respond to increased or decreased inputs.

Analytical Procedure and Data Sources

Nitrogen, phosphorus, and carbon budgets for the Guadalupe Estuary are presented for a dry year, 1984, and for a year of extremely high input of fresh water to the estuary, 1987. These two sets of conditions were chosen to span the range of fluvial input. In most years, conditions fall between these extremes, and the budgets for those years can be expected to fall within the range presented here. Nutrient budget results demonstrate the immediate and longer-term influence (through consideration of sediment storage) of freshwater inflows on the nutrient supply available for biological production.

Nutrient budget data. The analysis of nutrient loading rates, presented in Section 4.3, provides much of the data on which nutrient budgets can be developed. Bay water concentration data, used in calculations of intrabay exchanges, were collected from TWDB and TWC databases. These data were averaged over depth and among neighboring stations for selected bay areas. Balances for nitrogen, phosphorus, and carbon are based on unfiltered water samples to include the mass associated with detrital particles and small planktonic organisms with high rates of turnover. Total nitrogen is defined here as the sum of nitrate-N, nitrite-N, and total Kjeldahl-N. During the study years, total nitrogen, total phosphorus, and total organic carbon were measured at locations within the estuary approximately every other month. Values were interpolated from adjacent months to represent missing concentrations.

Volumes flowing into and out of the estuary from the Gulf and neighboring bays were developed from results of the TXBLEND model (Section 8.4). TXBLEND simulates water movement in the two horizontal dimensions based on the finite element method, with triangular elements and linear basis functions. Water circulation calculations are based on conservation of mass and momentum, and inputs include tidal elevations, freshwater inflows, precipitation, evaporation, and wind. The system of bays including Matagorda, San Antonio, and Aransas were modeled together to accurately estimate the exchange among them, as well as the exchange with the Gulf. The simulation of an entire year of estuary hydrodynamics for both high and low inflow years was based on daily freshwater inflows and meteorology, and on hourly tides. The 1984 simulation employed calculated tides, since measured tides were not available. Results of TXBLEND included daily volumes exchanged at each pass and channel connecting system components. These were summarized into monthly volumes coming in, going out, and the resulting net flow, for each pass controlling exchange with the Guadalupe Estuary. Tables 7.3.1 and 7.3.2 present these volumes as millions of cubic meters per month.

Tidal exchange in the budget. A major assumption applied in this mass balance concerns the mixing of tidal flow with recipient waters. It was assumed that some mixing occurs between bay waters and inflowing Gulf tidal waters (and vice versa), such that there is some entrainment of those Gulf waters. Following Smith's (1985) analysis of mixing between Corpus Christi Bay and the Gulf, an assumption of 50% tidal water entrainment was used in this analysis. For computation, net flow was separated from the volume moving through a pass, and 50% of the remainder was assumed to be the tidal volume transferred through the pass. This calculation is necessary to fully account for input from and losses to the Gulf and bays neighboring San Antonio Bay. The relationship between tidal exchange and net flow between bays is illustrated in Figure 7.3.1.

Table 7.3.1. 1984 monthly tidal inflows, outflows, and net flows at Guadalupe Estuary exchange points, in millions of m^3. Inflows on each flood tide were summed to give monthly inflows, outflows on each ebb tide were summed to give monthly outflows, and the daily inflows less the daily outflows were summed for the monthly net flow figures.

Type and location	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
Inflows												
Saluria	598.89	485.71	515.08	486.34	565.32	494.27	516.56	475.33	547.46	499.88	561.73	527.56
Big Bayou	150.80	117.43	126.72	122.23	144.60	120.45	124.52	113.63	145.13	132.34	144.52	135.76
Port O'Connor	138.17	97.56	96.88	87.40	94.78	67.67	84.03	71.86	125.19	98.84	117.25	105.56
Cedar Dugout	24.85	123.05	130.46	141.13	152.14	195.54	170.09	162.49	60.33	116.51	77.67	130.15
ICWW/ Bludworth	21.42	102.90	112.79	122.57	131.60	171.74	146.85	141.46	51.63	101.80	67.66	112.43
Outflows												
Saluria	506.48	492.68	506.67	532.09	580.13	613.41	565.13	515.52	458.50	568.94	507.92	552.78
Big Bayou	106.32	113.76	119.10	126.26	137.30	144.92	131.14	120.09	99.35	134.15	121.02	125.10
Port O'Connor	71.03	99.06	105.91	110.07	117.6	133.55	114.0	104.83	66.37	105.61	87.82	105.69
Cedar Dugout	236.58	128.94	140.48	122.67	80.34	9.24	48.37	32.79	186.15	125.69	160.89	119.77
ICWW/ Bludworth	214.85	114.87	125.78	109.24	72.65	7.62	45.49	30.13	167.47	114.09	145.77	107.43
Net Flows												
Saluria	92.41	-6.97	8.41	-45.75	-14.81	-119.14	-48.57	-40.19	88.96	-69.06	53.81	-25.22
Big Bayou	44.48	3.67	7.61	-4.03	7.30	-24.47	-6.62	-6.46	45.78	-1.81	23.51	10.66
Port O'Connor	67.15	-1.50	-9.03	-22.67	-22.89	-65.88	-30.02	-32.97	58.82	-6.77	29.43	-0.13
Cedar Dugout	-211.73	-5.88	-10.02	18.46	71.80	186.30	121.72	129.70	-125.82	-9.18	-83.22	10.38
ICWW/ Bludworth	-193.43	-11.97	-12.99	13.34	58.95	164.12	101.36	111.34	-115.84	-12.28	-78.11	5.00

Water balance. A balance of water volumes gained and lost by the estuary forms the basis for materials balances. Inflows accounted for included freshwater inflows from gaged, ungaged, and return flows; direct precipitation on bay waters; net flow between the connected bays; and tidal input into Espiritu Santo and Mesquite bays. Diversions were subtracted from surface flows and evaporation was included as a loss. Tidal input to Espiritu Santo Bay included simulated flows through Saluria Bayou, Big Bayou, and through the ICWW at Port O'Connor. Exchanges to Mesquite Bay included flows through Cedar Dugout and the ICWW at Bludworth Island. Cedar Bayou was not open during the years simulated. Table 7.3.3 summarizes the water balance, presenting monthly freshwater inflow, evaporation, and the totals of these and other inputs and outputs. As a result of error accumulation over the long duration of simulation, some flows are not accounted for in the balance. In terms of the total input volume, this error in the water balance was 6% for the 1984 simulation, and 5% for 1987. For purposes of the nutrient balances, the monthly balance remainder was distributed between tidal outflows to Matagorda and to Aransas bays, proportionate to the original ratio of outflows to those bays. This gave a net flow balance close to zero.

Biogeochemical processes in the budget. The product of the nutrient concentrations in the water and the volumes of water included in the water balance produces the terms of a balance of nutrients accounted for in water transport into and out of the estuary. Losses of nutrients to fisheries harvests, burial in bay sediments, and—for nitrogen—denitrification, were included to complete the material balances.

Denitrification results in loss of biologically available nitrogen from the estuary as nitrogen gas. This process typically takes place in low oxygen environments, such as are found in the sediments. Estimates of rates of denitrification were made in investigations performed in San Antonio Bay (Benner and Yoon 1989). An average rate was used in this balance, even though the rate should vary with temperature and nutrient availability.

Table 7.3.2. 1987 monthly tidal inflows, outflows, and net flows at Guadalupe Estuary exchange points, in millions of m^3. Inflows on each flood tide were summed to give monthly inflows, outflows on each ebb tide were summed to give monthly outflows, and the daily inflows less the daily outflows were summed for the monthly net flow figures.

Type and location	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
Inflow												
Saluria	606.84	613.19	615.16	406.55	703.45	163.34	485.58	462.75	674.36	632.32	641.24	580.04
Big Bayou	150.50	160.11	168.17	88.61	180.23	29.54	114.02	95.67	181.11	165.27	171.85	134.07
Port O'Connor	98.65	116.70	107.76	44.28	80.27	7.18	44.26	30.66	113.93	94.09	111.67	56.14
Cedar Dugout	48.48	76.32	69.13	202.78	213.57	39.53	174.70	280.53	93.28	125.47	128.98	223.90
ICWW/ Bludworth	43.12	65.04	59.76	174.23	182.97	36.10	155.01	240.65	80.75	107.92	109.76	190.66
Outflow												
Saluria	721.75	549.93	747.99	702.34	534.63	1,787.58	818.86	672.90	554.55	646.29	649.88	789.61
Big Bayou	185.17	135.19	188.78	183.04	123.38	551.27	235.49	190.57	136.00	164.55	163.11	209.18
Port O'Connor	139.30	91.89	117.45	177.20	89.67	496.18	199.57	182.49	82.96	113.21	101.64	189.72
Cedar Dugout	337.14	231.54	287.54	66.48	31.95	613.68	37.78	10.79	151.25	118.51	189.03	108.21
ICWW/ Bludworth	298.82	207.20	259.46	62.34	31.18	541.66	38.21	11.91	142.94	114.67	173.75	100.95
Net Flows												
Saluria	-114.91	63.26	-132.83	-295.79	168.82	-1,624.24	-333.28	-210.15	119.82	-13.97	-8.63	-209.57
Big Bayou	-34.67	24.92	-20.61	-94.42	56.85	-521.73	-121.46	-94.90	45.11	0.71	8.74	-75.11
Port O'Connor	-40.65	24.81	-9.69	-132.92	-9.40	-489.00	-155.30	-151.83	30.97	-19.12	10.03	-133.58
Cedar Dugout	-288.66	-155.22	-218.41	136.30	181.62	-574.15	136.92	269.74	-57.98	6.96	-60.05	115.69
ICWW/ Bludworth	-255.70	-142.16	-199.70	111.90	151.79	-505.56	116.80	228.74	-62.19	-6.75	-63.99	89.71

Fisheries harvests were available on an annual basis for this estuary. For this analysis, the harvest rate was distributed equally across the months. Numbers for 1984 and 1987 harvests are presented as part of the inflow-harvest regression analysis (Section 6.9). Some additional assumptions were made to arrive at harvest losses: recreational harvest equals commercial harvest from the bay; offshore harvest from zone 21 can be considered loss to the estuary when the harvest is prorated by bay area to San Antonio and Matagorda bays (SAB receives 36%); the mass of a shrimp leaving the bay is 10% of its mass when harvested in the Gulf; from species composition data in Zison et al. (1978), nitrogen is 2.5%, phosphorus is 0.63%, and carbon is 25% of the harvest wet weight.

Burial in bay sediments removes some nutrients from cycling within the system. Most of the sediment activity which can result in a flux of dissolved nutrients from the sediment to the water column or which results in assimilation of material into the benthic biota takes place in the top few centimeters of bay mud. A generous 10 cm active depth was assumed here. Considering only the dissolved fraction and fine particulates within this layer as available, sediment carbon is approximately 1% by dry weight and nitrogen is 0.1% (Patrick L. Parker, University of Texas Marine Science Institute, unpublished data; pers. comm. 1990). Based on a phosphorus-to-nitrogen ratio of 0.82:1 calculated from TNRCC Statewide Monitoring Network sediment samples, the sediments contained approximately 0.082% phosphorus, dry weight. Steed (1971) presented data that indicates that sediments at a 20-22 cm depth in upper San Antonio Bay averaged 34% less total organic carbon (TOC) than surface levels, and that below this depth TOC content did not change appreciably. This can be translated into a carbon loss rate in conjunction with information on the rate of burial of surface sediments, which should be close to the average rate of sediment deposition. Shepard (1960) estimated an average sedimentation rate of 0.8 ft per hundred years, or 0.20 cm per year. From the above assumptions plus an assumed 50% water content of the top 10 cm sediments, a general rate of carbon burial can be calculated. For the purpose of constructing nutrient budgets here, ratios of surface-to-

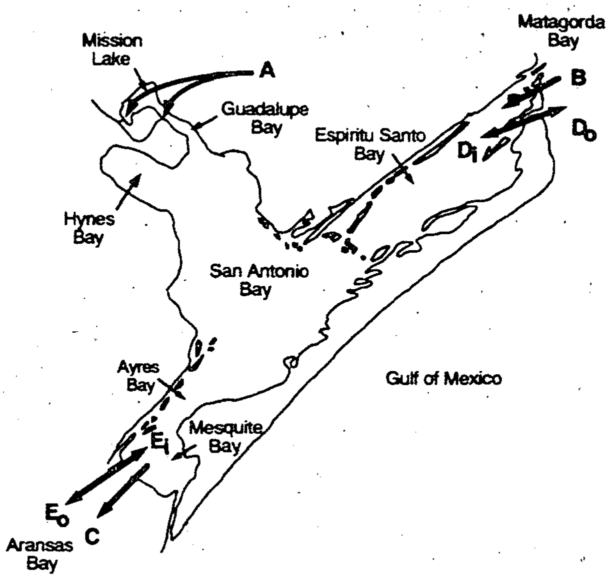


Figure 7.3.1. Conceptual relationship of tidal flows and net flows used to calculate water exchange and balances. Various flows are defined as follows: A = freshwater inflows; B and C = net flows; D_i and E_i = tidal inflows; D_o and E_o = tidal outflows; and entrained tidal volumes = 0.5^*D_i, 0.5^*D_o, 0.5^*E_i, and 0.5^*E_o. In the above scenario, total flow from Aransas Bay = 0.5^*E_i, and total flow to Aransas Bay = C + 0.5^*E_o.

subsurface phosphorus and nitrogen equal to the observed carbon ratio were used to extend the carbon burial rates to these other elements. Burial rates were assumed applicable to the entire estuary.

Nixon (1987) argues that basing an estimate of short-term sediment deposition rate on long-term sediment accumulation may bias results. Long-term sediment history incorporates episodic large inputs of sediment from major floods and storms, and so may not truly indicate routine deposition rates.

Water column storage. The balances include not only rates of input and loss, but also a water column storage term. This represents the change in concentration of nutrients in the estuary water column from beginning to end of the study period. This term accounts for a net increase or decrease in nutrients held in the active system. For this analysis, changes in water column nutrient mass were based on bay-wide average concentrations and total bay volume.

Results of Nutrient Budgets

Nitrogen budget. Results for the basic total nitrogen balance for 1984 and 1987 are presented in Tables 7.3.4 and 7.3.5, respectively. A summary of the annual totals for nitrogen is presented in Table 7.3.6. Some general conclusions are suggested in this summary. The total nitrogen loading from surface inflow sources in the high-flow year was approximately five times the loading in the low-flow year, but when tidal and rainfall inputs were also included, the years differed by a factor of only three. The mass of nitrogen exported by the estuary is also about three times higher in 1987 than in 1984. However, the proportion of input nitrogen exported from the estuary in high- and low-flow years is more similar, 58% and 69%, respectively. Losses of nitrogen to biological and geological sinks are significant, 10% of 1987 inputs and 22% of 1984 inputs. These estimates of loss seem conservative in comparison to

Table 7.3.3. Water balance summary for the Guadalupe Estuary, including tidal exchanges derived from simulation for years 1984 and 1987, in millions of m^3.

Type and location	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
1984												
Freshwater input	156.1	71.2	109.4	38.2	83.3	24.2	38.0	70.8	62.9	297.6	128.1	103.0
Evaporation	33.8	46.8	53.5	72.8	86.4	103.7	112.3	113.4	94.7	62.9	57.4	40.5
Total in	719.8	493.3	563.3	453.0	605.8	602.0	540.9	550.4	522.4	764.0	625.0	590.5
Total out	770.2	486.0	510.5	536.4	509.2	559.1	494.7	441.5	609.8	573.8	592.4	512.7
Balance[a]	-50.4	7.3	52.8	-83.4	96.6	42.9	46.2	108.9	-87.4	190.2	32.6	77.7
1987												
Freshwater input	416.5	372.3	394.3	169.9	290.2	2,557.7	620.0	374.9	218.3	163.0	185.0	126.9
Evaporation	32.0	29.1	52.3	72.2	68.1	81.5	103.6	129.1	109.9	80.3	50.8	27.8
Total in	947.8	995.3	931.7	702.7	1,228.9	3,191.1	1,250.4	1,107.4	803.3	664.6	752.3	817.4
Total out	1,208.4	756.6	1,091.2	857.3	405.4	3,852.5	969.9	762.8	593.9	594.8	700.7	907.9
Balance[a]	-260.5	238.8	-159.5	-154.5	823.6	-661.4	280.4	344.7	209.4	69.8	51.5	-90.6

[a] Balance = total in - total out

estimates for other estuaries (Seitzinger 1988; Nixon and Pilson 1983), so they may be underestimates. In terms of drainage basin sources only, biogeochemical sinks account for 16% of 1987 nitrogen input and 52% of 1984 input.

The bay has physical and biological mechanisms which can retain some surplus inputs. This has important implications for the ability of the estuary to maintain productivity in spite of large fluctuations in nutrient supply. Compare the two years with respect to the bottom line in Table 7.3.6. In 1984, only 9% of input nitrogen was remaining and could be accounted for as storage in animal, plant, or sediment active pools. In 1987, approximately 33% of input nitrogen was remaining and not accounted for in transport or losses. Either there are significant losses of nitrogen which were not identified in the budget, or this nitrogen is retained in some storage available for future system use. Storage of nutrients in upper estuarine sediments and estuarine biota is a natural result of biogeochemical processes. The nitrogen balance indicates that only in high inflow years is the storage likely to be a large proportion of inputs. Finally, the figures in Table 7.3.6 demonstrate that nitrogen processes occurring in the Guadalupe Estuary cannot be considered solely in terms of the flow from two rivers. Gains from waters neighboring the estuary as well as losses to them were more than twice as great in 1987 as in 1984. This may reflect generally higher inflows to the entire middle Texas coast during 1987 as compared with 1984.

Phosphorus budget. The annual summary of the phosphorus budget calculations is presented in Table 7.3.7. In some ways phosphorus behaves differently than nitrogen in the estuary. The mass of phosphorus entering and leaving the estuary was two to three times greater during 1987 than during 1984. However, only 40 to 50% of this incoming material left the estuary. Phosphorus concentrations in the estuarine water column can be greatly influenced by exchanges to and from a fraction bound to clay particles of the surface sediment. The degree to which clays can adsorb phosphorus is related to pH and salinity of the overlying water. Pomeroy et al. (1965) have suggested that suspended clays and sediments can control the water column phosphorus concentrations within an estuary. It appears as though flux of phosphorus from the water to the sediment may be significant in the Guadalupe Estuary. With respect to total inputs, the amount of phosphorus remaining and not

Table 7.3.4. 1984 monthly total nitrogen balance terms for the Guadalupe Estuary; units are millions of gm.

Load terms	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
Water-borne inputs												
Surface	439	366	543	206	275	144	109	145	124	471	434	280
Rain	28	5	8	0	19	2	12	21	22	64	16	8
Gulf & Matagorda	494	274	243	276	275	227	223	183	345	191	297	231
Aransas	22	62	59	138	244	351	196	169	49	55	34	63
Total input [a]	983	706	854	619	812	724	539	517	540	781	782	583
Water-borne outputs												
Gulf & Matagorda	189	231	277	318	436	503	387	380	239	426	304	354
Aransas	505	134	125	82	60	8	50	36	282	155	245	125
Total output [b]	694	365	402	400	496	511	437	416	521	582	549	479
Water storage	72	-133	71	-71	-28	44	44	-61	0	0	0	79
Balance [c]	217	473	381	290	344	169	58	162	19	199	233	24
Biogeochemical losses												
Denitrifying	126	126	126	126	126	126	126	126	126	126	126	126
Harvest	10.08	10.08	10.08	10.08	10.08	10.08	10.08	10.08	10.08	10.08	10.08	10.08
Burial	17.7	17.7	17.7	17.7	17.7	17.7	17.7	17.7	17.7	17.7	17.7	17.7
Remaining [d]	63	319	227	136	190	15	-96	8	-135	46	79	-130

[a] Total input = surface input + rain input + Gulf & Matagorda input + Aransas input

[b] Total output = Gulf & Matagorda output + Aransas output

[c] Balance = total input - total output - water storage

[d] Remaining = balance - denitrifying - harvest - burial

accounted for by transport or loss is large, 30% to 60%. The significant question may be how much of this phosphorus is buried and how much is stored temporarily in the active sediment pool.

Carbon budget. A complete carbon budget for the estuary would include exchange between atmospheric carbon dioxide, dissolved carbonates, and organic carbon fixed by primary production, as well as water column total organic carbon concentration. This has not been attempted for the Guadalupe Estuary, since the availability of carbon is generally not considered a controlling factor for estuarine productivity. A budget based on flows of total organic carbon (dissolved and particulate) can address some concerns, however. First there is the transport of materials into and from the estuary, in consideration of the heterotrophic metabolism that it can support. Secondly, the balance between organic carbon brought in by stream flow and the amount of carbon fixed by primary production describes the type of metabolism which predominates in the estuary. A summary of the organic carbon balance for the Guadalupe Estuary is presented in Table 7.3.8. High freshwater inflows bring in a much greater carbon input from terrestrial sources than low inflows, the 1987 amount was seven times the 1984 amount. However, when tidal exchanges are considered, the total inputs during 1984 were only slightly less than half the high inflow year amount. The amount of carbon exported from the system was similar both years in proportion to input amounts. During both high- and low-inflow years, export and other losses left deficits in the budget. The deficit remainder in the table relates only to the comparison of exchange at the openings to the system. Within the estuary carbon is gained and lost through other processes. The apparent deficits are small compared to an annual yearly primary productivity for the estuary, which is on the order of 250 million kg C (from data in MacIntyre and Cullen 1988). However, our knowledge of productivity, respiration, and changes in storage within the primary producer compartment of the system is limited. This hampers calculation of a full comparison of internal versus external contributions to the estuarine carbon budget.

Conclusions

The description of the gross balances of nutrient masses within the estuary is only a prelude to understanding the assimilation of nutrients into a healthy and productive biological community. Studies on several estuaries,

Table 7.3.5. 1987 monthly total nitrogen balance terms for the Guadalupe Estuary; units are millions of gm.

Load terms	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
Water-borne inputs												
Surface	1,189	968	1,349	645	813	7,087	1,862	909	558	542	457	448
Rain	13	29	4	0	22	43	27	17	11	11	17	5
Gulf & Matagorda	889	656	582	638	537	583	172	171	940	572	415	391
Aransas	144	15	102	420	539	197	378	670	67	145	146	416
Total input[a]	2,236	1,668	2,037	1,704	1,911	7,909	2,439	1,766	1,575	1,270	1,033	1,260
Water-borne outputs												
Gulf & Matagorda	736	531	581	959	221	3,500	1,259	1,059	510	478	365	588
Aransas	537	456	501	69	78	1,297	57	15	200	120	283	111
Total output[b]	1,273	987	1,082	1,028	1,298	4,797	1,316	1,074	709	598	648	699
Water storage	-143	-30	-30	101	101	-168	-168	-31	-31	79	79	-14
Balance[c]	1,106	710	984	575	511	3,279	1,291	723	897	593	307	575
Biogeochemical losses												
Denitrifying	126	126	126	126	126	126	126	126	126	126	126	126
Harvest	83.7	83.7	83.7	83.7	83.7	83.7	83.7	83.7	83.7	83.7	83.7	83.7
Burial	17.7	17.7	17.7	17.7	17.7	17.7	17.7	17.7	17.7	17.7	17.7	17.7
Remaining[d]	879	483	757	347	284	3,052	1,063	496	670	365	79	348

[a] Total input = surface input + rain input + Gulf & Matagorda input + Aransas input

[b] Total output = Gulf & Matagorda output + Aransas output

[c] Balance = total input - total output - water storage

[d] Remaining = balance - denitrifying - harvest - burial

Table 7.3.6. Summary of total nitrogen (TN) mass balance in 1984, the low-flow year, and 1987, a year of very high flows. Units are millions of gm nitrogen in the estuary as a whole.

Load terms	1984	1987
Water-borne inputs		
Surface	3,535	16,826
Rain	205	197
Gulf & Matagorda	3,258	6,545
Aransas	1,441	3,238
Total input [a]	8,440	26,807
Water-borne outputs		
Gulf & Matagorda	4,043	11,787
Aransas	1,808	3,723
Total output [b]	5,851	15,510
Water storage	19	-254
Balance [c]	2,569	11,551
Biogeochemical losses		
Denitrifying	1,512	1,512
Harvest	121	1,004
Burial	212	212
Remaining [d]	724	8,822

[a] Total input = surface input + rain input + Gulf & Matagorda input + Aransas input

[b] Total output = Gulf & Matagorda output + Aransas output

[c] Balance = total input - total output - water storage

[d] Remaining = balance - denitrifying - harvest - burial

Table 7.3.7. Summary of total phosphorous (TP) mass balance in 1984, the low-flow year, and 1987, a year of very high flows. Units are millions of gm phosphorous in the estuary as a whole.

Load terms	1984	1987
Water-borne inputs		
Surface	1,136	2,923
Rain	62	60
Gulf & Matagorda	217	348
Aransas	138	356
Total input[a]	1,554	3,687
Water-borne outputs		
Gulf & Matagorda	402	1,389
Aransas	182	572
Total output[b]	584	1,961
Water storage	-126	31
Balance[c]	1,096	1,685
Biogeochemical losses		
Harvest	30	557
Burial	175	175
Remaining[d]	891	963

[a] Total input = surface input + rain input + Gulf & Matagorda input + Aransas input

[b] Total output = Gulf & Matagorda output + Aransas output

[c] Balance = total input - total output - water storage

[d] Remaining = balance - harvest - burial

summarized in Whitledge (1989a, 1989b), Armstrong (1987), and in other sections of this report, begin to establish the processes, rates, and interactions which are important to the ecosystem. Through the exercise of calculating nutrient loadings and balances, it is possible to appreciate the relative influence of inputs from all inflow sources on the estuarine ecosystem.

Comparison of system function during a dry year and a wet year is important given the common fluctuations in inflows Texas estuaries experience. Because mechanisms of geochemical and biochemical trapping are dependent on a variety of factors, the relationship between nutrient input and losses is not a simple function of inflow rate. Total nitrogen inputs to the Guadalupe Estuary in 1987 were 3.2 times the level in 1984. Yet the total export of nitrogen from the system in 1987 was greater than 1984 by a factor of only 2.7. Thus, the estuary accumulated nitrogen in 1987. The comparative carbon budgets show the same pattern. Proportionately more carbon was input to the estuary during the wet year than was exported. Apparently, differences in geochemical cycling between phosphorus and the other nutrients cause phosphorus to follow a different pattern. The ratio of phosphorus inputs in the wet over the dry year was 2.4, but the wet-to-dry-year export ratio was 3.1. During the dry year, the estuary was able to retain more of this nutrient.

Although the budget remainders include unknown error, it may be useful to compare the relative amounts of nutrient inputs which cannot be accounted for in transport out of the system or in other losses, between high- and low-inflow years. The amount which remains is potentially stored to supply future production. In this regard, it is noteworthy that the total remaining nitrogen in 1987 was as much as the total nitrogen input in 1984. Though considerable effort was involved in determining loading rates and rates of biological processes which affect the fate of nutrients in the estuaries, there is still much uncertainty in some of the estimates of rates used in budget calculations. In particular, continued work on the effect of temperature and salinity on biological and geochemical process rates, and work on interactions within the biological community that affect sediment-water exchanges, would increase the accuracy of the budgets.

Table 7.3.8. Summary of total organic carbon (TOC) mass balance in 1984, the low-flow year, and 1987, a year of very high flows. Units are millions of gm organic carbon in the estuary as a whole.

Load terms	1984	1987
Water-borne inputs		
Surface	8,524	57,914
Rain	0	0
Gulf & Matagorda	17,706	13,563
Aransas	7,953	12,549
Total input [a]	34,183	84,026
Water-borne outputs		
Gulf & Matagorda	21,446	47,067
Aransas	10,293	21,566
Total output [b]	31,739	68,633
Water storage	-69	-979
Balance [c]	2,513	16,372
Biogeochemical losses		
Harvest	1,212	22,140
Burial	2,124	2,124
Remaining [d]	-823	-7,892

[a] Total input = surface input + rain input + Gulf & Matagorda input + Aransas input

[b] Total output = Gulf & Matagorda output + Aransas output

[c] Balance = total input - total output - water storage

[d] Remaining = balance - harvest - burial

7.4 A MINIMUM NITROGEN LOADING CONSTRAINT FOR THE GUADALUPE ESTUARY

Introduction

A major concern in assessing effects of water resource developments on bays and estuaries is maintenance of biological productivity in those waters. Both quantity and quality of freshwater inflows affect estuary productivity. Dissolved nutrients pose a particularly important question: will an estuary receive insufficient or excessive amounts as a result of developments upstream? Because reservoirs are considered to be nutrient traps (Hannan 1979), one concern in planning a reservoir above an estuary is that it would deprive the estuary of nutrients. This concern can be addressed by estimating the potential characteristics of the reservoir and by calculating estuary needs. A methodology is discussed here for assessing the amount of nutrients needed by an estuary from freshwater inflow. Describing a minimum nutrient load as a constraint in the mathematical programming model is an appropriate way to address an estuary's nutrient needs in the context of other competing planning considerations.

Assessing the Limiting Nutrient

Nitrogen. For the Guadalupe Estuary, as well as for many other estuaries, nitrogen is considered the nutrient most likely to control production (Whitledge 1989a); therefore, nitrogen is an appropriate indicator nutrient. Ideally, an estimated minimum nitrogen loading level should be based on a relationship between nitrogen input and estuary productivity. However, there are not enough estimates of primary or secondary productivity for Texas estuaries to construct a relationship. Loading rates influence concentrations in the bays, but using bay concentrations alone to establish a minimum loading would be problematic. Concentrations also depend on the bay retention time and biological processes, such as uptake and regeneration rates, which are strongly influenced by temperature and other factors. Instead of defining the relationship between inflow and nitrogen concentration, a relationship can be developed between inflow and the amount of nitrogen available to the estuarine food chain. Estimates of the proportion of inflow nitrogen potentially available for production can be derived from calculations of a nitrogen budget for the estuary. These data lead to estimations of minimal loading needed to sustain estuary productivity.

Nutrient budget computations completed for the estuary (presented in Section 7.3) provide information on the rates of nitrogen losses to the estuary against which rates of inputs can be balanced. The estuary loses nitrogen to interbay transport, through fisheries' harvests, and through denitrification and burial in the sediment.

Sediment storage. The sediment serves as both sink and source in the estuarine nitrogen cycle (Section 5.5). Guadalupe Estuary sediments support an average flux of NH_3 back into the water column of 1.3 billion gm per month (Benner and Yoon 1989). Other processes deposit nitrogen on the sediments from the water column. Indeed, in shallow estuaries such as those of the Texas coast, the loop from sediment to water column and back can drive much of the system productivity (Flint et al. 1983). The point is that there is nutrient storage in the estuary and this is a dynamic storage which maintains productivity levels. Assuming an even distribution of sediment types in the estuary, a 10 cm active depth and a 0.1% nitrogen content by dry weight (Section 7.3), the sediment nitrogen pool in this estuary can be estimated as 2.2 billion gm. This is between one and two orders of magnitude larger than average monthly nitrogen loading or average bay water column nitrogen content (data in Section 7.3; estuary concentration from Section 7.2 multiplied by estuary volume). This means that the active sediment layer has some capacity to provide nitrogen to the water column and thus sustain bay productivity between episodes of nitrogen delivery.

This important, interactive nutrient storage in estuarine sediments is, however, finite. Burial and denitrification deplete the sediment nitrogen store. An estimate of the average rate of denitrification which removes fixed nitrogen from the sediments is 126 million gm per month for the entire bay (Benner and Yoon 1989). Burial could remove nitrogen from the active sediment layer at a rate of approximately 35 million gm per month (Section 7.3). Under the assumption that these rates remain constant, the above estuarine pool of sediment nitrogen would last just 14 months without replacement.

Although there are uncertainties associated with these estimates of rates of loss, it is apparent that the sediment's capacity to maintain bay nutrient levels is on the order of one year without renewal. An approach, then, to defining a minimal nitrogen loading regime would be to determine a yearly loading rate which would maintain the average sediment nitrogen storage now observed. Unfortunately, the relationship between freshwater inflows and system nitrogen storage is difficult to assess directly. In particular, very few measurements of sediment nitrogen content are available, and the variation among these measurements is typically much higher than the amount of change which would be expected to result from annual variation in inflow volumes. Therefore, the nutrient budget approach is used to derive estuary system nitrogen storage. This is an indirect estimate, based on the assumption that a large proportion of nitrogen which is not removed by transport, harvest, burial, or denitrification remains stored in the system.

Minimum Nitrogen Loading Requirement

Estimate of inflow volume. From the contrasting nitrogen budgets constructed for the estuary for high-inflow and low-inflow years (Table 7.3.6), we can derive a first-order estimate of the inflow volume needed to maintain nutrient levels in the estuary and the sediment nutrient reservoir. The nitrogen budget demonstrated that the estuary was only marginally accumulating surplus nitrogen in the low-flow year, but had a substantial excess for storage in the high-flow year. Table 7.4.1 gives these figures and presents a simple linear relationship between inflow and reserve nitrogen. This relationship describes the amount of nitrogen stored in the system to provide for continued production and defines an inflow constraint. A yearly inflow which would not result in estuarine nitrogen depletion (that is, remaining N is greater than zero) should be more than 286,000 acre-ft. This flow is approximately the 95% yearly exceedance flow.

Table 7.4.1. The relationship between surface inflow and dissolved nitrogen remaining in the Guadalupe Estuary after budget calculations.

Year	Inflow (acre-ft)	Nitrogen remaining [a] (10^6 gm)
1984	708,260	724
1987	5,430,970	8,822

[a] Using the two points in the table to define a line, N remaining = -490.8 + 0.0017148 * inflow

Limitations to the method. It is possible that the relationship between inflow and remaining nitrogen is nonlinear, and it would be desirable in a real application of this approach to derive a constraint from more than two data points. Additional data points can be calculated in the manner described in Section 7.3, using flow results from the circulation model and measured values from other sources. However, basing a constraint on nutrient budget results establishes a good foundation for the constraint, since many aspects of estuarine function are incorporated into nutrient budget results.

7.5 SEDIMENT LOADING IN THE GUADALUPE ESTUARY

Sediment History of the Guadalupe Estuary

When sea level began to rise at the end of the late Wisconsin glaciation 18,000 years BP (before present), the head of today's San Antonio Bay was an eroding river valley that extended inland about 8 km (5 mi) northwest of Green Lake, close to the boundary dividing Calhoun and Victoria counties. Matagorda Island, as we know it today, did not exist; the river valley extended out into what is now the Gulf of Mexico. The bottom of the old river channel, where it crosses the present site of Matagorda Island, was 24 m (80 ft) below today's sea level (Shepard and Moore 1960). With the rise of sea level, waters of the Gulf of Mexico entered the river valley about 9,500 years BP (Shepard and Moore 1960), and the waters of the Gulf approached their current level about 2,800 years BP (McGowen et al. 1976b). As sea level rose and stabilized, Matagorda and San Jose islands grew from a series of small islands that formed when sand was transported landward (McGowen et al. 1976b).

Since Gulf waters entered San Antonio Bay, enough sediment has been transported into the bay from the Gulf and the contributing rivers to reduce the depth of the bay to an average of 1.4 m (4.6 ft) below today's sea level, with the deepest area less than 3.7 m (12 ft) (Diener 1975). Sediment transported by the Guadalupe and San Antonio rivers allowed the river delta to prograde (expand into the bay) some 24 km (15 mi) to its present location (McGowen et al. 1976b). The influx of sediment has varied from none, at the beginning of the period of sea-level rise, to amounts greater than today's influx, during occasional periods of wetter conditions such as the Little Ice Age that occurred between the 15th and 19th centuries.

The portion of the Guadalupe Delta that protrudes into San Antonio Bay below Mission Lake (Figure 7.5.1) developed over the past 2,000 years (Shepard and Moore 1960). It was built as a series of small adjacent lobes that formed and melded together (Section 4.4). The area of active delta building was always much smaller than the entire delta that exists today. At one time, the Guadalupe River ran all the way to the southeastern tip of the delta; an old river channel can still be seen in photos and maps. During the past century, most delta-building has occurred in the vicinity of the north and south forks of the river, in the lower delta. In 1935, Traylor Cut opened through the eastern bank of the river into Mission Lake; the cut provided the opportunity for the river to begin building a new subdelta lobe into the lake.

During a 20-year period ending in the 1970's, logjams directed about two-thirds of the water and sediment through Traylor Cut; the remainder flowed to the lower portion of the delta (Morton and McGowen 1980). When the logjams were removed, the distribution of water and suspended sediment between Traylor Cut and the lower portion of the delta was about equal. During periods of normal flow, most of the water flowing into the lower portion of the delta moves through the south fork of the river due to large accumulations of hyacinth in the north channel. A road along the western side of the Guadalupe River and the natural river bank and levee blocks most water and sediment from reaching the western part of the delta and Hynes Bay, except during major floods.

Sediment provides two ingredients to the Guadalupe Estuary: nutrients attached to the sediment particles; and building materials for the delta, adjacent marshes, and bay bottoms. Nutrient requirements for inflows have been considered in sections 7.3 and 7.4. This section will focus on sediment as a construction material that provides necessary physical structure and bathymetry for estuaries. Sediment in deltas and the upper portions of estuaries is subject to the natural processes of transport, deposition, resuspension, erosion, and subsidence. These processes are difficult to quantify, and some have high day-to-day variation. While the sediment load carried to the estuary during the past 40 years is known, sediment loading over the past 10,000 years has varied widely, so it is difficult to determine how today's sediment loading compares with earlier sediment input. Bay-filling and subdelta-building processes can be described qualitatively and a few estimates of historical rates for these processes are available, but quantitative relationships that equate sediment loading with the area or rate of growth of delta and shallow-water habitat do not currently exist. Therefore, it is necessary to take a simple approach so that a quantitative relationship between sediment loading and physical structure can be created for use in an analytical model for assessing inflow requirements.

River-borne Sediment Delivered to San Antonio Bay

Quantity delivered. The sediment load of rivers consists of two components: suspended sediment, the fine material carried in the water column; and bedload, the coarse particulate matter that moves along the bottom. Suspended sediment dominates the sediment load of the Guadalupe and San Antonio rivers; bedload is less than 3% of the total load (Morton and McGowen 1980). The majority of the rivers' sediment load is clay, with a lesser amount of fine silt. For this analysis, bedload is estimated to be 2% of the suspended load. Since suspended load has been the only component systematically measured in these river systems, the suspended values have been increased by 2% to estimate total sediment load for the analysis.

Table 7.5.1. Annual quantity of sediment that would be required to offset the relative sea-level rise of the lower Guadalupe Delta.

Equation number	Description and equations
1	Sediment required for lower Guadalupe Delta to offset relative sea-level rise. Where: Rate of relative sea-level rise = 8 mm/yr = 0.008 m/yr Lower delta area = 2,556 ha 0.008 m/yr * 2,556 ha * 10,000 m^2/ha = 204,480 m^3/yr

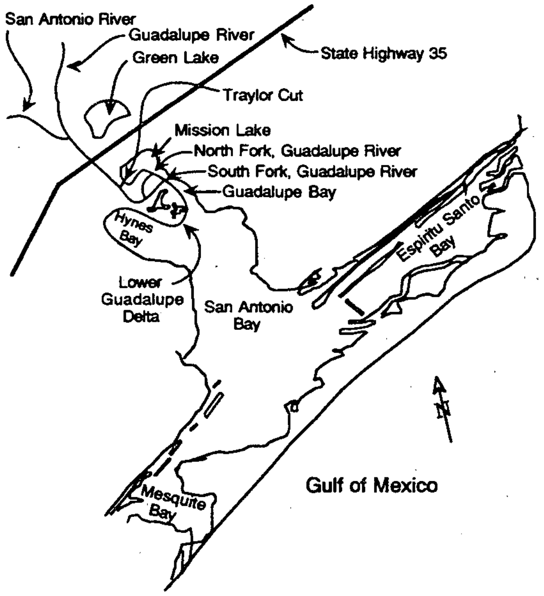


Figure 7.5.1. Map of the Guadalupe Estuary and delta.

Particulate organic material is transported by the Guadalupe and San Antonio rivers and measured as part of the sediment load. During normal river flow, the particulate organic material ranges from 1 to 22% of the total load, with an average of 10% (Childress et al. 1975). Steed (1971) found that 11.5% of the particulate organic carbon that was transported into the Guadalupe Estuary by the river was lost to the bay sediment. Steed also found that the surface sediments in Mission Lake, Guadalupe Bay, and upper San Antonio Bay contain 1.5% or less organic carbon, and assumed that little of the organic content of the surface sediments came from primary production in the water column. Recent studies in San Antonio, Corpus Christi, and Nueces bays have shown that microphytobenthos—algae mixed into the sediment—are present in the upper few centimeters of bay deposits (MacIntyre 1988; MacIntyre and Cullen 1988; Montagna and Yoon 1989). If the average particulate organic content of the sediment load from the rivers were 10%, and 11.5% of this organic material were deposited in the sediment, the organic content of newly deposited sediment would be about 1.15%. Microphytobenthos could constitute the remaining 0.35% (= 1.5% - 1.15%) of the sediment organic matter. Thus, microphytobenthos could contribute up to 23% [= (0.35% / 1.5%) * 100%] of the organic content of the sediment, consistent with Steed's observation that only a small portion of the organic content comes from production in the water column.

The quantitative relationship between annual gaged inflow and annual sediment load is shown in Figure 7.5.2. This relates the annual sediment load and gaged flow at the San Antonio and Guadalupe rivers' stream gages at Goliad and Victoria. The data is taken from the USGS stream gage data and TWDB sediment load data collected from 1946 through 1985.

There are 30,554 ha (75,500 acres) of ungaged area below the gaging stations and around the estuary which may contribute to the total sediment load. Reineck and Singh (1980) note that the sedimentation rate on land from rivers that primarily carry suspended sediment is low, but some deposition occurs in bank benches and on the floodplain during high water. Best (1986) noted that deposition may occur at the confluence of two rivers such as the San Antonio and Guadalupe. The processes of deposition upon and erosion from the floodplain tend to offset each other. Since there is a lack of good quantitative information for either process in the lower part of the Guadalupe and San Antonio river basins, we assume that the sediment eroded from the ungaged area and carried to the rivers equals the amount deposited in the floodplain and along the river bank. For this analysis, the ungaged area contributes no sediment to the delta, and the total load carried by the river is represented by the measurements at the stream gages.

From data collected by the TWDB from 1946 through 1985, the average annual total sediment load was 815,805 m^3/yr (661 acre-ft/yr), with a standard deviation of 619,416 m^3/yr (502 acre-ft/yr). The measurements varied from a low of 90,220 m^3/yr (73 acre-ft/yr) to a high of 3,053,646 m^3/yr (2,476 acre-ft/yr). Details of where deposition has occurred are known only indirectly from studies that have compared delta area and bathymetry over a span of time (Shepard and Moore 1960; Donaldson et al. 1970). Note that the sediment volumes given above are presented on the basis of sediment that has reached its ultimate consolidation, with a density of 1,121.4 kg/m^3 (70 lb/ft^3).

Lower portion of the delta. The lower delta below Traylor Cut (Figure 7.5.1) that protrudes into San Antonio Bay has an area of 2,556 ha (6,315 acres). From marsh inundation modeling studies (Hauck et al. 1976; TDWR 1981a), the southern portion of the lower delta below the south fork of the Guadalupe River rarely receives flood waters from the river. Mission Lake, Guadalupe Bay, the lower Guadalupe Delta, and the upper portion of San Antonio Bay are experiencing a relative rise in sea level. No direct measurement of the rate of sea-level rise is currently available for the Guadalupe Delta area, although a relative rise of about 8 mm/yr (0.3 inches/yr) has been estimated in the Colorado River Delta, based on the assumption that recent sedimentation has kept pace with sea-level rise (W.A. White, Bureau of Economic Geology, University of Texas at Austin; pers. comm. 1990). Since the Colorado Delta is the nearest delta site with a recent relative sea-level rise estimate, we assume that the same rate of sea-level rise holds for the lower Guadalupe Delta and surrounding bay areas.

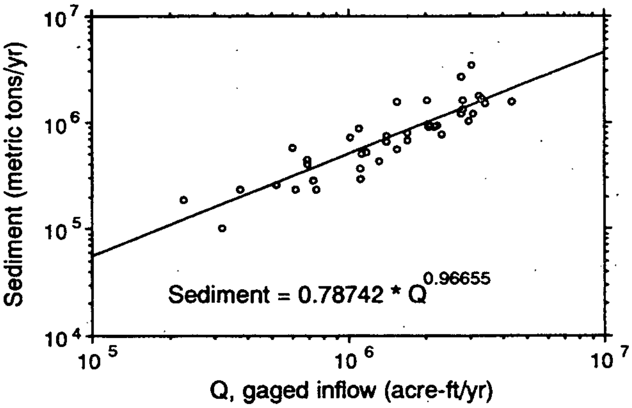


Figure 7.5-2. Relationship between the annual sediment load and the annual gaged river flow to the Guadalupe Estuary. Annual values were summed from daily values at gaging stations at Goliad (San Antonio River) and Victoria (Guadalupe River). The period of record is 1946 through 1985.

Relative sea-level rise consists of two components, a decrease in elevation of the land due to compaction of the sediments and a eustatic sea-level rise independent of land sinking. If the southern portion of the lower delta below the south fork of the river is to remain emergent, material must be deposited on it to offset the relative rise in sea level. On the basis of area alone, the lower delta would require 204,480 m^3 (166 acre-ft/yr) of sediment to offset the 8 mm/yr relative rise in sea level (Table 7.5.1, equation 1). This estimate does not include any organic material that may be produced by delta plants and deposited there.

The TDWR (1980b) determined that the daily peak flow needed to initiate delta inundation was 7934 acre-ft/day. During the period 1941 through 1987, there were 2,154 days with peak gaged flows equal to or greater than 7,934 acre-ft/day. During the 47-year period, the quantity of water flowing to the bay that exceeded 7,934 acre-ft/day and overflowed the river banks was a little less than 32% of the total volume. Therefore, on average, no more than about 32% of the sediment load (261,058 m^3/yr or 212 acre-ft/yr) was available to flow over the subaerial (emergent) portion of the lower Guadalupe Delta and be deposited; the rest was transported through Traylor Cut into Mission Lake or by the river channel into Guadalupe Bay. This estimate assumes that the relationship between gaged flow and sediment transported is linear, true over much of the range of flows. During periods of very high flows, the sediment load per volume of water actually declines, according to TDWR measurements. Therefore, the 32% percent estimate is an upper limit on the amount available for deposition.

During flood conditions on a low tide, the portion of the lower delta that lies between Mission Lake and the south fork of the river can be inundated. During flood conditions on a high tide, the southern portion of the lower delta below the south fork of the river, and the upper delta between Highway 35 and Mission Lake may be inundated (TDWR 1980b). The lower and upper delta areas are approximately the same size, so the amount of sediment that is available for deposition may have to supply an area about twice the size of the lower delta. Even if it were possible to direct the entire flood flow to the lower delta, especially the portion below the south fork of the river, it would have to trap about 78% of the 261,058 m^3/yr (212 acre-ft/yr) of sediment available for deposition to provide the 204,480 m^3/yr (166 acre-ft) needed to offset sea-level rise. This is an unrealistically high rate of retention, especially since the sediment load is dominated by clay-size particles which remain in suspension even with low-flow velocities.

The emergent portion of the lower delta continues to undergo subsidence under present inflow conditions. A series of aerial photographs of the lower delta taken during the period 1939 through 1979 show a gradual increase in the open water area of lakes within the delta (White and Calnan 1990a), a sign of deterioration in the normal growth and decay cycle. Sediment carried by the river under normal flow does not reach portions of the delta below the south fork of the river, and flood flows do not carry enough sediment to offset the relative sea-level rise. One must conclude that the only way the lower portion of the delta will receive the additional sediment needed to offset the relative sea-level rise through natural processes and the present water distribution pattern is by subsiding further and waiting for the deposition currently under way in Mission Lake to return to the lower delta.

Upper portion of the delta. According to modeling studies of the TDWR (1980b), the delta above Mission Lake and below Highway 35 is inundated during high tides with gaged flows greater than 7934 acre-ft/day, or by gaged flows greater than 13,885 acre-ft/day under normal tidal conditions. Using the 13,885 acre-ft/day criterion and gaged flow records for the San Antonio and Guadalupe rivers, flows sufficient to flood the upper portion of the delta occurred an average of five times each year; flows greater than 7,934 acre-ft/day occurred about seven times each year. Consequently, only two of the average seven floods per year were in the range of 7,934 to 13,885 acre-ft/yr; the rest were greater than 13,885 acre-ft/yr. Tide gage data is not available for most of the 47-year period; consequently, it is not possible to determine how many inundations resulted from the combination of high tides and inflows greater than 7,934 acre-ft/day but less than 13,885 acre-ft/day. Therefore, on average, there are five to seven inundations of the upper delta per year.

No direct measurements are available to show whether the upper delta is sinking relative to sea level. This part of the

Guadalupe Delta region is older than the lower delta; it has existed for more than 2,000 years, and the area fronting Mission Lake is one of the older parts of the upper delta (W.A. White, pers. comm.). While the upper delta occasionally receives flood waters from the Guadalupe River, the deeper underlying sediments have had a long time to compact and consolidate; compactional subsidence of these deltaic deposits may be very low. Inspection of maps based upon 1957 aerial photography, and photos made in the 1970's and 1980's does not show the expansion of open water areas on the upper delta that is clearly evident on the lower delta. The size and shape of the open water areas on the upper delta have been very constant over a 30-year period.

State Highway 35 was built across the upper delta and includes a leveed roadway and several bridges spanning the Guadalupe River and smaller bayous. The bridges were built upon pilings that have been driven deep into the deltaic deposits to solid substrate. If compactional subsidence were occurring on the old delta deposits, sinking of the road and levee would be evident compared to the bridges that rest on solid substrate. Highway engineers familiar with Highway 35 in the Guadalupe River floodplain note that differential subsidence between the bridges and the road has not been a problem (Billy D. Parks, Texas Highway Department, Yoakum, TX; pers. comm. 1990). Only a small amount of patching between the highway and the bridges has been required and was entirely attributable to settling within the man-made levee and leaching of sediment from rainfall. Therefore, it is likely that any relative sea-level rise with respect to the upper delta area is the result of eustatic sea-level change only, not sediment compaction.

Flood water reaching the upper delta spills out of the river bank northwest of Highway 35, spreads over the floodplain and Green Lake, passes under several Highway 35 bridges to the upper delta south of the highway, and then spreads out again until it enters Mission Lake. When the flood water first enters the floodplain and Green Lake above Highway 35, the water velocity slows and coarser fractions of the suspended load can be deposited. Consequently, it is unlikely that the suspended load of water that floods the delta between Highway 35 and Mission Lake is as great as the suspended load carried by waters when they leave the river above Highway 35. Since no direct elevation measurements are available from the area, it is not possible to determine whether there is any deposition of sediment from flood waters on the upper delta.

Mission Lake and San Antonio Bay. Since Traylor Cut breached the eastern bank of the river in 1935, 50 to 65% of the normal river flow has gone into Mission Lake. Measurements of the depth of Mission Lake made in 1935 and 1965 and reported by Donaldson et al. (1970) show that the average depth of the lake changed from less than 1.5 m (5 ft) to less than 0.6 m (2 ft) over the 31-year period. For purposes of calculation, we interpret this to mean an average depth change of 0.91 m (3 ft). During the same time span, the relative rise in sea level around the delta was 0.25 m (0.81 ft) (Table 7.5.2, equation 1). Since the depth measurements of Mission Lake were made relative to sea level, the total depth of sediment deposited during the 31-year period must be corrected to include the relative rise in sea level; therefore, the depth of material that was deposited in that time period was 1.16 m (3.81 ft) (Table 7.5.2, equation 2). Using the average surface area of Mission Lake (Diener 1975), the volume change in the lake during 1935 through 1965 period was 9,906,400 m^3 (8,034 acre-ft) (Table 7.5.2, equation 3).

Sediment initially deposited on the bottom of Mission Lake is not consolidated and contains water, some of which is expelled during compaction. Steed (1971) noted that the water content of the upper sediments may be 50%. Shepard and Moore (1960), sampling at several sites in San Antonio Bay, found the water content of the bay sediment to be 50% near the sediment surface, falling to as low as 30% at 18 m (60 ft) below the surface. For this analysis, we assume a 50% water content for the top 0.61 m (2 ft) of sediment, and 40% water content below that depth. Since the sediment in the top 1.16 m of lake bottom contains 40 to 50% water, the actual amount of particulate matter deposited constitutes only slightly more than half of the lake's volume change, calculated above. The volume of the sediment deposited in Mission Lake which reduced the depth to less than 0.6 m (2 ft) was 5,422,900 m^3 (4,398 acre-ft) (Table 7.5.2, equations 4a, 4b, and 4c). Sediment load measurements for both the San Antonio and Guadalupe rivers before 1946 are not available. Therefore, we assume that the average sediment load during the period 1935 through 1945 is the same as the average average load during the 1946 through 1986 period. On this basis, the total sediment load delivered to the estuary from 1935 through 1965 can be estimated and compared to the quantity of sediment actually retained in Mission Lake. Equation 5 in Table 7.5.2 shows that the lake retained 21% of the sediment delivered by the rivers.

The depth of Mission Lake has continued to decrease since 1965. The Traylor Cut subdelta has grown and now includes an emergent portion, decreasing the area of the lake. Steed (1971) reported the depth to range between 0.3 and 1 m (1 to 3.3 ft), and recent field studies by TWDB staff have shown the lake to be unnavigable except by small boats during high tides or high inflows. Calculations were made to extend the estimates of Mission Lake deposition from 1966 through 1986, to test whether the 21% retention rate was reasonable over the additional 21-year period (Table 7.5.3, equations 1 to 6). Equation 6 estimates the average depth of the lake to be 0.1 m (0.3 ft) in 1986.

The lake is not of uniform depth; there are areas deeper and shallower than 0.1 m (0.3 ft). The calculated average depth is a little shallower than seen in recent observations. If emergent sediments in the Traylor Cut Delta behave like Guadalupe Delta sediments, the subaerial portion of the Traylor Cut Delta may have a much lower water content than the submerged sediment (25 to 35% according to Shepard and Moore 1960) and would represent a larger volume of subaqueous sediment if spread through Mission Lake. It is also possible that the retention rate changes with depth; resuspension in Mission Lake may require less wind energy than in deeper waters resulting in a lower retention rate as the water passes through the lake. The retention rate may have changed slightly since the logjams were removed in the 1970's, and the lake area is slightly smaller than Diener's (1975) measurements. All of these uncontrolled factors could result in a slightly deeper average depth than calculated in Table 7.5.3. In the absence of additional information, the 21% retention rate seems reasonable, although it slightly overestimates sediment retention since 1965.

Sediment that is not retained is moved out of Mission Lake through Guadalupe Bay and rejoins the inflow and sediment that is transported through the north and south forks of the river in the lower delta. Some of this sediment moves into San Antonio Bay and the other bay arms of the estuary. Shepard (1953) noted that the greatest shoaling has

Table 7.5.2. Equations for Mission Lake, based on the period 1935 to 1965, showing the relative sea-level rise, depth of deposition, lake volume decrease, quantity of sediment retained, and the sediment load retention rate.

Equation number	Description and equations
1	Relative sea-level rise in Mission Lake, 1935 through 1965. Where: Rate of relative sea-level rise = 8 mm/yr = 0.008 m/yr Time from 1935 through 1965 = 31 yr 0.008 m/yr * 31 yr = 0.25 m
2	Depth of deposition in Mission Lake, 1935 through 1965. Where: Change in Mission Lake depth = 0.91 m Relative sea-level rise = 0.25 m 0.91 m + 0.25 m = 1.16 m
3	Volume decrease in Mission Lake, 1935 through 1965. Where: Depth of deposition from 1935 through 1965 = 1.16 m Mission Lake area = 854 ha 1.16 m * 854 ha * 10,000 m^2/ha = 9,906,400 m^3
4	Quantity of sediment retained in Mission Lake, 1935 through 1965. Where: Top 0.61 m of lake bottom has 50% sediment content Next 0.55 m of lake bottom has 60% sediment content Depth decrease during 31-year period = 1.16 m Volume of unconsolidated deposited material = 9,906,400 m^3 (a) (0.61 m / 1.16 m) * 9,906,400 m^3 * 50% sediment content = 2,604,700 m^3 (b) (0.55 m / 1.16 m) * 9,906,400 m^3 * 60% sediment content = 2,818,200 m^3 (c) 2,604,700 m^3 + 2,818,200 m^3 = 5,422,900 m^3
5	Percent of sediment load retained in Mission Lake, 1935 through 1965. Where: Quantity of sediment retained in Mission Lake = 5,422,900 m^3 Average annual sediment load = 815,805 m^3/yr Time from 1935 through 1965 = 31 yr 5,422,900 m^3 / (31 yr * 815,805 m^3/yr) = 21%

Table 7.5.3. Equations to check the retention rate calculated for 1935 to 1965 sediment loading by extrapolating the measurements to 1986 and estimating Mission Lake depth.

Equation number	Description and equations
1	Quantity of sediment retained in Mission Lake, 1935 through 1986. Where: Average total sediment load = 815,805 m^3/yr Time from 1935 through 1986 = 52 yr Sediment load retention rate = 21% 815,805 m^3/yr * 52 yr * 21% = 8,908,591 m^3
2	Quantity of retained sediment that constitutes the top 0.61 m of the deposits. Where: Vertical depth = 0.61 m Mission Lake area = 854 ha Sediment content of top 0.61 m = 50% 0.61 m * 854 ha * 10,000 m^2/ha * 50% = 2,604,700 m^3
3	Quantity of retained sediment from the river load that constitutes the remaining deposits, from equations 1 and 2 above. 8,908,591 m^3 - 2,604,700 m^3 = 6,303,891 m^3
4	Actual volume of the deposited sediment, from equations 2 and 3 above. Where: Top 0.61 m of lake bottom has 50% sediment content Deeper portion below 0.61 m has 60% sediment content (2,604,700 m^3/50% sediment) + (6,303,891 m^3/60% sediment) = 15,715,885 m^3
5	Volume of Mission Lake in 1986, had there been no sedimentation plus the effect of relative sea-level rise. Where: Average bay depth in 1935 = 1.52 m Time from 1935 through 1986 = 52 yr Rate of relative sea-level rise = 8 mm/yr = 0.008 m/yr Mission Lake area = 854 ha (1.52 m + (52 yr * 0.008 m/yr) * 854 ha * 10,000 m^2/ha = 16,533,440 m^3
6	Calculated average depth of Mission Lake in 1986 based on the 21% retention. Where: Volume of lake not filled in = 16,533,440 m^3 - 15,715,885 m^3 Mission Lake area = 854 ha (16,533,440 m^3 - 15,715,885 m^3)/(854 ha * 10,000 m^2/ha) = 0.1 m

occurred in the upper bay, so much of the deposition in the past century must have occurred in this area. White and Calnan (1990a) pointed out that relative sea-level rise in the lower portion of San Antonio Bay may be about 12 mm/yr (0.47 inch/yr) and could be outpacing deposition. If this is true, the lower bay may be deepening at a rate of 2 to 4 mm/yr (0.08 to 0.16 inch/yr). Unfortunately, there is no information concerning the pattern of movement of sediment in the estuary and deposition of river-borne materials. There are no estimates of how much of the load is deposited in the upper and lower portions of the bay and how much is transported to the Gulf. White and Calnan (1990a) estimated the trapping efficiency of the Guadalupe Estuary at 95%, but this was based on a very indirect capacity-inflow ratio used for reservoirs. It is not known whether the present average sediment loading rate (815,805 m^3/yr or 661 acre-ft/yr) is satisfying the need for sediment anywhere but the Mission Lake area. Without this basic sediment budget information, it is not possible to evaluate loading requirements over the entire estuary, although we can make an estimate based on the needs for maintenance in Mission Lake.

Minimum Sediment Requirements to Maintain Deltaic and Shallow-water Habitats

Justification for sediment input to upper bay areas and deltas. Sediment that is transported into the estuary provides habitat structure such as shallow bay bottoms, marsh sediments, or emergent land. Vegetation including seagrasses, marsh plants, and upland grasses and shrubs (in subaerial deposits) may develop on sediment that is carried into the estuary. The vegetation or the sediment itself provides cover and habitat for estuarine animals, or may contribute organic matter that becomes part of the food web for the animals living in the estuary. Shallow water and submerged vegetated habitats are particularly important in the upper portion of the estuary since mobile larvae and juvenile animals choose to congregate there for part of their life cycles. The supply of larvae and juvenile animals for recruitment into the adult populations in the bays or Gulf depends on the area of habitat available; if the quantity of that habitat is diminished so that fewer larval and juvenile animals can be supported, the populations of adults will eventually decline.

Lower delta sediment input. The lower portion of the Guadalupe Delta is undergoing an inevitable phase of decay and subsidence. Lower delta flooding is more associated with tidal conditions than river flooding (TDWR 1980b) so the delta below the south fork of the Guadalupe River rarely receives sediment from river flow. It is unlikely that sediment can be supplied to this area to offset sea-level rise except through artificial means; consequently, no sediment requirement can be defined for the lower delta.

Upper delta sediment input. The old portion of the Guadalupe Delta between Green Lake and Mission Lake has existed for the past 2000 years. This, plus the lack of expansion of open water areas on the old delta over the last 30 years and the absence of differential subsidence of Highway 35 on the upper delta compared to bridges anchored to solid substrate, indicate that there is little settling of this old delta. There are uncertainties about the amount of suspended load in the flood waters that flow over the old delta so no estimates of sediment transported to it are currently possible. Consequently, there is not sufficient information available to determine whether this area should be periodically flooded for the purpose of transporting sediment to maintain elevation or habitat structure. Therefore, no sediment requirement will be defined for the upper delta.

Quantification of a minimum sediment load for Mission Lake. Mission Lake is subject to two opposing processes with respect to sediment: material continues to be supplied, making the water body shallower, and relative sea-level rise continues to make the water body deeper. There is a balance point for sediment input below which Mission Lake continues to deepen and above which it continues to shoal. If sediment input is less than the balance point, shallow water habitats and the submerged delta front and prodelta will deepen, fringing marshes will become submerged, and subaerial portions of the delta will become subaqueous. If the supply of sediment is equal to this balance point, existing habitat and nursery areas in Mission Lake and around the new subdelta at Traylor Cut will maintain their status quo. If sediment input is greater than the balance point, Mission Lake will continue to fill, the Traylor Cut subdelta will continue to develop, and old habitats will shoal but new habitats will replace them. This balance point of sediment input to Mission Lake and the Traylor Cut subdelta can be determined and an inflow volume calculated that would maintain the sediment input needed to offset the relative rise in sea level.

Delta-building is hard to see from the air. Most of the sediment deposition occurs in the water, and the areal extent of deposition depends on the water depth. Since Mission Lake is a natural hydrologic unit, it is logical to treat it as a whole and not make an arbitrary decision about subdividing regions in it. Using the estimate of bay retention of sediment (21%) from Table 7.5.2, the quantity of sediment required to keep pace with relative sea-level rise is given in Table 7.5.4, equations 1 through 4. The annual sediment load of the river required to offset relative sea-level rise is 162,667 m^3/yr (132 acre-ft/yr). Equation 5 converts this sediment volume into an annual freshwater inflow volume, using the equation shown in Figure 7.5.2.

Proposed constraint. To maintain bathymetry in Mission Lake and the Traylor Cut subdelta, a minimum of 162,667 m^3/yr (132 acre-ft) of sediment must be transported to the estuary by the rivers. This will require an annual gaged inflow of at least 355,235 acre-ft/yr.

7.6 CHARACTERISTICS OF AN ECOLOGICALLY SOUND ENVIRONMENT FOR THE GUADALUPE ESTUARY

Introduction

The purpose of this section is to document the economically and ecologically important plant and animal species using the Guadalupe Estuary. In addition, this section will identify any significant changes that have occurred to plant and animal species due to recent anthropogenic development, provide an ecological characterization of the area, and identify the general salinity conditions that must be

Table 7.5.4. Equations for calculating the sediment load needed to offset the relative rise of sea level in Mission Lake, and the annual inflow volume needed to transport this sediment load.

Equation number	Description and equations
1	Volume increase in Mission Lake due to relative sea-level rise. Where: Rate of relative sea-level rise = 8 mm/yr = 0.008 m/yr Mission Lake area = 854 ha $0.008\ \text{m/yr} * 854\ \text{ha} * 10{,}000\ \text{m}^2/\text{ha} = 68{,}320\ \text{m}^3/\text{yr}$
2	Actual sediment volume needed to offset relative sea-level rise in Mission Lake, taking the water content of newly deposited material into account. Where: Content of newly deposited material is 50% sediment, 50% water Volume of newly deposited material required to offset relative rise in sea level = $68{,}320\ \text{m}^3/\text{yr}$ $50\% * 68{,}320\ \text{m}^3/\text{yr} = 34{,}160\ \text{m}^3/\text{yr}$
3	Annual sediment load required to offset sea-level rise in Mission Lake. Where: Sediment volume to offset relative sea-level rise = $34{,}160\ \text{m}^3/\text{yr}$ Sediment load retention rate for Mission Lake = 21% $(34{,}160\ \text{m}^3/\text{yr}) / 21\% = 162{,}667\ \text{m}^3/\text{yr}$
4	Conversion of sediment volume in m^3 to weight in metric tons. Where: $1\ \text{m}^3$ of sediment = 1121.4 kg Volume of sediment to offset relative sea-level rise = $162{,}667\ \text{m}^3/\text{yr}$ $162{,}667\ \text{m}^3/\text{yr} * 1121.4\ \text{kg/m}^3 * 0.001\ \text{metric tons/kg} = 182{,}415\ \text{metric tons/yr}$
5	Conversion of sediment weight in metric tons/yr needed to offset the relative rise in sea level to gaged inflow in acre-ft/yr. Where: Sediment required = 182,415 metric tons/yr Q is the combined gaged flow of the San Antonio River at Goliad and the Guadalupe River at Victoria, in acre-ft/yr $\text{Sediment} = 0.78742 * Q^{0.96655}$ $Q = 10^{\{(\log_{10}(\text{Sediment}) - \log_{10}(0.78742)) / 0.96655\}}$ $Q = 10^{\{(5.26106 - (-0.10379)) / 0.96655\}} = 10^{5.55052}$ $Q = 355{,}235\ \text{acre-ft/yr}$

maintained to ensure the continuance of the estuary's characteristic biodiversity and productivity.

Economically Important Species

Many estuarine species are exploited by commercial and recreational fishermen in the Guadalupe Estuary. Annual inshore landings from this estuary for commercial fishermen have ranged from 1,486 to 3,545 mt (3,269,742 to 7,799,084 lbs) during the period 1972 through 1989 (Johns 1990); recreational landings have ranged from approximately 30,000 to 350,000 fish during the period 1974 through 1989 (Green et al. 1991). Fesenmaier et al. (1987) recently estimated that the total economic output benefit to the state from the inshore and offshore fishing industry located in this area amounts to $80.3 million and $135.3 million per year for the Guadalupe Estuary region and state.

Commercial fishery harvest. During the period 1972 to 1989, commercial fishermen mainly depended on brown and white shrimp (*Penaeus aztecus* and *P. setiferus*)), blue crabs (*Callinectes sapidus*), eastern oysters (*Crassostrea virginica*), red drum (*Sciaenops ocellata*), spotted seatrout (*Cynoscion nebulosus*), black drum (*Pogonias cromis*), flounder

(*Paralichthys* sp.), sheepshead (*Archosargus probatocephalus*), and mullet (*Mugil* sp.) (Johns 1990). Recent commercial landings of shellfish from the Guadalupe Estuary have consistently been greater than finfish landings, 919 to 3,474 mt (2,022,000 to 7,642,000 lbs) compared with 4.5 to 210.4 mt (10,000 to 463,000 lbs), respectively. Blue crab have accounted for the greatest weight in landings (390.4 to 2,171 mt or 859,000 to 4,776,000 lbs) while white and brown shrimp have had the greatest value, $425,000 to $2,964,000. If Gulf landings were added to total estuarine landings, brown shrimp would account for the most landings and greatest value, and white shrimp would be the second most landed and most valuable species to the commercial industry located in the Guadalupe Estuary. Gulf menhaden (*Brevoortia patronus*) would be a close third. Eastern oysters have been the most variable with respect to annual weight landed, 4 to 880 mt (9,000 to 1,937,000 lbs). It has been illegal to sell red drum and spotted seatrout since September 1981. However, these species are still taken by recreational fishermen for personal pleasure and consumption.

Recreational fishery harvest. The most frequently landed species reported from the Guadalupe Estuary by recreational fishermen have been spotted seatrout, red drum, southern flounder (*P. lethostigma*), black drum, and sand seatrout (*Cynoscion* sp.) (Maddux 1989). Fifty to 60% of all recreational landings from the Guadalupe Estuary were comprised of spotted seatrout with red drum comprising 10 to 25%. Other fish which have accounted for 1 to 5% of the landings included Atlantic croaker (*Micropogonias undulatus*), gafftopsail catfish (*Bagre marinus*), requiem sharks (*Carcharhinidae*), and southern kingfish (*Menticirrhus americanus*).

Recent Changes in Species Composition or Abundance

Collins and Smith (1893) reported commercial landings from Calhoun and Refugio counties (counties most likely to have landings dominated by the Guadalupe Estuary) to be around 273,000 kg (600,000 lb). The eastern oyster was the major species landed in these two counties (136,000 kg or 300,000 lb). Shrimp landings amounted to less than 950 kg (2,000 lb). During 1988, reported commercial landings from the Guadalupe Estuary were 2,409,000 kg (5,300,000 lb; Quast et al. 1989). These landings were dominated by brown shrimp (904,000 kg or 1,989,000 lb), white shrimp (280,000 kg or 616,000 lb), blue crabs (1,636,000 kg or 2,770,000 lb) and oysters (4,040 kg or 8,900 lb). The 1988 landings were not unique; landings from 1977 through 1988 have ranged from a low of 1,640,000 kg (3,600,000 lb) during 1982 to a high of 3,545,000 kg (7,800,000 lb).

Historical accounts of species living in the Guadalupe Estuary were mainly commercial landing reports and brief popular accounts contained in newspapers and magazines. A review of this material leads to the conclusion that today, the species comprising the Guadalupe community does not differ from what historically occurred, with possibly marked decreases in tarpon, snook, striped bass, green sea turtles, diamond back terrapins, whooping cranes, colonial water birds, water fowl, some shorebirds, and very recently neotropical migrant birds (Hall 1984; Matlock and Osburn 1987; Hunt and Slack 1989). During the last 20 years, whooping cranes made significant increases in members (i.e., total number for 1989 and 1990 were 146 and 143, respectively).

Ecologically Important Species

To provide the productive fisheries found in this estuary, the habitat and the plants and animals on which the fishery species feed, or on which their prey feed, must be maintained. Spotted seatrout, southern flounder, and red drum depend on shrimp, pinfish (*Lagodon rhomboides*), menhaden, anchovy (*Anchoa* sp.), and mullet for food (Matlock and Garcia 1983). Many larval fish depend on plankton, polychaetes, and crustaceans (Matlock and Garcia 1983; Durako et al. 1988). Shrimp depend on detritus, polychaetes, epiphytes, and plankton (Zein-Eldin and Renaud 1986; McTigue and Zimmerman 1991). Eastern oysters, other bivalves, and polychaetes depend on planktonic food being in the right place and at the right time because they have limited or no mobility (Quast et al. 1988). Vascular plants provide habitat and food for estuarine organisms (Minello and Zimmerman 1985; Zimmerman et al. 1990a). Oyster reefs provide habitat for a different set of species than found in vegetated or unvegetated sand or mud habitats (Zimmerman et al. 1989). To remain productive over time, all of these species depend on their environment receiving at least a minimum nutrient and sediment load, and enough freshwater flow to keep the estuarine water within particular upper and lower salinity bounds. The Venice System for classification of marine waters according to salinity (Symposium on the Classification of Brackish Waters 1958) is used to describe the salinity characteristics of the Guadalupe Estuary. The salinity ranges defined in the Venice System are: fresh (0 to 0.5‰); oligohaline (0.5 to 5‰); mesohaline (5 to 18‰); polyhaline (18 to 30‰); and euhaline (30 to 40‰).

Primary producers. Phytoplankton, emergent marsh plants, submerged vascular vegetation, algae, floating plants (e.g., water hyacinth [*Eichhornia crassipes*]) and duckweed (*Lemna* spp.), high marsh plants, deltaic floodplain swamps, and detritus and nutrients from inland areas are the major sources of energy and matter used to drive the Guadalupe estuarine ecosystem. In addition to providing energy and matter, all these plants provide habitat and offer protection from disease and predation for many coastal and estuarine animals (Minello and Zimmerman 1985; Quast et al. 1988).

Most plant material from coastal wetlands enters the estuarine food web through a detritus-microbial, algae-detritus-zooplankton-benthic, or algae-detritus-zooplankton-larval fish pathway. The phytoplankton community in the Guadalupe Estuary is varied, having diatoms, blue-green algae, flagellates, and green algae. It is most important to ensure that a variety of phytoplankton species are maintained in the Guadalupe Estuary so that no single species becomes dominant and noxious (Section 5.1), and to provide a variety of forage. This also guarantees that several phytoplankton species with varying physiological requirements are present at any time to efficiently take advantage of the different environmental conditions that frequently occur in the estuary. Phytoplankton can take up nutrients rapidly and quickly respond to changes in the environment. For this reason, they are an indication of current conditions in the estuary. Conversely, major vascular plant species within the Guadalupe Estuary are fewer in number and their physical and biological ability to live in a specific location is governed more by the general environmental conditions that persist at a site through time. Therefore, their presence is an indication of the long-term average conditions of an area within the estuary.

Zooplankton. One of the most important animals in this estuary is *Acartia tonsa*. This copepod generally constitutes more than half of the number and biomass of the estuarine zooplankton community (Lee et al. 1987). It plays a major role in transferring nutrients from phytoplankton to larval fish, polychaetes, and bivalves. It is a major grazer on phytoplankton and is used extensively by larval fish and shellfish as food.

Filter and detritus feeders. Chironomid insects, *Rangia cuneata*, *Hobsonia florida*, *Texadina sphinctostoma*, and blue crabs are species that have been identified to trap detritus and phytoplankton in oligohaline areas of the Guadalupe Estuary. Several species of phytoplankton rapidly uptake dissolved nutrients from freshwater inflows, recycle nutrients from metabolic waste, and become "pasture" for the zooplankton. Filter feeders, such as the clam (*R. cuneata)* and eastern oysters, rapidly incorporate the carbohydrates and other organic molecules created by phytoplankton and zooplankton, and convert them to animal tissue (Soniat et al. 1984; Soniat and Ray 1985).

Capturing the material coming in with freshwater inflows as quickly as possible guarantees the maximum use and cycling of the nutrients within the estuary. If nutrients and detritus are not taken up quickly, they have a greater chance of being buried or washed out to sea where they are more easily lost to living organisms.

Gizzard shad (*Dorosoma cepedianum*), striped and white mullet, Gulf menhaden, bay anchovy, clams (*R. cuneata* and *R. flexuosa*), and eastern oyster represent an ecologically important group that feeds directly on detritus and plankton. This group serves to maintain plankton and detritus at acceptable levels, and may represent a shortened food chain to higher level predators such as spotted seatrout, red drum, southern flounder, and black drum.

Other consumers. Pinfish, Gulf and longnose killifish (*Fundulus* spp.), sheepshead minnows (*Cyprinodon variegatus*), silversides (*Menidia* spp.), grass shrimp (*Palaemonetes* spp.), shrimp (brown and white), silver perch, and many juvenile fishes of larger predators represent a group of secondary consumers that are a major food source for higher level consumers such as red drum, herons, egrets, porpoises, and spotted seatrout. This latter group of tertiary consumers are for the most part carnivores eating most anything but mainly crustaceans, insects, mollusks, small fish, and crabs (McTigue and Zimmerman 1991).

Communities in the Guadalupe Estuary

The Guadalupe Estuary has a definite salinity gradient with relatively large areas having different salinities at intermediate inflow volumes. It has fresher areas near the Guadalupe River mouth (i.e., Mission Lake, Guadalupe Bay, and Hynes Bay), and high salinity areas in Espiritu Santo Bay near Pass Cavallo, one of the major bay-Gulf of Mexico passes. During times of low freshwater inflows, a large part of the estuary becomes saline and the gradient is compressed with higher salinities (greater than 15‰) approaching the lower part of the river delta. During floods, a large part of the estuary becomes brackish with lower salinities occurring in the southern part of San Antonio Bay (less than 10‰) and in Espiritu Santo Bay (less than 25‰). During a recent major flood (1987), this estuary has had salinities of less than 10‰ near the Gulf passes (TPWD Coastal Fisheries Resource Monitoring Data).

Four distinct areas of the Guadalupe Estuary. Four areas were used for the ecological characterization of the Guadalupe Estuary (Figure 7.6.1). A visual inspection of salinity and nutrient isopleths during intermediate inflow conditions showed that each of the four areas had relatively homogeneous salinity and nutrient concentrations that were distinct from one area to the next. The upper portion of the estuary near the Guadalupe River mouth (Figure 7.6.1, Area I, including Mission Lake, Guadalupe Bay, Hynes Bay, and a portion of San Antonio Bay immediately adjacent to these

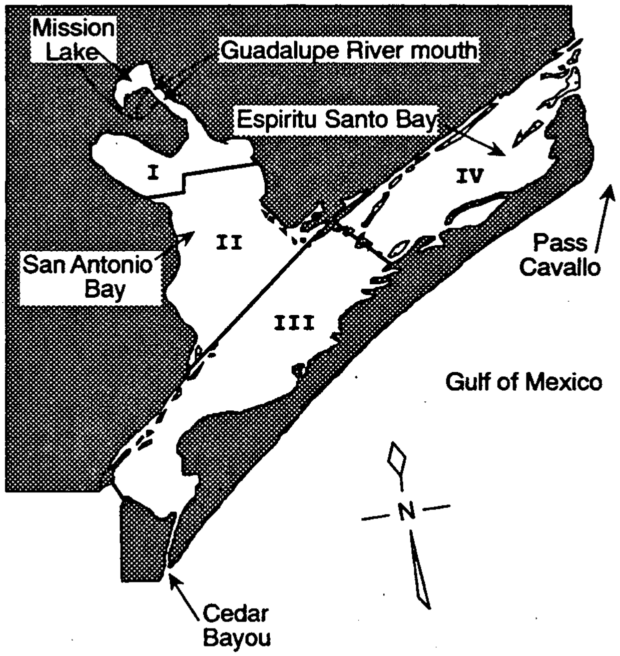


Figure 7.6.1. Map of the four distinct areas of the Guadalupe Estuary.

bays) is highly variable with low salinities (0 to 10‰, figures 4.1.6, 4.1.7, and 7.1.7) and high concentrations of dissolved nutrients (0.2 to 1.0 mg/l dissolved inorganic nitrogen [DIN] and 0.2 to 0.5 mg/l total phosphorus [TP], figures 4.2.1, 4.2.3, and Table 7.2.1). The salinity data from the TPWD monitoring program used in the shrimp analyses in Section 6.2 showed that Mission Lake and the Guadalupe Bay area actually had salinities of less than 5‰ most of the time.

An upper area of San Antonio Bay (Figure 7.6.1, Area II), starting just south of Area I and extending to a line running parallel to the Intracoastal Waterway, is less variable. Salinities are higher and generally range from 10 to 20‰; nutrient concentrations are lower and range from 0.1 to 0.5 mg/l for DIN, and 0.1 to 0.5 mg/l for TP.

Area III (Figure 7.6.1) includes the lower third of San Antonio and Ayres bays and is less variable than areas I and II. Salinities are generally between 15 and 25‰, DIN ranges from 0.05 to 0.2 mg/l, and TP ranges from 0.1 to 0.2 mg/l.

Espiritu Santo Bay is the most stable area (Figure 7.6.1, Area IV) within the Guadalupe Estuary with respect to salinity and nutrients. Salinities generally range from 25 to 35‰, and nutrient concentrations range from 0.05 to 0.2 mg/l for DIN and 0.05 to 0.1 mg/l for TP.

Characterization of Mesquite Bay is more difficult because it has salinities and total phosphorus concentrations which are similar to the lower part of San Antonio Bay, and nitrogen concentrations similar to those found in Espiritu Santo Bay. Mesquite Bay salinities, and possibly nutrient concentrations, are highly influenced by whether Cedar Bayou is open to the Gulf of Mexico. For this discussion, Mesquite Bay was judged to be more like the lower part of San Antonio Bay and was made a part of Area III.

Characterization of biotic communities. For each of the four areas, the biotic community of the Guadalupe Estuary was divided into three biological groups: vegetation, benthos, and finfish-shellfish. Characteristic biotic assemblages for each area of the estuary were identified from published surveys and data from the TPWD Coastal Fisheries Resource Monitoring Program. A set of species or higher taxa was identified for each area, which quantitatively and ecologically contributes significantly to the area's biotic identity. Often, these species are the dominant members of their communities, so any changes in their populations would constitute a major change in the ecosystem. The characteristic assemblages identified for each of the four areas ranged from 50 to 76 species of plants, benthos, and finfish-shellfish. These represent only a fraction of the total biotic community. The remainder of the species which can be found in each area are minor components, either temporary residents or organisms which occurred or were caught in low numbers or frequency.

Vegetative communities. Vegetation types included submerged, emergent aquatic, and upland plants. The latter are found in the transition from high marsh to upland, and at higher elevations in the delta (Area I).

The major wetland habitat types were identified from available surveys of the estuary (Childress, 1975; White et al. 1989; Pulich 1990; TPWD 1990b). These major habitat types are freshwater marshes, brackish-water marshes, and saltwater marshes. The saltwater and brackish-water marshes were subdivided into low and high zones. Pulich (1990) further subdivided the high marsh zone into low and high subzones.

Freshwater, brackish-water, and saltwater marshes are distinguished by characteristic salinity regimes. Low and high marsh zones are distinguished by inundation frequency. Low zones are inundated more often (and generally for longer periods) than high zones. The vegetation found in each marsh type are those most suited to the salinity and inundation regimes of that habitat. Because most wetland vegetation can exist over a wide range of salinities and inundation regimes, many species are not limited to a single marsh type. Heterogeneous vegetation communities are characteristic especially of transitional areas going from saltwater to brackish-water or brackish-water to freshwater marshes.

For a species to be included in the characteristic species assemblage of the vegetation community, it had to be reported in a published survey of the Guadalupe Estuary (Table 7.6.1). Some members of this list were dominant in one survey and uncommon in another. Some minor species occurring in the estuary were not listed. As a result, the vegetation list includes species which were dominant at sites within the area. Obviously, this is not a complete list of all plants occurring in each area of the estuary.

Benthic communities. Benthos are bottom-dwelling macroinvertebrates which include mollusks, polychaetes, crustaceans, and other similar organisms that are retained by a 0.5 mm mesh screen. Benthic invertebrates were captured with a bottom sediment sampler such as an Ekman Dredge, Ponar sampler, or a core sampler.

Table 7.6.1. Occurrence of dominant (X) vegetation in each area within the Guadalupe Estuary based on reviews of vegetation surveys (Childress 1975; White et al. 1989; Pulich 1990; TPWD 1990) and probable salinity preference.

Common name and species	Area				Probable salinity preference (ppt)[a]
	I	II	III	IV	
Arrowhead (*Sagittaria* sp.)	X				0 to 5^{2}
Annual glasswort (*Salicornia bigelovii*)		X	X	X	10 to 35^{1}
Bald cypress (*Taxodium distichum*)	X				0 to 10^{1}
Bermuda grass (*Cynodon dactylon*)	X	X			0 to 20^{1}
Black mangrove (*Avicennia germinans*)		X		X	10 to 37^{2}
Black willow (*Salix nigra*)	X				0 to 10^{1}
Bulrush (*Scirpus maritima*)	X				0 to 15^{2}
Carolina wolfberry (*Lycium carolinianum*)		X		X	10 to 35^{1}
Cattail (*Typha* sp.)	X				0 to 10^{2}
Clover grass (*Halophila engelmanni*)				X	23 to 37^{2}
Common reed (*Phragmites australis*)	X	X			0 to 15^{2}
Elephant ear (*Colocasia antiquorum*)	X				0 to 10^{1}
Gulf cordgrass (*Spartina spartinae*)	X	X		X	0 to 35^{1}
Marsh aster (*Aster tenuifolius*)		X	X	X	10 to 35^{1}
Marsh elder (*Iva frutescens*)	X			X	10 to 35^{1}
Palmetto (*Sabal minor*)	X				0 to 10^{1}
Perennial glasswort (*Salicornia virginica*)		X	X	X	10 to 37^{2}
Purslane (*Sesuvium portulacastrum*)		X	X	X	10 to 37^{1}
Salt-flat grass (*Monanthochloe littoralis*)		X		X	10 to 37^{1}
Saltgrass (*Distichlis spicata*)	X	X	X	X	5 to 37^{2}
Saltmeadow cordgrass (*Spartina patens*)	X		X	X	0 to 37^{1}
Saltwort (*Batis maritima*)		X	X	X	5 to 40^{2}
Sea blite (Suaeda *maritima*)		X	X	X	10 to 37^{1}
Sea oxeye daisy (*Borrichia frutescens*)	X	X	X		0 to 25^{1}
Shoalgrass (*Halodule wrightii*)	X	X	X	X	23 to 37^{2}
Smooth cordgrass (*Spartina alterniflora*)	X	X	X	X	5 to 20^{2}
Water nymph (*Najas guadalupensis*)	X				0 to 9^{2}
Widgeongrass (*Ruppia maritima*)[b]	X	X	X	X	22 to 32^{2}
Water hyacinth (*Eichhornia crassipes*)	X				0 to 1^{2}

[a] The superscript indicates source of information; 1 - indicates range from occurrence as shown within table (X), and 2 - indicates information is from Table 5.2.2.

[b] Shoalgrass out-competes widgeongrass when salinities are above 20‰.

Several surveys were used to compile data for the benthic communities (Harper 1973; Matthews et al. 1974; Academy of Natural Sciences of Philadelphia 1977, 1979; White et al. 1989; Montagna and Kalke 1990). Only Harper, Matthews et al., and Montagna and Kalke collected samples more than once at each location; the organisms found to be common in their investigations are summarized in tables 7.6.2, 7.6.3, and 7.6.4. During these three surveys, high inflow conditions prevailed. The Montagna and Kalke survey occurred during the record inflow year 1987. For this reason, the characteristic benthic assemblages identified from these studies are biased toward low salinity conditions. Membership in the benthic community was generally limited to those species which occurred in samples at least 25% of the time, or in 50% or more of the sample sites within an area, during any survey.

Finfish-shellfish communities. Finfish-shellfish communities included all fish, shrimp, crabs, and oysters captured by gear used by the TPWD Coastal Fisheries Resource Monitoring Program. It was possible to divide the data for many of the finfish-shellfish species into life stages (adult, sub-adult, and juvenile) based on the size selectivity of three of the gears used (Table 7.6.5). The gear types (and target organisms) were gill nets (adult and subadult fish and adult crabs), shrimp trawls (subadult shrimp and crabs, and small adult and sub adult fish and crabs), bag seines (juveniles of all types and small adults like silversides and killifish), and oyster dredges (adult oysters). The relative density of each life stage for each species within each area was then evaluated (Table 7.6.6).

A total of 215 finfish-shellfish species were captured and identified by the TPWD Coastal Fisheries Resource Monitoring Program from 1975 to 1987. These included 149 teleosts (bony fish), 49 invertebrates, 13 chondrichthys (cartilaginous fish), and 3 reptiles. Data discussed in this section were collected between November 1975 and December 1987 and were used to identify species characteristic of each of the previously defined estuarine areas (Table 7.6.7). The finfish-shellfish species included in the

Table 7.6.2. Summary of macroinvertebrate data by area in the Guadalupe Estuary from Harper (1973) for those organisms occurring in at least 25% of the samples taken. Symbols are as follows: o = occurred in 25-49% of samples taken; + = occurred in 50-74% of the samples taken; * = occurred in more than 74% of the samples taken.

Taxa	Area and site																			
	I							II					III							
	1	2	3	4	5	6	7	1	2	3	4	5	1	2	3	4	5	6	7	8
Rangia cuneata	o																			
Ostracoda A	o																			
Hobsonia florida	o	+	+																	
Chironomidae	o	*	+	o																
Capitella capitata				o	o		o													
Polydora ligni	o				o			o												
Nematoda				o	o	o						o								
Texadina sphinctostoma	*	*	*	*	*	o	o	o												
Rangia flexuosa	o	+	+	o	*		o	o												
Peloscolex gabriella	+		o	o			*	+	o											
Texadina A							o		o											
Harpacticoida A	o							o	+	o										
Macoma mitchelli					o		o	o				o	o							
Nereis succinea					o														o	
Monoculoides sp.					o	o	o	+	o										o	
Parandalia fauvelli					o		+	o	*	+	+		o	o	o		o		o	
Tubulanus pellucidus	o			o	o	o	+	+	+	+		+	+	+	o		+	+	+	o
Streblospio benedicti	*	*	+	*	*	*	*	+	*	*	o	+	+	o		o	o		o	o
Mediomastus californiensis	+	*	*	*	*	*	*	*	*	*	*	*	*	*	*	*	*	*	*	*
Mysidopsis bahia								o												
Mysidopsis almyra									o											
Diopatra cuprea										o										
Xenanthura brevitelson											o									
Glycinde solitaria												o	o	+	+	+	o	o	+	o
Cumacea A															o		o		o	
Oxyurostylis salinoi															o			o		o
Scoloplos foliosus																	o			
Scoloplos fragilis																	o			
Spiochaetopterus oculatus																		o		
Prionospio pinnata																			o	o
Spio setosa																				o

characteristic species assemblage list were restricted to those that occurred in at least 20% of the samples within at least one gear type in an area. Preferred areas (areas having a greater abundance) for a given species and life stage (adult, subadult, or juvenile) within the Guadalupe Estuary were identified by comparing the difference between the highest mean catch (X_j) from an area and the mean catch of any other area(X_i), with the sum of their standard errors ($SE_j + SE_i$). If the difference between the mean catches exceeded the sum of the standard errors (i.e., $X_j - X_i > SE_j + SE_i$), the area with the higher mean was considered to be the preferred area.

Ecological Characterization of the Guadalupe Estuary

Vegetation in fresh-to-oligohaline Area I. The vegetation of the upper Guadalupe Estuary area has been characterized by several surveys (Childress et al. 1975; Benton et al. 1977; White et al. 1989; Pulich 1990; TPWD 1990). Using these surveys, a characteristic species assemblage for this area was identified which contains 19 species (Table 7.6.1).

The majority of the vegetation in this region that was discussed by the surveys is on and around the Guadalupe River delta, but it also occurs in Hynes Bay and Mission Lake. Dominant vegetation types in Benton's 1976 survey were: palmetto (*Sabal minor*), arrowhead (*Sagittaria* sp.), and water hyacinth (*Eichhornia crassipes*) along both forks of the Guadalupe River; and saltgrass (*Distichlis spicata*), sea oxeye daisy (*Borrichia frutescens*), saltmeadow cordgrass (*Spartina patens*), Gulf cordgrass (*Spartina spartinae*), and Bermuda grass (*Cynodon dactylon*) in the brackish marshes. Along relict tributaries in the lower delta, bulrush (*Scirpus* sp.) was a major component; common reed (*Phragmites australis*) and cattail (*Typha*) were also common, while smooth cordgrass (*Spartina alterniflora*), was not mentioned.

Ten years later, Pulich (1990) reported vegetation types from the same general area as the Benton surveys. Changes in the vegetation assemblages since the Benton survey indicated a more brackish-water community, including density decreases in some low-saline, freshwater-tolerant species (common reed, water hyacinth, and cattail) and density increases in brackish-water-tolerant species (bulrush). In contrast with the Benton survey, Pulich reported the low brackish and saltwater marsh plant, smooth cordgrass was also present.

According to Pulich (1990), the primary difference between average and wet years for the delta appears to be the duration and not the frequency of flood events. Decreased duration coupled with increasing salinity of the delta lakes may lead to an increase in brackish species in the delta vegetative community.

The submerged species, widgeongrass (*Ruppia maritima*) and water nymph (*Najas guadalupensis*), have both been reported from the delta area (Pulich 1990). Submerged species have optimum salinity bounds, but tolerate a range of salinities. The effect of salinity on their distribution is

Table 7.6.3. Summary of benthic data, by area in the Guadalupe Estuary from Matthews et al. (1974). Symbols are as follows: + = occurred in 50-74% of the samples taken; * = occurred in more than 74% of the samples taken.

Taxa	Area and site																				
	I							II								III					
	1	2	3	4	5	6	7	1	2	3	4	5	6	7	8	1	2	3	4	5	6
Hobsonia florida	+	+	*																		
Chironomidae	*	+	+	+	+																
Rangia cuneata		+	*	*	*	*	+		*												
Texadina sphinctostoma			*	*	*	*	*	+	+												
Texadina B							*														
Macoma mitchelli				+	+			+										+			
Mediomastus californiensis							+	*	*	*	+		*	+		+	*	+	+	*	+
Parandalia fauvelli					+		*		*	*			*	+		*	*	*	+	*	+
Nemertea							+		+	+			+				+	+			
Streblospio benedicti							+	+	+	+			+								
Mulinia lateralis											+				+			+	+	+	
Neanthes succinea													+							+	

Table 7.6.4. Summary of benthic data by area in the Guadalupe Estuary from Montagna and Kalke (1990). Symbols in the table are as follows: o = occurred in 25-49% of samples taken; + = occurred in 50-74% of the samples taken; and * = occurred in more than 74% of the samples taken.

Taxa	Area and site			
	I	II		III
	1	1	2	1
Hobsonia florida	+			
Chironomidae	o			
Parandalia ocularis	+			o
Capitella capitata	+	+		o
Oligochaeta	*	+		
Texadina sphinctostoma	*	*	o	o
Rhynchocoela	+	+	+	+
Mediomastus californiensis	*	*	*	*
Streblospio benedicti	*	*	*	*
Mulinia lateralis	*	*	*	*
Macoma mitchelli	o	*	+	*
Monoculoides sp.	+	o	+	o
Cyclaspis varians		o	o	o
Glycinde solitaria		o	+	+
Oxyurostylis smithi		o		o
Haploscoloplus foliosus			o	+
Haraprionospio pinnata			o	+
Cossura delta			o	
Turbellaria			o	
Diopatra cuprea			o	
Polydora caulleryi				o
Pyramidella sp.				o
Polydora websterii				o
Acetocina canicula				o

largely determined by the average salinity regime rather than short-term fluctuations. In fresh-to-oligohaline (0 to 5.0‰) areas, naiad production will be favored, while in mesohaline (greater than 5‰) areas, widgeongrass will be the prevalent submerged species.

Emergent vegetation throughout the estuary was also identified and categorized into habitat types and dominant species (Pulich 1990). The low saltmarsh species, smooth cordgrass, was well established at all sites surveyed throughout the estuary. High marsh zone community species composition varied with salinity (brackish or salt) and elevation. The low, more frequently inundated subzone consisted primarily of glasswort (*Salicornia* sp.) and saltwort (*Batis maritima*), while the high subzone saltgrass consisted of marsh aster (*Aster tenuifolius*), salt-flat grass (*Monanthochloe littoralis*), Carolina wolfberry (*Lycium carolinianum*, a forage for whooping cranes according to Hunt and Slack 1989), and sea oxeye daisy. The high-marsh subzone was found to grade into an upland community in all areas of the estuary.

Benthos in oligohaline Area I. A characteristic benthic assemblage was identified from the available studies which included 18 taxa (tables 7.6.2, 7.6.3, and 7.6.4). The most common members of this assemblage were the polychaetes *Mediomastus californiensis*, *Streblospio benedicti*, and *Hobsonia florida*; the oligochaete *Peloscolex gabriella*; the gastropod *Texadina sphinctostoma*; the pelecypod *Rangia cuneata*; and larvae of the insect family chironomidae (midges). Of these species, *M. californiensis* and *S. benedicti* were reported as most dominant and frequently encountered. *Hobsonia florida*, *R. flexuosa*, and chironomid larvae were reported as dominant only in this area of the estuary.

Finfish-shellfish in oligohaline Area I. There have not been many thorough investigations of the fish, shrimp, and crab communities of this estuary. The species composition of each area was best shown with the TPWD Coastal Fisheries Resource Monitoring data. The finfish-shellfish community of the upper bay area had a mixture of fresh and saltwater organisms. Twenty-five species were identified as characteristic of the fresh-to-oligohaline upper-bay area (Table 7.6.7).

The more mobile finfish-shellfish community is naturally a more dynamic assemblage than vegetation or benthic animals, and can quickly react to salinity changes. Mobility allows these animals to migrate throughout the estuary in response to environmental stimuli. Consequently, some species are not uniformly distributed over the estuary during portions of their life (see adult, subadult, and juveniles in Table 7.6.2).

Two species in Area I exhibited their highest catch rates (for at least one life stage) in the estuary, adult blue catfish (*Ictalurus furcatus*) and longnose gar (*Lepisosteus osseus*, Table 7.6.7). Several other species, which did not have 20% frequency in catch, also had the highest catch rates from Area I. These included juvenile black drum (trawls), juvenile gizzard shad (bag seines), juvenile striped mullet (trawls), adult blue crab (gill nets), subadult red drum, and juvenile Gulf menhaden (Table 7.6.6). Several species had high catch rates in more than one area including Area I (Table 7.6.7). These were subadult red drum, subadult and juvenile Atlantic croaker, subadult brown shrimp, and adult and subadult alligator gar (*Lepisosteus spatula*). Data presented in Section 6.2 also showed high catch rates for subadult blue crab and white shrimp concentrated in the two upper areas of the estuary (Figures 6.2.1 and 6.2.4), and each species showed a negative relationship with salinity (Figures 6.2.2 and 6.2.5).

Table 7.6.5. Mean sizes (total length or width) for various species caught with different gear in the TPWD Coastal Monitoring Program illustrating selection for different life stages: adults (A) denotes reproducing groups, subadults (S) designates a group approaching age of first reproduction, and juveniles (J) are younger individuals—mainly young of year.

Common name	Gear		
Gillnet	Trawl	Bag seine	
Atlantic croaker	A (7 to 12) [a]	S(3 to 4)	J (2 to 4)
Black drum	A (10 to 18)		J (4 to 6)
Blue crab	A (5 to 6)	S (3 to 4)	J (1 to 2)
Brown shrimp	S (3 to 4)	J (2 to 3)	
Gafftopsail catfish	A (19 to 22)	S (4 to 5)	
Gulf menhaden	A (5 to 11)	S (2 to 4)	J (1 to 2)
Pinfish	A (7 to 10)	S (3 to 4)	J (1 to 4)
Red drum	S (15 to 20)		J (2 to 5)
Sea catfish	A (12 to 14)	S (6 to 8)	J (4 to 6)
Sheepshead	A (12 to 18)	S (3 to 5)	J (1 to 4)
Southern flounder	A (8 to 15)	S (5 to 6)	J (2 to 3)
Spot	A (9 to 11)	S (3 to 4)	J (2 to 3)
Spotted seatrout	A (15 to 19)		J (2 to 3)
Striped mullet	A (12 to 14)		J (2 to 5)
White shrimp		S (3 to 4)	J (2 to 3)
All other finfishes	A (16 to 32)	S (2 to 4)	J (2 to 3)

[a] Sizes are means reported in Mambretti et al. (1990) for the Guadalupe Estuary: gillnet values are the range of means in inches from spring or fall seasons from 1975 to 1988; trawl and bag seine values are annual means (in inches) from 1982 to 1988 and 1978 to 1988, respectively.

A few species were captured primarily by bag seines throughout their lives, so it was not possible to determine an area of preference for any particular life stage based on gear selectivity. One species, sheepshead minnow, had its highest catch rates in Area I.

It is clear from Table 7.6.7 that these few species are only a small part of this community. Other commonly captured species (though not in large numbers or high frequency) include adult and subadult sea catfish, and adult bay anchovy, and sheepshead, eastern oyster, spotted seatrout, gizzard shad, and black drum.

Vegetation in oligohaline-to-mesohaline Area II. Vegetation surveys of the mid-bay area, which concentrated on the Welder Flats Coastal Preserve and Aransas National Wildlife Refuge sites, have been less detailed than those of the upper-bay. Childress et al. (1975) sampled vegetation at four sites between Dagger Point and Live Oak Point. TPWD (1990) surveyed the vegetation in the Welder Flats Coastal Preserve and the Aransas National Wildlife Refuge. White et al. (1989) used 1979 NASA photographs to map the vegetation communities for the estuary while Pulich (1990) used 1987 NASA-AMES aerial photographs to map the vegetation. A characteristic vegetation assemblage for this area was identified and included 19 species (Table 7.6.1).

Childress et al. (1975) reported the dominant submerged vegetation on the southeast side of the mid-bay area from 1971 to 1974 was shoalgrass, widgeongrass, and the algae *Polysiphonia gorgoniae*. The dominant emergent vegetation types were common reed, smooth cordgrass, and cattail. The submerged species widgeongrass and shoalgrass were also reported from this region by Pulich (1990). The Seadrift area often exhibited the highest turbidities and predictably the lowest submerged vegetation densities (Pulich 1990). Farther south, along the Aransas National Wildlife Refuge shoreline, Pulich found smooth cordgrass occurred in the low saltwater marsh areas, and uplands typically contained saltmeadow cordgrass, Gulf cordgrass, and seacoast bluestem.

When TPWD personnel surveyed the Welder Flats area in 1990, the upland habit was dominated by a Gulf cordgrass-halophyte community. It also had saltmarsh, sandflats, and submerged seagrass. The plant species occurring in the marsh habitat were those commonly associated with saltwater and brackish-water marshes and transitional wetlands, with some occurrence of species usually associated with oligohaline areas (common reed and water nymph).

Benthos in oligohaline-to-mesohaline Area II. The benthic animals found in this area were the subject of the same surveys described for the oligohaline area (Area I). Characteristic benthos of Area II contained 18 species including: the dominant polychaetes *Mediomastus californiensis*, *Streblospio benedicti*, and *Parandalia fauvelli*; the pelecypod *Rangia cuneata*; and the gastropods *Hydrobia* sp., *Texadina sphinctostoma*, and *Texadina* B (unidentified) (tables 7.6.2, 7.6.3, and 7.6.4). *Hydrobia* was reported only by the Academy of Natural Sciences of Philadelphia (1978; 1981) and may have been confused with *Texadina* sp.

Finfish-shellfish in oligohaline-to-mesohaline Area II. The characteristic finfish-shellfish community of this mid-bay area was comprised of 25 species of both freshwater and marine species (Table 7.6.7). Of fish inhabiting Area II, only adult gizzard shad from gill nets were caught more often than in any other area.

Several species exhibited high catch rates in Area II as well as in one or more other areas. These were adult and subadult spotted gar, subadult Gulf menhaden, all life stages of Gulf killifish, subadult and juvenile Atlantic croaker, adult and subadult brown shrimp, adult and subadult blue crab, and subadult white shrimp.

Eastern oysters were captured most frequently in this region and the adjacent lower San Antonio Bay area (Area III). The majority of oyster reefs in the estuary are located in these two areas (Quast et al. 1988). There are several reefs in Area IV, but they are not as extensive.

Vegetation in mesohaline-to-polyhaline Area III. A characteristic vegetation assemblage containing five species was developed from the submerged vegetation surveys (Table 7.6.1). Childress et al. (1975) surveyed two sites near Clear Lake on the North shore of Matagorda Island and listed shoalgrass and the algae *Spyrida filamentosa* and *Polysiphonia gorgoniae* as the dominant submerged vegetation, and smooth cordgrass and sea blite as the dominant emergent vegetation. Pulich reported large expanses of widgeongrass and/or shoalgrass could be found near the Matagorda Island shoreline during the summer months.

Like the mid-San Antonio Bay area, the submerged vegetation of Area III was dominated by widgeongrass and shoalgrass. The dominant species was related to the average salinity regime, with widgeongrass favored during periods of lower salinities (less than 20‰), and shoalgrass favored during higher salinities (more than 20‰).

The effects of salinity fluctuations around Cedar Bayou serve to demonstrate how salinity can affect submerged vegetation. Cedar Bayou, located in Mesquite Bay, has a history of periodically opening and closing. When closed, the salinity regime in Mesquite Bay is largely determined by freshwater inflow from the Guadalupe River and evaporation. When the pass is open, water exchange with the Gulf of Mexico buffers the salinity toward sea water (35‰). All of these factors can influence the salinity in Mesquite Bay and can change the dominant species of the local seagrass bed (Pulich 1990). When the drought of the early 1950's ended with the flood of May 1957, salinities went from between 36 and 50‰ in May to 1.7‰ in June 1957, and remained below an average of 19‰ for at least one year. Concomitant with this change in the salinity regime was a change in dominant seagrass flora from shoalgrass to widgeongrass (Hoese 1960).

A similar but opposite occurrence was noted in October 1987 when Cedar Bayou was opened. The pre-opening brackish salinities of Mesquite Bay were raised to seawater salinity by seawater exchange with the Gulf of Mexico and caused the dominant submerged species to shift from widgeongrass to shoalgrass.

Benthos in mesohaline-to-polyhaline Area III. Four studies were used to characterize the benthos of this area (Harper 1973; Matthews et al. 1974; White et al. 1989; Montagna and Kalke 1992). The characteristic benthic assemblage contained 19 species (tables 7.6.2, 7.6.3, and 7.6.4) dominated by the polychaetes *M. californiensis* and *Parandalia fauvelli*. Although there are fewer dominant species in Area III compared to the previous two areas, there were several very common species including the polychaetes *S. benedicti* and *Glycinde solitaria*, the pelecypod *Mulinia lateralis*, and the rhynchocoel *Tubulanus pellucidus*. *Tubulanus pellucidus* and *G. solitaria* were more common in this area than in areas I and II. No benthic species was solely limited to this area or found more frequently than in other areas.

Finfish-shellfish in mesohaline-to-polyhaline Area III. Twenty-six species were identified as being characteristic finfish-shellfish of this area (Table 7.6.7). Two species had their highest catch rates in this area, adult bay anchovy and all stages of silversides. Several other species had high catch rates in other areas in addition to Area III including adult gafftopsail catfish, juvenile Gulf menhaden, Gulf killifish, longnose killifish, sub-adult and juvenile spot, adult Atlantic croaker, subadult brown shrimp, juvenile red drum, and adult ladyfish (*Elops saurus*).

Vegetation in polyhaline-to-euhaline Area IV. Shoalgrass, widgeongrass and other saltwater algae were prominent submerged vegetation types near Shoalwater Bay, while smooth cordgrass was the dominant emergent

Table 7.6.6. Mean catch per sample for organisms caught in upper (area 1), mid (area 2), and lower (area 3) San Antonio Bay and Espiritu Santo Bay (area 4) by TPWD Coastal Fisheries Monitoring Program.

Area	Scientific Name	Common Name	Mean (M), standard error (SE), and sample size (n)												Percent of samples			
			Gillnet			Trawl			Bag seine			Oyster dredge			Gill-net	Trawl	Bag seine	Oyster dredge
			M	SE	n	M	SE	n	M	SE	n	M	SE	n				
3	*Acetes americanus*	Sergestid shrimp	.	.	.	.	.	.	0.00	0.05	456	.	.	.	.	.	0.22	.
2	*Achirus lineatus*	Lined sole	.	.	.	0.01	0.01	402	.	.	.	.	.	.	.	0.50	.	.
3	*Achirus lineatus*	Lined sole	.	.	.	0.00	0.00	825	0.00	0.05	456	.	.	.	.	0.48	0.22	.
4	*Achirus lineatus*	Lined sole	.	.	.	.	.	.	0.00	0.04	544	.	.	.	.	.	0.18	.
3	*Adinia xenica*	Diamond killifish	.	.	.	.	.	.	0.04	0.01	456	.	.	.	.	.	2.19	.
4	*Adinia xenica*	Diamond killifish	.	.	.	.	.	.	0.13	0.06	544	.	.	.	.	.	3.31	.
4	*Alectis ciliaris*	African pompano	0.00	0.06	313	.	.	.	.	.	.	.	.	.	0.32	.	.	.
2	*Alligator mississippiensis*	American alligator	0.00	0.06	262	.	.	.	.	.	.	.	.	.	0.38	.	.	.
1	*Alosa chrysochloris*	Skipjack herring	0.01	0.12	69	.	.	.	.	.	.	.	.	.	1.45	.	.	.
2	*Alosa chrysochloris*	Skipjack herring	0.03	0.02	262	.	.	.	0.04	0.34	78	.	.	.	1.53	.	1.28	.
3	*Alosa chrysochloris*	Skipjack herring	0.01	0.01	298	.	.	.	.	.	.	.	.	.	1.34	.	.	.
4	*Alosa chrysochloris*	Skipjack herring	0.00	0.06	313	.	.	.	.	.	.	.	.	.	0.32	.	.	.
2	*Alpheus heterochaelis*	Pistol shrimp	.	.	.	0.00	0.05	402	.	.	.	.	.	.	.	0.25	.	.
3	*Alpheus heterochaelis*	Pistol shrimp	.	.	.	0.00	0.00	825	.	.	.	.	.	.	.	0.36	.	.
4	*Alpheus heterochaelis*	Pistol shrimp	.	.	.	0.01	0.01	348	.	.	.	0.01	0.10	103	.	0.57	.	0.97
4	*Aluterus schoepfi*	Orange filefish	.	.	.	0.00	0.05	348	.	.	.	.	.	.	.	0.29	.	.
4	*Aluterus scriptus*	Scrawled filefish	.	.	.	0.00	0.05	348	.	.	.	.	.	.	.	0.29	.	.
2	*Anchoa hepsetus*	Striped anchovy	.	.	.	0.00	0.05	402	.	.	.	.	.	.	.	0.25	.	.
3	*Anchoa hepsetus*	Striped anchovy	.	.	.	0.00	0.00	825	0.01	0.01	456	.	.	.	.	0.24	0.66	.
4	*Anchoa hepsetus*	Striped anchovy	.	.	.	0.01	0.00	348	0.02	0.01	544	.	.	.	.	0.57	0.92	.
1	*Anchoa lyolepis*	Dusky anchovy	0.01	0.12	69	.	.	.	.	.	.	.	.	.	1.45	.	.	.
1	*Anchoa mitchilli*	Bay anchovy	.	.	.	1.65	0.44	66	0.49	0.16	65	.	.	.	.	36.4	18.5	.
2	*Anchoa mitchilli*	Bay anchovy	.	.	.	2.35	0.60	402	1.72	1.19	78	.	.	.	.	35.3	16.7	.
3	*Anchoa mitchilli*	Bay anchovy	.	.	.	3.52	0.41	825	3.19	0.83	456	.	.	.	.	40.6	24.3	.
4	*Anchoa mitchilli*	Bay anchovy	.	.	.	1.78	0.32	348	5.22	2.16	544	.	.	.	.	31	18	.
4	*Ancylopsetta quadrocellata*	Ocellated flounder	.	.	.	0.02	0.01	348	.	.	.	.	.	.	.	0.86	.	.
1	*Archosargus probatocephalus*	Sheepshead	0.81	0.21	69	0.14	0.07	66	.	.	.	.	.	.	31.9	7.58	.	.
2	*Archosargus probatocephalus*	Sheepshead	0.79	0.18	262	0.01	0.01	402	0.01	0.11	78	.	.	.	24.4	1.49	1.28	.
3	*Archosargus probatocephalus*	Sheepshead	0.92	0.15	298	0.02	0.01	825	0.04	0.01	456	.	.	.	28.9	1.94	2.85	.
4	*Archosargus probatocephalus*	Sheepshead	3.85	0.51	313	0.01	0.00	348	0.20	0.07	544	.	.	.	60.7	0.86	5.33	.
1	*Arius felis*	Sea catfish	6.28	1.13	69	0.47	0.17	66	0.42	0.36	65	.	.	.	63.8	18.2	3.08	.
2	*Arius felis*	Sea catfish	10.8	0.99	262	0.63	0.09	402	1.18	0.54	78	.	.	.	75.2	24.1	16.7	.
3	*Arius felis*	Sea catfish	14	1.06	298	0.55	0.07	825	0.31	0.08	456	.	.	.	81.2	21.2	7.89	.
4	*Arius felis*	Sea catfish	13.8	1.00	313	0.47	0.08	348	0.29	0.06	544	.	.	.	85	21	7.17	.
3	*Astroscopus y-graecum*	Southern stargazer	.	.	.	.	.	.	0.00	0.09	456	.	.	.	.	.	0.22	.
4	*Astroscopus y-graecum*	Southern stargazer	.	.	.	0.00	0.05	348	.	.	.	.	.	.	.	0.29	.	.
1	*Aurelia aurita*	Moon jellyfish	15.4	15.2	69	.	.	.	.	.	.	.	.	.	2.90	.	.	.
2	*Aurelia aurita*	Moon jellyfish	.	.	.	0.13	0.12	402	.	.	.	.	.	.	.	0.50	.	.
3	*Aurelia aurita*	Moon jellyfish	2.01	1.72	298	0.08	0.06	825	.	.	.	.	.	.	1.68	0.61	.	.
4	*Aurelia aurita*	Moon jellyfish	3.35	59.3	313	0.06	0.03	348	.	.	.	.	.	.	0.32	1.15	.	.
1	*Bagre marinus*	Gafftopsail catfish	3.38	1.30	69	0.52	0.29	66	.	.	.	.	.	.	26.1	9.09	.	.
2	*Bagre marinus*	Gafftopsail catfish	5.73	1.13	262	0.56	0.24	402	.	.	.	.	.	.	37.4	9.70	.	.
3	*Bagre marinus*	Gafftopsail catfish	5.96	0.76	298	1.29	0.33	825	0.00	0.00	456	.	.	.	48	13.5	0.44	.

Table 7.6.6. (continued) Mean catch per sample for organisms caught in upper (area 1), mid (area 2), and lower (area 3) San Antonio Bay and Espiritu Santo Bay (area 4) by TPWD Coastal Fisheries Monitoring Program.

Area	Scientific Name	Common Name	Mean (M), standard error (SE), and sample size (n)												Percent of samples			
			Gillnet			Trawl			Bag seine			Oyster dredge			Gill-net	Trawl	Bag seine	Oyster dredge
			M	SE	n	M	SE	n	M	SE	n	M	SE	n				
4	*Bagre marinus*	Gafftopsail catfish	7.79	1.04	313	1.03	0.42	348	0.00	0.04	544	.	.	.	50.2	10.6	0.18	.
1	*Bairdiella chrysoura*	Silver perch	0.01	0.12	69	0.20	0.09	66	0.02	0.12	65	.	.	.	1.45	12.1	1.54	.
2	*Bairdiella chrysoura*	Silver perch	0.05	0.02	262	0.27	0.05	402	0.09	0.07	78	.	.	.	2.29	11.9	3.85	.
3	*Bairdiella chrysoura*	Silver perch	0.02	0.01	298	0.34	0.05	825	0.33	0.13	456	.	.	.	1.68	12.4	5.04	.
4	*Bairdiella chrysoura*	Silver perch	0.03	0.01	313	0.47	0.11	348	0.35	0.09	544	.	.	.	2.88	19	8.27	.
4	*Balistes capriscus*	Gray triggerfish	.	.	.	0.00	0.05	348	.	.	.	.	.	.	.	0.29	.	.
2	*Beroe ovata*	Large comb jelly	.	.	.	1.88	1.27	402	0.77	0.65	78	.	.	.	.	1.74	3.85	.
3	*Beroe ovata*	Large comb jelly	.	.	.	5.42	1.92	825	0.26	0.16	456	.	.	.	.	2.79	1.32	.
4	*Beroe ovata*	Large comb jelly	.	.	.	1.87	1.46	348	.	.	.	.	.	.	.	1.15	.	.
1	*Brevoortia gunteri*	Finescale menhaden	0.14	0.11	69	.	.	.	.	.	.	.	.	.	4.35	.	.	.
2	*Brevoortia gunteri*	Finescale menhaden	0.81	0.50	262	.	.	.	0.03	0.22	78	.	.	.	8.78	.	1.28	.
3	*Brevoortia gunteri*	Finescale menhaden	0.49	0.13	298	0.02	0.01	825	0.06	1.26	456	.	.	.	12.1	0.24	0.22	.
4	*Brevoortia gunteri*	Finescale menhaden	0.88	0.22	313	.	.	.	0.04	1.03	544	.	.	.	16	.	0.18	.
1	*Brevoortia patronus*	Gulf menhaden	0.96	0.61	69	3.18	0.68	66	45.1	37.2	65	.	.	.	14.5	54.5	21.5	.
2	*Brevoortia patronus*	Gulf menhaden	1.52	0.53	262	7.58	2.01	402	7.96	7.60	78	.	.	.	21	53.5	7.69	.
3	*Brevoortia patronus*	Gulf menhaden	2.06	0.34	298	7.89	1.38	825	11.9	5.27	456	.	.	.	33.9	51	8.99	.
4	*Brevoortia patronus*	Gulf menhaden	2.64	0.44	313	1.97	0.59	348	9.45	5.54	544	.	.	.	34.8	23	9.74	.
2	*Busycon perversum*	Lightning whelk	.	.	.	0.00	0.10	402	.	.	.	.	.	.	.	0.25	.	.
3	*Busycon perversum*	Lightning whelk	0.00	0.06	298	.	.	.	.	.	.	.	.	.	0.34	.	.	.
1	*Callinectes sapidus*	Blue crab	1.54	0.39	69	6.20	1.74	66	1.22	0.25	65	0.14	0.35	7	37.7	51.5	47.7	14.3
2	*Callinectes sapidus*	Blue crab	1.08	0.15	262	6.67	0.55	402	1.64	0.51	78	0.04	0.01	349	27.9	77.9	37.2	3.15
3	*Callinectes sapidus*	Blue crab	1.22	0.21	298	4.27	0.30	825	1.88	0.21	456	0.05	0.02	171	30.9	56.4	42.3	3.51
4	*Callinectes sapidus*	Blue crab	1.25	0.17	313	1.41	0.16	348	2.72	0.30	544	0.06	0.02	103	35.1	39.1	50.6	5.83
1	*Callinectes similis*	Lesser blue crab	.	.	.	.	.	.	0.05	0.37	65	.	.	.	.	.	1.54	.
2	*Callinectes similis*	Lesser blue crab	.	.	.	0.08	0.02	402	.	.	.	.	.	.	.	6.22	.	.
3	*Callinectes similis*	Lesser blue crab	.	.	.	0.38	0.13	825	0.02	0.33	456	.	.	.	.	8.61	0.22	.
4	*Callinectes similis*	Lesser blue crab	0.00	0.06	313	0.23	0.05	348	0.07	0.02	544	.	.	.	0.32	10.3	2.76	.
4	*Cantherhines pullus*	Orangespot filefish	.	.	.	0.00	0.05	348	.	.	.	.	.	.	.	0.29	.	.
4	*Caranx bartholomaei*	Yellow jack	0.00	0.06	313	.	.	.	.	.	.	.	.	.	0.32	.	.	.
1	*Caranx hippos*	Crevalle jack	0.01	0.12	69	.	.	.	.	.	.	.	.	.	1.45	.	.	.
2	*Caranx hippos*	Crevalle jack	0.02	0.01	262	.	.	.	.	.	.	.	.	.	1.53	.	.	.
3	*Caranx hippos*	Crevalle jack	0.02	0.01	298	0.01	0.00	825	0.04	0.02	456	.	.	.	1.68	0.85	1.75	.
4	*Caranx hippos*	Crevalle jack	0.08	0.02	313	0.04	0.02	348	0.01	0.01	544	.	.	.	5.43	1.44	0.92	.
4	*Caranx latus*	Horse-eye jack	.	.	.	0.01	0.01	348	.	.	.	.	.	.	.	0.57	.	.
4	*Carcharhinus acronotus*	Blacknose shark	0.00	0.06	313	.	.	.	.	.	.	.	.	.	0.32	.	.	.
3	*Carcharhinus brevipinna*	Spinner shark	0.01	0.23	298	.	.	.	.	.	.	.	.	.	0.34	.	.	.
4	*Carcharhinus brevipinna*	Spinner shark	0.02	0.01	313	.	.	.	.	.	.	.	.	.	1.28	.	.	.
2	*Carcharhinus isodon*	Finetooth shark	0.02	0.01	262	.	.	.	.	.	.	.	.	.	0.76	.	.	.
3	*Carcharhinus isodon*	Finetooth shark	0.08	0.06	298	.	.	.	.	.	.	.	.	.	1.68	.	.	.
4	*Carcharhinus isodon*	Finetooth shark	0.22	0.06	313	.	.	.	.	.	.	.	.	.	6.39	.	.	.
1	*Carcharhinus leucas*	Bull shark	0.42	0.11	69	.	.	.	.	.	.	.	.	.	21.7	.	.	.
2	*Carcharhinus leucas*	Bull shark	0.53	0.08	262	.	.	.	.	.	.	.	.	.	25.2	.	.	.
3	*Carcharhinus leucas*	Bull shark	0.71	0.11	298	.	.	.	.	.	.	.	.	.	30.5	.	.	.

Table 7.6.6. (continued) Mean catch per sample for organisms caught in upper (area 1), mid (area 2), and lower (area 3) San Antonio Bay and Espiritu Santo Bay (area 4) by TPWD Coastal Fisheries Monitoring Program.

Area	Scientific Name	Common Name	Mean (M), standard error (SE), and sample size (n)												Percent of samples			
			Gillnet			Trawl			Bag seine			Oyster dredge			Gill-net	Trawl	Bag seine	Oyster dredge
			M	SE	n	M	SE	n	M	SE	n	M	SE	n				
4	*Carcharhinus leucas*	Bull shark	0.82	0.11	313	.	.	.	.	.	.	.	.	.	31.3	.	.	.
2	*Carcharhinus limbatus*	Blacktip shark	0.00	0.06	262	.	.	.	.	.	.	.	.	.	0.38	.	.	.
3	*Carcharhinus limbatus*	Blacktip shark	0.00	0.06	298	.	.	.	.	.	.	.	.	.	0.34	.	.	.
4	*Carcharhinus limbatus*	Blacktip shark	0.02	0.01	313	.	.	.	.	.	.	.	.	.	1.60	.	.	.
3	*Carcharhinus plumbeus*	Sandbar shark	0.03	0.02	298	.	.	.	.	.	.	.	.	.	0.67	.	.	.
4	*Carcharhinus plumbeus*	Sandbar shark	0.00	0.06	313	.	.	.	.	.	.	.	.	.	0.32	.	.	.
1	*Chaetodipterus faber*	Atlantic spadefish	0.01	0.12	69	.	.	.	.	.	.	.	.	.	1.45	.	.	.
2	*Chaetodipterus faber*	Atlantic spadefish	0.01	0.12	262	.	.	.	.	.	.	.	.	.	0.38	.	.	.
3	*Chaetodipterus faber*	Atlantic spadefish	0.00	0.06	298	0.01	0.01	825	.	.	.	.	.	.	0.34	0.36	.	.
4	*Chaetodipterus faber*	Atlantic spadefish	0.08	0.02	313	0.01	0.00	348	0.01	0.00	544	.	.	.	5.43	0.86	0.37	.
4	*Chasmodes bosquianus*	Striped blenny	.	.	.	0.00	0.05	348	0.00	0.04	544	.	.	.	.	0.29	0.18	.
3	*Chelonia mydas*	Green sea turtle	0.00	0.06	298	.	.	.	.	.	.	.	.	.	0.34	.	.	.
4	*Chilomycterus schoepfi*	Striped burrfish	.	.	.	0.00	0.05	348	.	.	.	.	.	.	.	0.29	.	.
4	*Chione cancellata*	Cross-barred venus	.	.	.	.	.	.	.	.	.	0.01	0.10	103	.	.	.	0.97
2	*Chloroscombrus chrysurus*	Atlantic bumper	.	.	.	0.01	0.01	402	.	.	.	.	.	.	.	0.50	.	.
3	*Chloroscombrus chrysurus*	Atlantic bumper	.	.	.	0.06	0.02	825	0.00	0.05	456	.	.	.	.	3.27	0.22	.
4	*Chloroscombrus chrysurus*	Atlantic bumper	.	.	.	4.85	2.54	348	.	.	.	.	.	.	.	12.4	.	.
1	*Citharichthys spilopterus*	Bay whiff	.	.	.	0.09	0.04	66	.	.	.	.	.	.	.	7.58	.	.
2	*Citharichthys spilopterus*	Bay whiff	.	.	.	0.20	0.05	402	0.15	1.35	78	.	.	.	.	6.72	1.28	.
3	*Citharichthys spilopterus*	Bay whiff	.	.	.	0.09	0.01	825	0.06	0.02	456	.	.	.	.	5.94	3.51	.
4	*Citharichthys spilopterus*	Bay whiff	.	.	.	0.11	0.02	348	0.07	0.02	544	.	.	.	.	7.76	3.68	.
3	Class Osteichthyes	Class bony fishes	.	.	.	0.00	0.03	825	.	.	.	.	.	.	.	0.12	.	.
2	Class Polychaeta	Polychaete worms	.	.	.	.	.	.	.	.	.	0.04	0.01	349	.	.	.	3.15
3	Class Polychaeta	Polychaete worms	.	.	.	.	.	.	.	.	.	0.05	0.03	171	.	.	.	2.92
4	Class Polychaeta	Polychaete worms	.	.	.	.	.	.	.	.	.	0.03	0.02	103	.	.	.	1.94
2	*Clibanarius vittatus*	Striped hermit crab	.	.	.	0.00	0.05	402	0.06	0.04	78	.	.	.	.	0.25	3.85	.
3	*Clibanarius vittatus*	Striped hermit crab	0.01	0.12	298	0.01	0.00	825	0.02	0.01	456	.	.	.	0.34	0.48	1.54	.
4	*Clibanarius vittatus*	Striped hermit crab	0.00	0.06	313	0.02	0.01	348	0.09	0.04	544	.	.	.	0.32	1.44	3.86	.
1	*Crassostrea virginica*	Eastern oyster	.	.	.	0.65	0.36	66	0.02	0.12	65	4.71	1.66	7	.	6.06	1.54	71.4
2	*Crassostrea virginica*	Eastern oyster	0.01	0.18	262	1.09	0.38	402	0.05	0.45	78	45.2	2.73	349	0.38	5.47	1.28	98.6
3	*Crassostrea virginica*	Eastern oyster	0.11	1.97	298	0.20	0.09	825	0.02	0.01	456	42.5	4.07	171	0.34	2.42	0.66	99.4
4	*Crassostrea virginica*	Eastern oyster	0.17	0.13	313	0.07	0.03	348	0.01	0.00	544	35	3.89	103	0.64	2.30	0.37	95.1
1	*Cynoscion arenarius*	Sand seatrout	0.01	0.12	69	0.03	0.02	66	.	.	.	.	.	.	1.45	3.03	.	.
2	*Cynoscion arenarius*	Sand seatrout	0.00	0.06	262	0.09	0.02	402	.	.	.	.	.	.	0.38	7.46	.	.
3	*Cynoscion arenarius*	Sand seatrout	0.01	0.01	298	0.10	0.02	825	0.02	0.01	456	.	.	.	1.34	6.18	1.10	.
4	*Cynoscion arenarius*	Sand seatrout	0.04	0.01	313	0.06	0.02	348	0.02	0.01	544	.	.	.	3.83	4.60	1.29	.
1	*Cynoscion nebulosus*	Spotted seatrout	2.09	0.55	69	0.09	0.04	66	0.14	0.06	65	.	.	.	47.8	7.58	7.69	.
2	*Cynoscion nebulosus*	Spotted seatrout	2.95	0.40	262	0.08	0.02	402	0.19	0.09	78	.	.	.	59.5	5.97	7.69	.
3	*Cynoscion nebulosus*	Spotted seatrout	5.78	0.44	298	0.04	0.01	825	0.40	0.07	456	.	.	.	85.6	3.39	13.8	.
4	*Cynoscion nebulosus*	Spotted seatrout	8.12	0.67	313	0.03	0.01	348	0.35	0.08	544	.	.	.	90.7	2.30	12.5	.
2	*Cynoscion nothus*	Silver seatrout	.	.	.	0.01	0.01	402	.	.	.	.	.	.	.	0.75	.	.
4	*Cynoscion nothus*	Silver seatrout	.	.	.	0.00	0.05	348	.	.	.	.	.	.	.	0.29	.	.
1	*Cyprinodon variegatus*	Sheepshead minnow	.	.	.	.	.	.	33.4	21.9	65	.	.	.	.	.	38.5	.

Table 7.6.6. (continued) Mean catch per sample for organisms caught in upper (area 1), mid (area 2), and lower (area 3) San Antonio Bay and Espiritu Santo Bay (area 4) by TPWD Coastal Fisheries Monitoring Program.

Area	Scientific Name	Common Name	Mean (M), standard error (SE), and sample size (n)												Percent of samples			
			Gillnet			Trawl			Bag seine			Oyster dredge			Gill-net	Trawl	Bag seine	Oyster dredge
			M	SE	n	M	SE	n	M	SE	n	M	SE	n				
2	*Cyprinodon variegatus*	Sheepshead minnow	.	.	.	0.00	0.00	402	11.9	3.54	78	.	.	.	.	0.50	48.7	.
3	*Cyprinodon variegatus*	Sheepshead minnow	.	.	.	0.01	0.17	825	12.6	2.77	456	.	.	.	.	0.12	36	.
4	*Cyprinodon variegatus*	Sheepshead minnow	.	.	.	0.00	0.05	348	13.6	3.01	544	.	.	.	.	0.29	38.2	.
1	*Cyprinus carpio*	Common carp	0.46	0.26	69	.	.	.	.	.	.	.	.	.	10.1	.	.	.
2	*Cyprinus carpio*	Common carp	0.09	0.03	262	.	.	.	.	.	.	.	.	.	5.73	.	.	.
3	*Cyprinus carpio*	Common carp	0.34	0.20	298	0.00	0.00	825	0.01	0.00	456	.	.	.	2.01	0.24	0.44	.
2	*Dactylometra quinquecirrha*	Stinging nettle	.	.	.	0.25	0.13	402	.	.	.	.	.	.	.	2.99	.	.
3	*Dactylometra quinquecirrha*	Stinging nettle	0.02	0.29	298	0.41	0.12	825	0.02	0.02	456	.	.	.	0.34	4.24	0.44	.
4	*Dactylometra quinquecirrha*	Stinging nettle	0.64	11.3	313	0.11	0.04	348	.	.	.	.	.	.	0.32	2.30	.	.
2	*Dasyatis americana*	Southern stingray	0.01	0.01	262	.	.	.	.	.	.	.	.	.	0.76	.	.	.
3	*Dasyatis americana*	Southern stingray	0.00	0.06	298	.	.	.	.	.	.	.	.	.	0.34	.	.	.
4	*Dasyatis americana*	Southern stingray	0.01	0.00	313	.	.	.	.	.	.	.	.	.	0.64	.	.	.
1	*Dasyatis sabina*	Atlantic stingray	0.04	0.02	69	0.03	0.02	66	0.06	0.04	65	.	.	.	4.35	3.03	4.62	.
2	*Dasyatis sabina*	Atlantic stingray	0.13	0.03	262	0.04	0.01	402	0.28	0.14	78	.	.	.	8.78	2.99	7.69	.
3	*Dasyatis sabina*	Atlantic stingray	0.07	0.02	298	0.02	0.01	825	0.05	0.01	456	.	.	.	5.37	1.82	3.95	.
4	*Dasyatis sabina*	Atlantic stingray	0.08	0.02	313	0.03	0.01	348	0.06	0.01	544	.	.	.	7.03	2.30	3.49	.
4	*Diodon hystrix*	Porcupinefish	0.00	0.06	313	.	.	.	.	.	.	.	.	.	0.32	.	.	.
1	*Dorosoma cepedianum*	Gizzard shad	4.96	1.49	69	0.17	0.08	66	0.43	0.23	65	.	.	.	36.8	9.09	6.15	.
2	*Dorosoma cepedianum*	Gizzard shad	10.2	1.31	262	0.05	0.02	402	0.01	0.11	78	.	.	.	63.4	2.24	1.28	.
3	*Dorosoma cepedianum*	Gizzard shad	3.23	0.40	298	0.09	0.02	825	0.02	0.01	456	.	.	.	54.4	3.88	0.66	.
4	*Dorosoma cepedianum*	Gizzard shad	0.68	0.14	313	0.05	0.04	348	0.28	0.24	544	.	.	.	21.1	1.15	1.84	.
1	*Dorosoma petenense*	Threadfin shad	.	.	.	0.05	0.03	66	.	.	.	.	.	.	.	3.03	.	.
2	*Dorosoma petenense*	Threadfin shad	0.01	0.01	262	0.04	0.01	402	.	.	.	.	.	.	0.76	3.23	.	.
3	*Dorosoma petenense*	Threadfin shad	.	.	.	0.03	0.01	825	.	.	.	.	.	.	.	2.18	.	.
4	*Dorosoma petenense*	Threadfin shad	.	.	.	0.02	0.01	348	0.01	0.01	544	.	.	.	.	1.72	0.55	.
3	*Echiophis intertinctus*	Spotted spoon-nose eel	0.01	0.12	298	.	.	.	.	.	.	.	.	.	0.34	.	.	.
1	*Elops saurus*	Ladyfish	0.06	0.05	69	0.02	0.12	66	0.14	0.12	65	.	.	.	2.90	1.52	3.08	.
2	*Elops saurus*	Ladyfish	0.20	0.07	262	0.00	0.00	402	0.05	0.45	78	.	.	.	6.87	0.50	1.28	.
3	*Elops saurus*	Ladyfish	0.45	0.07	298	0.00	0.07	825	0.02	0.01	456	.	.	.	21.1	0.12	1.10	.
4	*Elops saurus*	Ladyfish	0.36	0.05	313	.	.	.	0.01	0.01	544	.	.	.	18.8	.	1.10	.
4	*Erotelis smaragdus*	Emerald sleeper	.	.	.	.	.	.	0.00	0.04	544	.	.	.	.	.	0.18	.
2	*Etropus crossotus*	Fringed flounder	.	.	.	0.00	0.05	402	.	.	.	.	.	.	.	0.25	.	.
3	*Etropus crossotus*	Fringed flounder	.	.	.	0.04	0.02	825	.	.	.	.	.	.	.	1.09	.	.
4	*Etropus crossotus*	Fringed flounder	.	.	.	0.02	0.01	348	.	.	.	.	.	.	.	1.72	.	.
1	*Eucinostomus argenteus*	Spotfin mojarra	.	.	.	0.02	0.12	66	0.03	0.02	65	.	.	.	.	1.52	3.08	.
2	*Eucinostomus argenteus*	Spotfin mojarra	.	.	.	.	.	.	0.04	0.02	78	.	.	.	.	.	3.85	.
3	*Eucinostomus argenteus*	Spotfin mojarra	.	.	.	0.00	0.03	825	0.37	0.17	456	.	.	.	.	0.12	7.68	.
4	*Eucinostomus argenteus*	Spotfin mojarra	.	.	.	0.05	0.03	348	1.71	0.74	544	.	.	.	.	1.72	14.3	.
3	*Eucinostomus gula*	Silver jenny	.	.	.	0.00	0.00	825	0.00	0.05	456	.	.	.	.	0.24	0.22	.
4	*Eucinostomus gula*	Silver jenny	.	.	.	0.01	0.01	348	0.15	0.08	544	.	.	.	.	0.57	1.29	.
3	*Eucinostomus lefroyi*	Mottled mojarra	.	.	.	.	.	.	0.00	0.05	456	.	.	.	.	.	0.22	.
4	*Eucinostomus lefroyi*	Mottled mojarra	.	.	.	.	.	.	0.00	0.09	544	.	.	.	.	.	0.18	.
3	*Evorthodus lyricus*	Lyre goby	.	.	.	.	.	.	0.00	0.05	456	.	.	.	.	.	0.22	.

Table 7.6.6. (continued) Mean catch per sample for organisms caught in upper (area 1), mid (area 2), and lower (area 3) San Antonio Bay and Espiritu Santo Bay (area 4) by TPWD Coastal Fisheries Monitoring Program.

Area	Scientific Name	Common Name	Mean (M), standard error (SE), and sample size (n)												Percent of samples			
			Gillnet			Trawl			Bag seine			Oyster dredge			Gill-net	Trawl	Bag seine	Oyster dredge
			M	SE	n	M	SE	n	M	SE	n	M	SE	n				
2	Family Astacidae	Family crayfishes	.	.	.	.	.	.	0.01	0.11	78	.	.	.	.	.	1.28	.
3	Family Astacidae	Family crayfishes	.	.	.	0.00	0.03	825	.	.	.	.	.	.	.	0.12	.	.
3	Family Bothidae	Family left-hand flounder	.	.	.	0.00	0.07	825	.	.	.	.	.	.	.	0.12	.	.
2	Family Carangidae	Family jacks	.	.	.	0.02	0.02	402	0.01	0.11	78	.	.	.	.	0.75	1.28	.
3	Family Carangidae	Family jacks	.	.	.	0.00	0.03	825	.	.	.	.	.	.	.	0.12	.	.
4	Family Carangidae	Family jacks	.	.	.	0.01	0.11	348	0.04	0.02	544	.	.	.	.	0.29	1.65	.
1	Family Clupeidae	Family herrings	.	.	.	.	.	.	0.02	0.12	65	.	.	.	.	.	1.54	.
2	Family Clupeidae	Family herrings	.	.	.	0.07	0.04	402	.	.	.	.	.	.	.	1.24	.	.
3	Family Clupeidae	Family herrings	.	.	.	0.27	0.12	825	0.04	0.02	456	.	.	.	.	2.18	1.32	.
4	Family Clupeidae	Family herrings	.	.	.	0.09	0.06	348	0.07	0.04	544	.	.	.	.	1.15	0.55	.
2	Family Engraulidae	Family anchovies	.	.	.	0.01	0.25	402	.	.	.	.	.	.	.	0.25	.	.
3	Family Engraulidae	Family anchovies	.	.	.	0.02	0.01	825	.	.	.	.	.	.	.	0.24	.	.
4	Family Engraulidae	Family anchovies	.	.	.	0.05	0.03	348	.	.	.	.	.	.	.	1.72	.	.
1	Family Mugilidae	Family mullets	.	.	.	.	.	.	1.00	0.68	65	.	.	.	.	.	6.15	.
2	Family Mugilidae	Family mullets	.	.	.	.	.	.	0.35	0.19	78	.	.	.	.	.	5.13	.
3	Family Mugilidae	Family mullets	.	.	.	.	.	.	0.31	0.16	456	.	.	.	.	.	3.73	.
4	Family Mugilidae	Family mullets	.	.	.	.	.	.	0.30	0.14	544	.	.	.	.	.	3.13	.
1	Family Penaeidae	Family penaeids	.	.	.	.	.	.	0.06	0.49	65	.	.	.	.	.	1.54	.
3	Family Penaeidae	Family penaeids	.	.	.	0.00	0.03	825	2.74	58.4	456	.	.	.	.	0.12	0.22	.
4	Family Penaeidae	Family penaeids	.	.	.	.	.	.	0.00	0.04	544	.	.	.	.	.	0.18	.
1	Family Pomacentridae	Family damselfishes	0.00	0.00	69	.	.	.	.	.	.	.	.	.	1.45	.	.	.
2	Family Pomacentridae	Family damselfishes	0.00	0.00	262	.	.	.	.	.	.	.	.	.	0.38	.	.	.
3	Family Pomacentridae	Family damselfishes	0.00	0.00	298	.	.	.	.	.	.	.	.	.	0.34	.	.	.
4	Family Pomacentridae	Family damselfishes	0.00	0.00	313	.	.	.	.	.	.	.	.	.	0.96	.	.	.
4	Family Sciaenidae	Family drums	.	.	.	.	.	.	0.00	0.04	544	.	.	.	.	.	0.18	.
3	Family Sphyrnidae	Family hammerhead shark	0.00	0.06	298	.	.	.	.	.	.	.	.	.	0.34	.	.	.
4	Family Sphyrnidae	Family hammerhead shark	0.02	0.40	313	.	.	.	.	.	.	.	.	.	0.32	.	.	.
2	Family Xanthidae	Family mud crabs	.	.	.	0.00	0.05	402	.	.	.	0.15	0.03	349	.	0.25	.	8.31
3	Family Xanthidae	Family mud crabs	.	.	.	0.00	0.03	825	0.00	0.05	456	0.08	0.02	171	.	0.12	0.22	6.43
4	Family Xanthidae	Family mud crabs	.	.	.	0.02	0.01	348	0.01	0.13	544	0.02	0.01	103	.	1.72	0.18	1.94
1	*Fundulus grandis*	Gulf killifish	.	.	.	.	.	.	4.23	2.16	65	.	.	.	.	.	23.1	.
2	*Fundulus grandis*	Gulf killifish	.	.	.	.	.	.	3.36	1.10	78	.	.	.	.	.	29.5	.
3	*Fundulus grandis*	Gulf killifish	.	.	.	0.01	0.17	825	3.69	0.65	456	.	.	.	.	0.12	30.9	.
4	*Fundulus grandis*	Gulf killifish	.	.	.	.	.	.	1.65	0.42	544	.	.	.	.	.	24.3	.
2	*Fundulus pulvereus*	Bayou killifish	.	.	.	.	.	.	0.01	0.11	78	.	.	.	.	.	1.28	.
3	*Fundulus pulvereus*	Bayou killifish	.	.	.	.	.	.	0.04	0.02	456	.	.	.	.	.	1.10	.
4	*Fundulus pulvereus*	Bayou killifish	.	.	.	.	.	.	0.01	0.01	544	.	.	.	.	.	0.74	.
1	*Fundulus similis*	Longnose killifish	.	.	.	.	.	.	0.09	0.06	65	.	.	.	.	.	4.62	.
2	*Fundulus similis*	Longnose killifish	.	.	.	.	.	.	1.14	0.44	78	.	.	.	.	.	23.1	.
3	*Fundulus similis*	Longnose killifish	.	.	.	.	.	.	3.45	0.55	456	.	.	.	.	.	29.2	.
4	*Fundulus similis*	Longnose killifish	.	.	.	.	.	.	4.47	0.57	544	.	.	.	.	.	34.4	.
1	*Gerres cinereus*	Yellowfin mojarra	.	.	.	0.03	0.24	66	.	.	.	.	.	.	.	1.52	.	.
2	*Gobiesox punctulatus*	Stippled clingfish	.	.	.	0.00	0.05	402	.	.	.	.	.	.	.	0.25	.	.

Table 7.6.6. (continued) Mean catch per sample for organisms caught in upper (area 1), mid (area 2), and lower (area 3) San Antonio Bay and Espiritu Santo Bay (area 4) by TPWD Coastal Fisheries Monitoring Program.

Area	Scientific Name	Common Name	Mean (M), standard error (SE), and sample size (n)												Percent of samples			
			Gillnet			Trawl			Bag seine			Oyster dredge			Gill-net	Trawl	Bag seine	Oyster dredge
			M	SE	n	M	SE	n	M	SE	n	M	SE	n				
2	*Gobiesox strumosus*	Skilletfish	.	.	.	0.00	0.05	402	.	.	.	0.00	0.05	349	.	0.25	.	0.29
3	*Gobiesox strumosus*	Skilletfish	.	.	.	0.00	0.00	825	0.00	0.00	456	.	.	.	.	0.36	0.44	.
4	*Gobioides broussoneti*	Violet goby	.	.	.	0.00	0.05	348	.	.	.	.	.	.	.	0.29	.	.
4	*Gobiomorus dormitor*	Bigmouth sleeper	.	.	.	.	.	.	0.00	0.00	544	.	.	.	.	.	0.37	.
3	*Gobionellus boleosoma*	Darter goby	.	.	.	.	.	.	0.02	0.42	456	.	.	.	.	.	0.22	.
4	*Gobionellus boleosoma*	Darter goby	.	.	.	.	.	.	0.09	0.04	544	.	.	.	.	.	2.39	.
4	*Gobionellus hastatus*	Sharptail goby	.	.	.	.	.	.	0.02	0.01	544	.	.	.	.	.	1.65	.
1	*Gobiosoma bosci*	Naked goby	.	.	.	.	.	.	0.15	0.05	65	.	.	.	.	.	13.8	.
2	*Gobiosoma bosci*	Naked goby	.	.	.	0.00	0.00	402	.	.	.	0.01	0.01	349	.	0.50	.	0.86
3	*Gobiosoma bosci*	Naked goby	.	.	.	0.00	0.00	825	0.07	0.02	456	0.02	0.01	171	.	0.24	3.29	1.75
4	*Gobiosoma bosci*	Naked goby	.	.	.	0.00	0.05	348	0.07	0.02	544	.	.	.	.	0.29	3.68	.
3	*Gobiosoma robustum*	Code goby	.	.	.	.	.	.	0.00	0.00	456	.	.	.	.	.	0.44	.
4	*Gobiosoma robustum*	Code goby	.	.	.	.	.	.	0.01	0.01	544	.	.	.	.	.	0.92	.
3	*Harengula jaguana*	Scaled sardine	.	.	.	0.00	0.03	825	0.01	0.01	456	.	.	.	.	0.12	0.44	.
4	*Harengula jaguana*	Scaled sardine	.	.	.	0.01	0.01	348	0.11	0.06	544	.	.	.	.	0.86	1.29	.
2	*Hemicaranx amblyrhynchus*	Bluntnose jack	.	.	.	0.01	0.01	402	.	.	.	.	.	.	.	0.50	.	.
3	*Hemicaranx amblyrhynchus*	Bluntnose jack	.	.	.	0.00	0.00	825	0.00	0.00	456	.	.	.	.	0.36	0.44	.
4	*Hemicaranx amblyrhynchus*	Bluntnose jack	.	.	.	0.02	0.01	348	0.02	0.01	544	.	.	.	.	1.15	0.37	.
4	*Hippocampus zosterae*	Dwarf seahorse	.	.	.	0.01	0.00	348	.	.	.	.	.	.	.	0.57	.	.
4	*Hippolysmata wurdemanni*	Peppermint shrimp	.	.	.	0.01	0.11	348	.	.	.	.	.	.	.	0.29	.	.
4	*Histrio histrio*	Sargassumfish	.	.	.	.	.	.	0.00	0.04	544	.	.	.	.	.	0.18	.
3	*Hyporhamphus unifasciatus*	Halfbeak	.	.	.	.	.	.	0.02	0.01	456	.	.	.	.	.	0.44	.
4	*Hyporhamphus unifasciatus*	Halfbeak	.	.	.	0.00	0.05	348	0.00	0.09	544	.	.	.	.	0.29	0.18	.
1	*Ictalurus furcatus*	Blue catfish	3.86	0.95	69	1.86	0.56	66	0.05	0.03	65	.	.	.	42	33.3	4.62	.
2	*Ictalurus furcatus*	Blue catfish	0.89	0.32	262	0.36	0.14	402	.	.	.	.	.	.	9.92	5.97	.	.
3	*Ictalurus furcatus*	Blue catfish	0.44	0.22	298	0.21	0.07	825	.	.	.	.	.	.	2.35	3.76	.	.
4	*Ictalurus furcatus*	Blue catfish	.	.	.	.	.	.	0.11	0.05	544	.	.	.	.	.	1.84	.
2	*Ictalurus punctatus*	Channel catfish	0.00	0.06	262	0.01	0.20	402	.	.	.	.	.	.	0.38	0.25	.	.
3	*Ictalurus punctatus*	Channel catfish	0.01	0.12	298	.	.	.	.	.	.	.	.	.	0.34	.	.	.
4	*Ictalurus punctatus*	Channel catfish	.	.	.	.	.	.	0.01	0.00	544	.	.	.	.	.	0.55	.
1	*Ictiobus bubalus*	Smallmouth buffalo	0.65	0.31	69	.	.	.	0.02	0.12	65	.	.	.	11.6	.	1.54	.
2	*Ictiobus bubalus*	Smallmouth buffalo	0.02	0.01	262	.	.	.	0.01	0.11	78	.	.	.	1.15	.	1.28	.
3	*Ictiobus bubalus*	Smallmouth buffalo	0.05	0.04	298	.	.	.	.	.	.	.	.	.	1.01	.	.	.
1	*Ischadium recurvum*	Hooked mussel	.	.	.	0.41	3.30	66	.	.	.	.	.	.	.	1.52	.	.
2	*Ischadium recurvum*	Hooked mussel	0.01	0.18	262	.	.	.	.	.	.	.	.	.	0.38	.	.	.
3	*Ischadium recurvum*	Hooked mussel	0.36	6.25	298	.	.	.	.	.	.	.	.	.	0.34	.	.	.
4	*Ischadium recurvum*	Hooked mussel	0.00	0.06	313	.	.	.	.	.	.	.	.	.	0.32	.	.	.
4	*Kyphosus incisor*	Yellow chub	0.00	0.06	313	.	.	.	.	.	.	.	.	.	0.32	.	.	.
4	*Lagocephalus laevigatus*	Smooth puffer	.	.	.	0.01	0.11	348	.	.	.	.	.	.	.	0.29	.	.
1	*Lagodon rhomboides*	Pinfish	0.01	0.12	69	0.70	0.34	66	0.51	0.24	65	.	.	.	1.45	16.7	9.23	.
2	*Lagodon rhomboides*	Pinfish	0.03	0.01	262	1.17	0.26	402	4.60	1.42	78	.	.	.	2.67	22.9	39.7	.
3	*Lagodon rhomboides*	Pinfish	0.11	0.03	298	1.67	0.24	825	8.93	1.48	456	.	.	.	5.37	28.4	43.6	.
4	*Lagodon rhomboides*	Pinfish	0.23	0.07	313	10.4	1.46	348	14.3	2.47	544	.	.	.	8.63	56.3	43.9	.

Table 7.6.6. (continued) Mean catch per sample for organisms caught in upper (area 1), mid (area 2), and lower (area 3) San Antonio Bay and Espiritu Santo Bay (area 4) by TPWD Coastal Fisheries Monitoring Program.

Area	Scientific Name	Common Name	Mean (M), standard error (SE), and sample size (n)												Percent of samples			
			Gillnet			Trawl			Bag seine			Oyster dredge			Gill-net	Trawl	Bag seine	Oyster dredge
			M	SE	n	M	SE	n	M	SE	n	M	SE	n				
4	*Larimus fasciatus*	Banded drum	.	.	.	0.00	0.05	348	.	.	.	.	.	.	.	0.29	.	.
1	*Leiostomus xanthurus*	Spot	0.51	0.26	69	0.61	0.40	66	1.25	0.61	65	.	.	.	13	15.2	16.9	.
2	*Leiostomus xanthurus*	Spot	0.38	0.08	262	1.89	0.40	402	2.99	1.18	78	.	.	.	16	27.4	24.4	.
3	*Leiostomus xanthurus*	Spot	0.97	0.14	298	5.94	0.75	825	10.7	2.36	456	.	.	.	31.5	42.5	35.1	.
4	*Leiostomus xanthurus*	Spot	1.37	0.23	313	6.88	0.94	348	4.96	0.88	544	.	.	.	30	48.6	31.3	.
1	*Lepisosteus oculatus*	Spotted gar	1.39	0.65	69	.	.	.	.	.	.	.	.	.	20.3	.	.	.
2	*Lepisosteus oculatus*	Spotted gar	1.06	0.22	262	.	.	.	.	.	.	.	.	.	20.2	.	.	.
3	*Lepisosteus oculatus*	Spotted gar	0.23	0.15	298	0.00	0.03	825	.	.	.	.	.	.	3.69	0.12	.	.
4	*Lepisosteus oculatus*	Spotted gar	0.01	0.11	313	.	.	.	.	.	.	.	.	.	0.32	.	.	.
1	*Lepisosteus osseus*	Longnose gar	2.28	0.73	69	.	.	.	.	.	.	.	.	.	23.2	.	.	.
2	*Lepisosteus osseus*	Longnose gar	1.12	0.27	262	.	.	.	.	.	.	.	.	.	15.3	.	.	.
3	*Lepisosteus osseus*	Longnose gar	0.02	0.01	298	.	.	.	.	.	.	.	.	.	1.68	.	.	.
4	*Lepisosteus osseus*	Longnose gar	0.01	0.00	313	.	.	.	.	.	.	.	.	.	0.64	.	.	.
1	*Lepisosteus platostomus*	Shortnose gar	0.71	0.22	69	0.02	0.12	66	.	.	.	.	.	.	18.8	1.52	.	.
2	*Lepisosteus platostomus*	Shortnose gar	0.43	0.12	262	.	.	.	.	.	.	.	.	.	10.7	.	.	.
3	*Lepisosteus platostomus*	Shortnose gar	0.23	0.13	298	.	.	.	.	.	.	.	.	.	4.03	.	.	.
4	*Lepisosteus platostomus*	Shortnose gar	0.02	0.34	313	.	.	.	.	.	.	.	.	.	0.32	.	.	.
1	*Lepisosteus spatula*	Alligator gar	1.36	0.27	69	.	.	.	.	.	.	.	.	.	44.9	.	.	.
2	*Lepisosteus spatula*	Alligator gar	1.62	0.18	262	0.01	0.00	402	.	.	.	.	.	.	46.9	0.75	.	.
3	*Lepisosteus spatula*	Alligator gar	0.79	0.10	298	.	.	.	.	.	.	.	.	.	31.9	.	.	.
4	*Lepisosteus spatula*	Alligator gar	0.21	0.05	313	.	.	.	0.00	0.04	544	.	.	.	13.4	.	0.18	.
1	*Libinia dubia*	Spider crab	0.01	0.12	69	.	.	.	.	.	.	.	.	.	1.45	.	.	.
2	*Libinia dubia*	Spider crab	.	.	.	0.00	0.05	402	.	.	.	.	.	.	.	0.25	.	.
3	*Libinia dubia*	Spider crab	0.00	0.06	298	0.00	0.00	825	.	.	.	.	.	.	0.34	0.24	.	.
4	*Libinia dubia*	Spider crab	0.03	0.01	313	0.01	0.01	348	.	.	.	.	.	.	1.92	1.15	.	.
4	*Libinia emarginata*	Spider crab	0.00	0.06	313	0.01	0.00	348	.	.	.	.	.	.	0.32	0.57	.	.
2	*Lobotes surinamensis*	Tripletail	0.01	0.01	262	.	.	.	.	.	.	.	.	.	0.76	.	.	.
3	*Lobotes surinamensis*	Tripletail	0.02	0.01	298	.	.	.	.	.	.	.	.	.	2.01	.	.	.
4	*Lobotes surinamensis*	Tripletail	0.07	0.02	313	.	.	.	.	.	.	.	.	.	6.07	.	.	.
2	*Lolliguncula brevis*	Brief squid	.	.	.	0.05	0.02	402	.	.	.	.	.	.	.	2.99	.	.
3	*Lolliguncula brevis*	Brief squid	.	.	.	0.32	0.05	825	.	.	.	.	.	.	.	13.3	.	.
4	*Lolliguncula brevis*	Brief squid	.	.	.	1.79	0.27	348	.	.	.	.	.	.	.	39.7	.	.
1	*Lucania parva*	Rainwater killifish	.	.	.	.	.	.	0.08	0.06	65	.	.	.	.	.	3.08	.
2	*Lucania parva*	Rainwater killifish	.	.	.	.	.	.	0.14	0.13	78	.	.	.	.	.	2.56	.
3	*Lucania parva*	Rainwater killifish	.	.	.	.	.	.	0.20	0.09	456	.	.	.	.	.	5.70	.
4	*Lucania parva*	Rainwater killifish	.	.	.	.	.	.	0.18	0.09	544	.	.	.	.	.	5.33	.
4	*Luidia clathrata*	Starfish	.	.	.	0.00	0.05	348	.	.	.	.	.	.	.	0.29	.	.
3	*Lutjanus griseus*	Gray snapper	0.01	0.01	298	.	.	.	0.00	0.05	456	.	.	.	1.34	.	0.22	.
4	*Lutjanus griseus*	Gray snapper	0.02	0.01	313	.	.	.	0.01	0.01	544	.	.	.	0.96	.	0.74	.
3	*Lutjanus jocu*	Dog snapper	.	.	.	0.00	0.03	825	.	.	.	.	.	.	.	0.12	.	.
1	*Macrobrachium ohione*	River shrimp	.	.	.	0.20	0.13	66	0.28	0.26	65	.	.	.	.	6.06	3.08	.
2	*Macrobrachium ohione*	River shrimp	.	.	.	0.04	0.01	402	0.36	3.15	78	.	.	.	.	3.73	1.28	.
3	*Macrobrachium ohione*	River shrimp	.	.	.	0.02	0.01	825	0.01	0.00	456	.	.	.	.	0.73	0.66	.

Table 7.6.6. (continued) Mean catch per sample for organisms caught in upper (area 1), mid (area 2), and lower (area 3) San Antonio Bay and Espiritu Santo Bay (area 4) by TPWD Coastal Fisheries Monitoring Program.

Area	Scientific Name	Common Name	Mean (M), standard error (SE), and sample size (n)												Percent of samples			
			Gillnet			Trawl			Bag seine			Oyster dredge			Gill-net	Trawl	Bag seine	Oyster dredge
			M	SE	n	M	SE	n	M	SE	n	M	SE	n				
4	*Macrobrachium ohione*	River shrimp	.	.	.	.	.	.	0.12	0.11	544	.	.	.	.	.	1.10	.
1	*Malaclemys terrapin*	Diamondback terrapin	0.01	0.12	69	.	.	.	.	.	.	.	.	.	1.45	.	.	.
2	*Malaclemys terrapin*	Diamondback terrapin	.	.	.	0.00	0.00	402	.	.	.	.	.	.	.	0.50	.	.
3	*Malaclemys terrapin*	Diamondback terrapin	0.01	0.00	298	0.00	0.03	825	.	.	.	.	.	.	0.67	0.12	.	.
1	*Megalops atlanticus*	Tarpon	0.01	0.12	69	.	.	.	.	.	.	.	.	.	1.45	.	.	.
2	*Megalops atlanticus*	Tarpon	0.02	0.01	262	.	.	.	.	.	.	.	.	.	1.53	.	.	.
3	*Megalops atlanticus*	Tarpon	0.01	0.01	298	0.00	0.03	825	.	.	.	.	.	.	1.01	0.12	.	.
4	*Megalops atlanticus*	Tarpon	0.01	0.01	313	.	.	.	.	.	.	.	.	.	0.96	.	.	.
1	*Membras martinica*	Rough silverside	.	.	.	.	.	.	0.08	0.06	65	.	.	.	.	.	3.08	.
2	*Membras martinica*	Rough silverside	.	.	.	0.00	0.05	402	0.29	0.26	78	.	.	.	.	0.25	3.85	.
3	*Membras martinica*	Rough silverside	.	.	.	.	.	.	0.20	0.08	456	.	.	.	.	.	3.07	.
4	*Membras martinica*	Rough silverside	.	.	.	.	.	.	0.27	0.13	544	.	.	.	.	.	2.21	.
1	*Menidia* sp.	Silverside	.	.	.	.	.	.	2.02	1.06	65	.	.	.	.	.	20.00	.
2	*Menidia* sp.	Silverside	.	.	.	0.01	0.05	402	1.27	0.41	178	.	.	.	.	0.75	15.4	.
3	*Menidia* sp.	Silverside	.	.	.	0.02	0.03	825	12.31	2.14	456	.	.	.	.	0.48	46.5	
4	*Menidia* sp.	Silverside	.	.	.	0.01	0.00	348	8.72	1.24	544	.	.	.	.	0.86	45.2	
1	*Menippe mercenaria*	Stone crab	0.07	0.05	69	.	.	.	.	.	.	.	.	.	4.35	.	.	.
2	*Menippe mercenaria*	Stone crab	0.03	0.01	262	0.01	0.01	402	.	.	.	0.02	0.01	349	2.67	0.75	.	1.15
3	*Menippe mercenaria*	Stone crab	0.20	0.04	298	0.01	0.00	825	0.00	0.05	456	0.05	0.02	171	12.1	0.48	0.22	4.09
4	*Menippe mercenaria*	Stone crab	0.20	0.05	313	0.01	0.01	348	0.00	0.04	544	0.01	0.10	103	9.58	1.15	0.18	0.97
2	*Menticirrhus americanus*	Southern kingfish	0.02	0.37	262	0.00	0.05	402	.	.	.	.	.	.	0.38	0.25	.	.
3	*Menticirrhus americanus*	Southern kingfish	0.00	0.06	298	.	.	.	0.00	0.05	456	.	.	.	0.34	.	0.22	.
4	*Menticirrhus americanus*	Southern kingfish	0.04	0.01	313	0.00	0.05	348	0.03	0.01	544	.	.	.	3.19	0.29	1.84	.
2	*Menticirrhus littoralis*	Gulf kingfish	0.00	0.06	262	.	.	.	.	.	.	.	.	.	0.38	.	.	.
4	*Menticirrhus littoralis*	Gulf kingfish	0.05	0.02	313	.	.	.	0.03	0.01	544	.	.	.	2.88	.	1.29	.
4	*Menticirrhus saxatilis*	Northern kingfish	0.01	0.17	313	.	.	.	.	.	.	.	.	.	0.32	.	.	.
2	*Mercenaria campechiensis*	Southern quahog	.	.	.	0.00	0.10	402	.	.	.	0.01	0.00	349	.	0.25	.	0.57
3	*Mercenaria campechiensis*	Southern quahog	.	.	.	.	.	.	.	.	.	0.01	0.08	171	.	.	.	0.58
4	*Mercenaria campechiensis texana*	Texas quahog	.	.	.	0.00	0.05	348	.	.	.	.	.	.	.	0.29	.	.
2	*Microgobius gulosus*	Clown goby	.	.	.	.	.	.	0.05	0.45	78	.	.	.	.	.	1.28	.
3	*Microgobius gulosus*	Clown goby	.	.	.	.	.	.	0.01	0.01	456	.	.	.	.	.	0.44	.
3	*Microgobius thalassinus*	Green goby	.	.	.	.	.	.	0.00	0.05	456	.	.	.	.	.	0.22	.
1	*Micropogonias undulatus*	Atlantic croaker	0.59	0.16	69	12.4	3.06	66	1.71	0.53	65	.	.	.	23.2	69.7	29.2	.
2	*Micropogonias undulatus*	Atlantic croaker	0.64	0.14	262	13.8	2.01	402	0.72	0.32	78	.	.	.	17.2	57.2	16.7	.
3	*Micropogonias undulatus*	Atlantic croaker	1.93	0.30	298	9.25	0.95	825	1.35	0.52	456	0.01	0.15	171	34.9	55.9	17.3	0.58
4	*Micropogonias undulatus*	Atlantic croaker	1.44	0.28	313	4.57	0.73	348	1.16	0.25	544	.	.	.	26.5	43.7	13.2	.
2	*Micropterus salmoides*	Largemouth bass	0.00	0.06	262	.	.	.	.	.	.	.	.	.	0.38	.	.	.
2	*Mnemiopsis mccradyi*	Phosphorus jelly	.	.	.	0.52	0.25	402	0.64	5.62	78	.	.	.	.	1.49	1.28	.
3	*Mnemiopsis mccradyi*	Phosphorus jelly	.	.	.	1.53	0.87	825	1.45	1.11	456	.	.	.	.	1.21	1.32	.
2	*Monacanthus hispidus*	Planehead filefish	.	.	.	0.00	0.05	402	.	.	.	.	.	.	.	0.25	.	.
3	*Monacanthus hispidus*	Planehead filefish	.	.	.	0.00	0.00	825	.	.	.	.	.	.	.	0.24	.	.
1	*Morone chrysops*	White bass	0.06	0.48	69	0.03	0.24	66	.	.	.	.	.	.	1.45	1.52	.	.

Table 7.6.6. (continued) Mean catch per sample for organisms caught in upper (area 1), mid (area 2), and lower (area 3) San Antonio Bay and Espiritu Santo Bay (area 4) by TPWD Coastal Fisheries Monitoring Program.

Area	Scientific Name	Common Name	Mean (M), standard error (SE), and sample size (n)												Percent of samples			
			Gillnet			Trawl			Bag seine			Oyster dredge			Gill-net	Trawl	Bag seine	Oyster dredge
			M	SE	n	M	SE	n	M	SE	n	M	SE	n				
2	*Morone chrysops*	White bass	0.01	0.01	262	.	.	.	.	.	.	.	.	.	0.76	.	.	.
4	*Morone chrysops*	White bass	.	.	.	.	.	.	0.01	0.01	544	.	.	.	.	.	0.37	.
2	*Morone saxatilis*	Striped bass	0.00	0.06	262	.	.	.	.	.	.	.	.	.	0.38	.	.	.
1	*Mugil cephalus*	Striped mullet	6.81	1.21	69	4.20	2.68	66	2.20	0.54	65	.	.	.	69.6	16.7	40	.
2	*Mugil cephalus*	Striped mullet	6.92	0.87	262	0.38	0.16	402	2.53	0.83	78	.	.	.	69.8	4.73	35.9	.
3	*Mugil cephalus*	Striped mullet	6.94	0.84	298	0.44	0.10	825	4.55	0.78	456	.	.	.	74.8	7.03	42.3	.
4	*Mugil cephalus*	Striped mullet	4.72	0.41	313	0.48	0.31	348	3.14	0.57	544	.	.	.	73.5	3.74	35.7	.
1	*Mugil curema*	White mullet	.	.	.	.	.	.	0.72	0.39	65	.	.	.	.	.	12.3	.
2	*Mugil curema*	White mullet	.	.	.	0.00	0.05	402	0.53	0.20	78	.	.	.	.	0.25	14.1	.
3	*Mugil curema*	White mullet	.	.	.	0.00	0.00	825	1.16	0.26	456	.	.	.	.	0.24	13.4	.
4	*Mugil curema*	White mullet	.	.	.	0.01	0.00	348	1.78	0.46	544	.	.	.	.	0.57	15.6	.
4	*Myrophis punctatus*	Speckled worm eel	.	.	.	0.00	0.05	348	.	.	.	.	.	.	.	0.29	.	.
1	*Negaprion brevirostris*	Lemon shark	0.03	0.02	69	.	.	.	.	.	.	.	.	.	2.90	.	.	.
2	*Negaprion brevirostris*	Lemon shark	0.00	0.06	262	.	.	.	.	.	.	.	.	.	0.38	.	.	.
3	*Negaprion brevirostris*	Lemon shark	0.01	0.01	298	.	.	.	.	.	.	.	.	.	1.01	.	.	.
4	*Negaprion brevirostris*	Lemon shark	0.04	0.01	313	.	.	.	.	.	.	.	.	.	3.51	.	.	.
3	*Nemopsis bachei*	Hydromedusa	.	.	.	0.67	0.61	825	.	.	.	.	.	.	.	0.24	.	.
1	*Neopanope texana*	Xanthid crab	.	.	.	.	.	.	0.02	0.12	65	.	.	.	.	.	1.54	.
2	*Neopanope texana*	Xanthid crab	.	.	.	0.01	0.00	402	.	.	.	0.00	0.05	349	.	0.75	.	0.29
3	*Neopanope texana*	Xanthid crab	.	.	.	0.00	0.03	825	0.00	0.05	456	.	.	.	.	0.12	0.22	.
2	*Oligoplites saurus*	Leatherjacket	.	.	.	.	.	.	0.05	0.45	78	.	.	.	.	.	1.28	.
3	*Oligoplites saurus*	Leatherjacket	.	.	.	.	.	.	0.04	0.02	456	.	.	.	.	.	1.54	.
4	*Oligoplites saurus*	Leatherjacket	.	.	.	.	.	.	0.06	0.02	544	.	.	.	.	.	3.68	.
4	*Ophichthus gomesi*	Shrimp eel	.	.	.	.	.	.	0.00	0.04	544	.	.	.	.	.	0.18	.
3	*Opisthonema oglinum*	Atlantic thread herring	.	.	.	0.02	0.02	825	.	.	.	.	.	.	.	0.36	.	.
4	*Opisthonema oglinum*	Atlantic thread herring	.	.	.	0.19	0.10	348	.	.	.	.	.	.	.	3.45	.	.
2	*Opsanus beta*	Gulf toadfish	.	.	.	0.02	0.01	402	.	.	.	0.00	0.05	349	.	1.49	.	0.29
3	*Opsanus beta*	Gulf toadfish	.	.	.	0.01	0.00	825	.	.	.	.	.	.	.	0.73	.	.
4	*Opsanus beta*	Gulf toadfish	.	.	.	0.01	0.01	348	0.01	0.00	544	.	.	.	.	1.15	0.37	.
4	Order Actiniaria	Order anemones	.	.	.	0.00	0.05	348	.	.	.	.	.	.	.	0.29	.	.
4	Order Hydroidea	Order hydroids	.	.	.	0.01	0.27	348	.	.	.	.	.	.	.	0.29	.	.
1	*Orthopristis chrysoptera*	Pigfish	0.07	0.06	69	.	.	.	.	.	.	.	.	.	2.90	.	.	.
2	*Orthopristis chrysoptera*	Pigfish	0.01	0.01	262	0.01	0.00	402	0.12	0.09	78	.	.	.	0.76	1.00	2.56	.
3	*Orthopristis chrysoptera*	Pigfish	0.04	0.01	298	0.02	0.01	825	0.26	0.11	456	.	.	.	3.36	1.09	4.17	.
4	*Orthopristis chrysoptera*	Pigfish	0.18	0.04	313	0.15	0.04	348	0.30	0.18	544	.	.	.	8.31	7.76	3.13	.
3	*Ovalipes guadulpensis*	Lady crab	.	.	.	0.00	0.03	825	.	.	.	.	.	.	.	0.12	.	.
3	*Pagurus longicarpus*	Long claw hermit crab	.	.	.	0.00	0.00	825	.	.	.	.	.	.	.	0.24	.	.
4	*Pagurus longicarpus*	Long claw hermit crab	.	.	.	.	.	.	0.01	0.00	544	.	.	.	.	.	0.55	.
2	*Pagurus pollicaris*	Big claw hermit crab	.	.	.	0.00	0.05	402	.	.	.	.	.	.	.	0.25	.	.
3	*Pagurus pollicaris*	Big claw hermit crab	.	.	.	0.00	0.03	825	0.00	0.05	456	.	.	.	.	0.12	0.22	.
4	*Pagurus pollicaris*	Big claw hermit crab	.	.	.	0.02	0.01	348	0.00	0.00	544	.	.	.	.	2.01	0.37	.
1	*Palaemonetes pugio*	Grass shrimp	.	.	.	.	.	.	0.18	1.48	65	.	.	.	.	.	1.54	.
3	*Palaemonetes pugio*	Grass shrimp	.	.	.	.	.	.	0.01	0.19	456	.	.	.	.	.	0.22	.

Table 7.6.6. (continued) Mean catch per sample for organisms caught in upper (area 1), mid (area 2), and lower (area 3) San Antonio Bay and Espiritu Santo Bay (area 4) by TPWD Coastal Fisheries Monitoring Program.

Area	Scientific Name	Common Name	Mean (M), standard error (SE), and sample size (n)												Percent of samples			
			Gillnet			Trawl			Bag seine			Oyster dredge			Gill-net	Trawl	Bag seine	Oyster dredge
			M	SE	n	M	SE	n	M	SE	n	M	SE	n				
1	*Palaenonetes* sp.	Grass shrimp-unidentified	.	.	.	.	.	.	2.14	0.75	65	.	.	.	.	.	33.8	.
2	*Palaenonetes* sp.	Grass shrimp-unidentified	.	.	.	0.02	0.01	402	11.4	6.48	78	.	.	.	.	1.00	32.1	.
3	*Palaenonetes* sp.	Grass shrimp-unidentified	.	.	.	0.04	0.02	825	3.08	0.88	456	.	.	.	.	1.33	22.4	.
4	*Palaenonetes* sp.	Grass shrimp-unidentified	.	.	.	0.01	0.11	348	4.86	1.07	544	.	.	.	.	0.29	25.4	.
1	*Paralichthys albigutta*	Gulf flounder	0.01	0.12	69	.	.	.	.	.	.	.	.	.	1.45	.	.	.
2	*Paralichthys albigutta*	Gulf flounder	0.02	0.02	262	.	.	.	.	.	.	.	.	.	0.76	.	.	.
3	*Paralichthys albigutta*	Gulf flounder	0.01	0.01	298	.	.	.	0.01	0.00	456	.	.	.	1.01	.	0.88	.
4	*Paralichthys albigutta*	Gulf flounder	0.10	0.02	313	.	.	.	0.00	0.00	544	.	.	.	8.63	.	0.37	.
1	*Paralichthys lethostigma*	Southern flounder	1.01	0.16	69	0.42	0.14	66	0.08	0.04	65	.	.	.	52.2	18.2	6.15	.
2	*Paralichthys lethostigma*	Southern flounder	0.80	0.08	262	0.12	0.02	402	0.03	0.22	78	.	.	.	38.9	9.45	1.28	.
3	*Paralichthys lethostigma*	Southern flounder	1.05	0.09	298	0.16	0.03	825	0.08	0.02	456	.	.	.	50	8.48	4.39	.
4	*Paralichthys lethostigma*	Southern flounder	1.57	0.15	313	0.02	0.01	348	0.08	0.02	544	.	.	.	55.3	2.01	4.60	.
1	*Penaeus aztecus*	Brown shrimp	.	.	.	11.2	6.15	66	2.48	0.80	65	.	.	.	.	24.2	29.2	.
2	*Penaeus aztecus*	Brown shrimp	.	.	.	16.1	2.15	402	8.55	3.41	78	.	.	.	.	52.2	35.9	.
3	*Penaeus aztecus*	Brown shrimp	.	.	.	13.2	1.93	825	7.63	0.99	456	.	.	.	.	40.8	38.2	.
4	*Penaeus aztecus*	Brown shrimp	.	.	.	4.30	0.93	348	12.6	1.63	544	.	.	.	.	31.9	43.4	.
2	*Penaeus duorarum*	Pink shrimp	.	.	.	0.31	0.10	402	0.04	0.03	78	.	.	.	.	7.96	2.56	.
3	*Penaeus duorarum*	Pink shrimp	.	.	.	0.76	0.15	825	0.61	0.48	456	.	.	.	.	9.21	3.07	.
4	*Penaeus duorarum*	Pink shrimp	.	.	.	0.42	0.10	348	0.21	0.09	544	.	.	.	.	10.3	2.76	.
1	*Penaeus setiferus*	White shrimp	.	.	.	3.74	1.51	66	4.28	1.83	65	.	.	.	.	37.9	30.8	.
2	*Penaeus setiferus*	White shrimp	0.01	0.12	262	7.20	1.11	402	1.40	0.42	78	.	.	.	0.38	47.3	20.5	.
3	*Penaeus setiferus*	White shrimp	.	.	.	3.57	0.35	825	6.80	1.90	456	.	.	.	.	37.1	24.1	.
4	*Penaeus setiferus*	White shrimp	0.00	0.06	313	0.82	0.15	348	9.66	1.76	544	.	.	.	0.32	17	32.9	.
1	*Peprilus alepidotus*	Harvestfish	0.01	0.12	69	.	.	.	.	.	.	.	.	.	1.45	.	.	.
2	*Peprilus alepidotus*	Harvestfish	0.01	0.01	262	.	.	.	.	.	.	.	.	.	0.76	.	.	.
3	*Peprilus alepidotus*	Harvestfish	0.00	0.06	298	0.01	0.01	825	.	.	.	.	.	.	0.34	0.97	.	.
4	*Peprilus alepidotus*	Harvestfish	0.03	0.01	313	0.03	0.01	348	.	.	.	.	.	.	2.24	2.01	.	.
2	*Peprilus burti*	Gulf butterfish	.	.	.	0.01	0.00	402	.	.	.	.	.	.	.	0.75	.	.
3	*Peprilus burti*	Gulf butterfish	.	.	.	0.12	0.02	825	.	.	.	.	.	.	.	5.21	.	.
4	*Peprilus burti*	Gulf butterfish	0.00	0.06	313	0.61	0.15	348	.	.	.	.	.	.	0.32	15.2	.	.
2	*Petrolisthes armatus*	Porcellanid crab	.	.	.	.	.	.	.	.	.	0.03	0.03	349	.	.	.	0.86
3	*Petrolisthes armatus*	Porcellanid crab	.	.	.	.	.	.	.	.	.	0.02	0.01	171	.	.	.	1.17
4	*Petrolisthes armatus*	Porcellanid crab	.	.	.	0.01	0.01	348	.	.	.	0.07	0.03	103	.	.	0.57	3.88
1	Phylum Ctenophora	Phylum ctenophores	.	.	.	0.23	0.13	66	0.08	0.62	65	.	.	.	.	4.55	1.54	.
2	Phylum Ctenophora	Phylum ctenophores	.	.	.	21.5	4.74	402	0.83	0.65	78	.	.	.	.	15.4	5.13	.
3	Phylum Ctenophora	Phylum ctenophores	.	.	.	8.08	2.01	825	1.21	1.10	456	.	.	.	.	6.67	2.41	.
4	Phylum Ctenophora	Phylum ctenophores	.	.	.	2.49	1.48	348	0.23	0.13	544	.	.	.	.	5.75	1.29	.
4	Phylum Porifera	Phylum sponges	.	.	.	0.27	0.06	348	.	.	.	0.16	0.08	103	.	6.32	.	3.88
3	*Podiceps caspicus*	Eared grebe	0.00	0.06	298	.	.	.	.	.	.	.	.	.	0.34	.	.	.
3	*Poecilia latipinna*	Sailfin molly	.	.	.	.	.	.	0.08	0.04	456	.	.	.	.	.	1.97	.
4	*Poecilia latipinna*	Sailfin molly	.	.	.	.	.	.	0.03	0.01	544	.	.	.	.	.	1.29	.
1	*Pogonias cromis*	Black drum	5.46	0.98	69	0.23	0.13	66	0.02	0.12	65	.	.	.	60.9	7.58	1.54	.
2	*Pogonias cromis*	Black drum	6.05	0.62	262	0.03	0.02	402	0.05	0.03	78	.	.	.	75.2	0.75	3.85	.

Table 7.6.6. (continued) Mean catch per sample for organisms caught in upper (area 1), mid (area 2), and lower (area 3) San Antonio Bay and Espiritu Santo Bay (area 4) by TPWD Coastal Fisheries Monitoring Program.

Area	Scientific Name	Common Name	Mean (M), standard error (SE), and sample size (n)												Percent of samples			
			Gillnet			Trawl			Bag seine			Oyster dredge			Gill-net	Trawl	Bag seine	Oyster dredge
			M	SE	n	M	SE	n	M	SE	n	M	SE	n				
3	*Pogonias cromis*	Black drum	5.59	0.46	298	0.01	0.01	825	0.21	0.10	456	.	.	.	76.5	0.73	5.92	.
4	*Pogonias cromis*	Black drum	5.29	0.44	313	0.01	0.01	348	0.05	0.02	544	.	.	.	72.2	1.15	2.57	.
2	*Polinices duplicatus*	Shark's eye	.	.	.	0.05	0.02	402	.	.	.	.	.	.	.	3.73	.	.
3	*Polinices duplicatus*	Shark's eye	.	.	.	0.01	0.00	825	0.00	0.05	456	.	.	.	.	1.09	0.22	.
4	*Polinices duplicatus*	Shark's eye	.	.	.	0.03	0.01	348	0.01	0.00	544	.	.	.	.	2.59	0.92	.
1	*Polydactylus octonemus*	Atlantic threadfin	.	.	.	0.05	0.03	66	.	.	.	.	.	.	.	4.55	.	.
2	*Polinices duplicatus*	Atlantic threadfin	.	.	.	0.09	0.03	402	0.01	0.11	78	.	.	.	.	3.48	1.28	.
3	*Polinices duplicatus*	Atlantic threadfin	0.00	0.06	298	0.14	0.03	825	0.03	0.02	456	.	.	.	0.34	5.45	0.66	.
4	*Polinices duplicatus*	Atlantic threadfin	0.01	0.01	313	0.08	0.02	348	0.04	0.02	544	.	.	.	0.96	5.46	1.47	.
2	*Pomatomus saltatrix*	Bluefish	0.00	0.06	262	.	.	.	.	.	.	.	.	.	0.38	.	.	.
3	*Pomatomus saltatrix*	Bluefish	0.00	0.06	298	.	.	.	.	.	.	.	.	.	0.34	.	.	.
4	*Pomatomus saltatrix*	Bluefish	0.08	0.04	313	.	.	.	0.00	0.04	544	.	.	.	2.56	.	0.18	.
2	*Pomoxis annularis*	White crappie	.	.	.	0.00	0.05	402	.	.	.	.	.	.	.	0.25	.	.
1	*Pomoxis nigromaculatus*	Black crappie	.	.	.	.	.	.	0.05	0.03	65	.	.	.	.	.	3.08	.
3	*Pomoxis nigromaculatus*	Black crappie	.	.	.	.	.	.	0.00	0.05	456	.	.	.	.	.	0.22	.
2	*Porcellana sayana*	Porcellanid crab	.	.	.	0.00	0.05	402	.	.	.	.	.	.	.	0.25	.	.
4	*Porcellana sayana*	Porcellanid crab	.	.	.	.	.	.	.	.	.	0.03	0.29	103	.	.	.	0.97
2	*Porichthys plectrodon*	Atlantic midshipman	.	.	.	0.01	0.01	402	.	.	.	.	.	.	.	0.75	.	.
3	*Porichthys plectrodon*	Atlantic midshipman	.	.	.	0.01	0.00	825	.	.	.	.	.	.	.	0.73	.	.
2	*Portunus gibbesii*	Purple crab	.	.	.	0.00	0.05	402	.	.	.	.	.	.	.	0.25	.	.
3	*Portunus gibbesii*	Purple crab	.	.	.	0.00	0.00	825	.	.	.	.	.	.	.	0.24	.	.
4	*Portunus gibbesii*	Purple crab	.	.	.	0.02	0.01	348	.	.	.	.	.	.	.	2.01	.	.
3	*Portunus spinicarpus*	Portunid crab	.	.	.	0.00	0.03	825	.	.	.	.	.	.	.	0.12	.	.
2	*Prionotus rubio*	Blackfin searobin	.	.	.	0.00	0.05	402	.	.	.	.	.	.	.	0.25	.	.
2	*Prionotus tribulus*	Bighead searobin	0.00	0.06	262	0.03	0.01	402	.	.	.	.	.	.	0.38	2.24	.	.
3	*Prionotus tribulus*	Bighead searobin	0.01	0.00	298	0.06	0.01	825	.	.	.	.	.	.	0.67	3.64	.	.
4	*Prionotus tribulus*	Bighead searobin	0.01	0.00	313	0.02	0.01	348	0.01	0.00	544	.	.	.	0.64	1.72	0.74	.
1	*Pylodictis olivaris*	Flathead catfish	0.03	0.02	69	.	.	.	.	.	.	.	.	.	2.90	.	.	.
4	*Rachycentron canadum*	Cobia	.	.	.	0.00	0.05	348	.	.	.	.	.	.	.	0.29	.	.
1	*Rangia cuneata*	Common rangia	.	.	.	0.27	0.16	66	0.02	0.12	65	7.71	4.86	7	.	7.58	1.54	42.9
2	*Rangia cuneata*	Common rangia	.	.	.	0.02	0.01	402	.	.	.	0.02	0.01	349	.	1.49	.	1.15
3	*Rangia cuneata*	Common rangia	.	.	.	0.02	0.01	825	.	.	.	0.05	0.03	171	.	0.73	.	1.75
1	*Rangia flexuosa*	Brown rangia	.	.	.	0.03	0.24	66	.	.	.	.	.	.	.	1.52	.	.
3	*Rangia flexuosa*	Brown rangia	.	.	.	0.00	0.03	825	.	.	.	.	.	.	.	0.12	.	.
1	*Rhinoptera bonasus*	Cownose ray	0.01	0.12	69	.	.	.	.	.	.	.	.	.	1.45	.	.	.
2	*Rhinoptera bonasus*	Cownose ray	0.00	0.06	262	.	.	.	.	.	.	.	.	.	0.38	.	.	.
3	*Rhinoptera bonasus*	Cownose ray	0.03	0.01	298	.	.	.	.	.	.	.	.	.	2.35	.	.	.
4	*Rhinoptera bonasus*	Cownose ray	0.08	0.03	313	.	.	.	.	.	.	.	.	.	4.15	.	.	.
3	*Rhizoprionodon terraenovae*	Atlantic sharpnose shark	0.01	0.01	298	.	.	.	.	.	.	.	.	.	0.67	.	.	.
4	*Rhizoprionodon terraenovae*	Atlantic sharpnose shark	0.11	0.05	313	.	.	.	.	.	.	.	.	.	2.88	.	.	.
4	*Saurida brasiliensis*	Largescale lizardfish	.	.	.	0.01	0.11	348	.	.	.	.	.	.	.	0.29	.	.
1	*Sciaenops ocellatus*	Red drum	13.1	2.02	69	.	.	.	0.57	0.17	65	.	.	.	79.7	.	24.6	.
2	*Sciaenops ocellatus*	Red drum	7.53	0.73	262	0.00	0.00	402	0.22	0.06	78	.	.	.	79	0.50	15.4	.

Table 7.6.6. (continued) Mean catch per sample for organisms caught in upper (area 1), mid (area 2), and lower (area 3) San Antonio Bay and Espiritu Santo Bay (area 4) by TPWD Coastal Fisheries Monitoring Program.

Area	Scientific Name	Common Name	Mean (M), standard error (SE), and sample size (n)												Percent of samples			
			Gillnet			Trawl			Bag seine			Oyster dredge			Gill-net	Trawl	Bag seine	Oyster dredge
			M	SE	n	M	SE	n	M	SE	n	M	SE	n				
3	*Sciaenops ocellatus*	Red drum	7.51	0.54	298	0.01	0.00	825	0.53	0.09	456	.	.	.	83.6	0.61	16.4	.
4	*Sciaenops ocellatus*	Red drum	10.7	0.62	313	0.00	0.05	348	0.67	0.12	544	.	.	.	91.4	0.29	19.5	.
4	*Scomberomorus cavalla*	King mackerel	.	.	.	.	.	.	0.00	0.04	544	.	.	.	.	.	0.18	.
2	*Scomberomorus maculatus*	Spanish mackerel	0.05	0.74	262	.	.	.	.	.	.	.	.	.	0.38	.	.	.
4	*Scomberomorus maculatus*	Spanish mackerel	0.01	0.01	313	.	.	.	0.00	0.04	544	.	.	.	0.96	.	0.18	.
2	*Selene setapinnis*	Atlantic moonfish	.	.	.	0.00	0.05	402	.	.	.	.	.	.	.	0.25	.	.
3	*Selene setapinnis*	Atlantic moonfish	.	.	.	0.00	0.03	825	.	.	.	.	.	.	.	0.12	.	.
4	*Selene setapinnis*	Atlantic moonfish	.	.	.	0.08	0.03	348	.	.	.	.	.	.	.	3.45	.	.
2	*Selene vomer*	Lookdown	.	.	.	0.00	0.05	402	.	.	.	.	.	.	.	0.25	.	.
3	*Selene vomer*	Lookdown	.	.	.	0.00	0.03	825	.	.	.	.	.	.	.	0.12	.	.
4	*Selene vomer*	Lookdown	0.02	0.01	313	0.03	0.01	348	.	.	.	.	.	.	0.64	2.30	.	.
4	*Sicyonia brevirostris*	Rock shrimp	.	.	.	0.00	0.05	348	.	.	.	.	.	.	.	0.29	.	.
4	*Sicyonia dorsalis*	Lesser rock shrimp	.	.	.	.	.	.	0.00	0.04	544	.	.	.	.	.	0.18	.
2	*Sphoeroides parvus*	Least puffer	.	.	.	0.06	0.02	402	0.09	0.04	78	.	.	.	.	3.73	7.69	.
3	*Sphoeroides parvus*	Least puffer	.	.	.	0.12	0.03	825	0.35	0.15	456	.	.	.	.	5.58	7.89	.
4	*Sphoeroides parvus*	Least puffer	.	.	.	0.05	0.02	348	0.09	0.05	544	.	.	.	.	4.31	2.94	.
4	*Sphoeroides spengleri*	Bandtail puffer	.	.	.	0.00	0.05	348	.	.	.	.	.	.	.	0.29	.	.
4	*Sphyrna lewini*	Scalloped hammerhead	.	.	.	0.00	0.05	348	.	.	.	.	.	.	.	0.29	.	.
1	*Sphyrna tiburo*	Bonnethead	0.01	0.12	69	.	.	.	.	.	.	.	.	.	1.45	.	.	.
2	*Sphyrna tiburo*	Bonnethead	0.81	0.81	262	.	.	.	.	.	.	.	.	.	0.76	.	.	.
3	*Sphyrna tiburo*	Bonnethead	0.12	0.05	298	.	.	.	.	.	.	.	.	.	4.03	.	.	.
4	*Sphyrna tiburo*	Bonnethead	0.48	0.13	313	.	.	.	.	.	.	.	.	.	14.4	.	.	.
2	*Squilla empusa*	Mantis shrimp	.	.	.	0.00	0.00	402	.	.	.	.	.	.	.	0.50	.	.
3	*Sphyrna tiburo*	Mantis shrimp	.	.	.	0.01	0.01	825	.	.	.	.	.	.	.	0.85	.	.
4	*Sphyrna tiburo*	Mantis shrimp	.	.	.	0.04	0.01	348	.	.	.	.	.	.	.	2.87	.	.
3	*Stellifer lanceolatus*	Star drum	.	.	.	0.03	0.02	825	.	.	.	.	.	.	.	0.48	.	.
4	*Stellifer lanceolatus*	Star drum	.	.	.	.	.	.	0.00	0.04	544	.	.	.	.	.	0.18	.
1	*Stomolophus meleagris*	Cabbagehead	.	.	.	.	.	.	0.31	0.28	65	.	.	.	.	.	3.08	.
2	*Stomolophus meleagris*	Cabbagehead	.	.	.	4.79	2.29	402	0.22	1.91	78	0.03	0.53	349	.	8.71	1.28	0.29
3	*Stomolophus meleagris*	Cabbagehead	0.00	0.06	298	2.03	0.45	825	0.18	0.15	456	.	.	.	0.34	12	0.88	.
4	*Stomolophus meleagris*	Cabbagehead	0.01	0.01	313	0.25	0.09	348	0.00	0.09	544	0.01	0.10	103	0.64	6.32	0.18	0.97
2	*Strongylura marina*	Atlantic needlefish	0.01	0.01	262	.	.	.	0.03	0.02	78	.	.	.	0.76	.	2.56	.
3	*Strongylura marina*	Atlantic needlefish	0.00	0.06	298	.	.	.	0.06	0.02	456	.	.	.	0.34	.	3.73	.
4	*Strongylura marina*	Atlantic needlefish	0.00	0.06	313	.	.	.	0.08	0.04	544	.	.	.	0.32	.	2.21	.
2	Suborder Reptantia	Suborder reptantia	.	.	.	0.00	0.05	402	.	.	.	.	.	.	.	0.25	.	.
4	Suborder Reptantia	Suborder reptantia	.	.	.	.	.	.	0.00	0.04	544	.	.	.	.	.	0.18	.
2	*Symphurus plagiusa*	Blackcheek tonguefish	.	.	.	0.01	0.01	402	.	.	.	.	.	.	.	1.00	.	.
3	*Symphurus plagiusa*	Blackcheek tonguefish	.	.	.	0.01	0.00	825	0.03	0.01	456	.	.	.	.	1.21	1.54	.
4	*Symphurus plagiusa*	Blackcheek tonguefish	.	.	.	0.01	0.01	348	0.03	0.01	544	.	.	.	.	0.86	2.39	.
1	*Syngnathus floridae*	Dusky pipefish	.	.	.	.	.	.	0.02	0.12	65	.	.	.	.	.	1.54	.
3	*Syngnathus floridae*	Dusky pipefish	.	.	.	.	.	.	0.00	0.05	456	.	.	.	.	.	0.22	.
4	*Syngnathus floridae*	Dusky pipefish	.	.	.	.	.	.	0.00	0.04	544	.	.	.	.	.	0.18	.
2	*Syngnathus louisianae*	Chain pipefish	.	.	.	0.00	0.10	402	.	.	.	.	.	.	.	0.25	.	.

Table 7.6.6. (concluded) Mean catch per sample for organisms caught in upper (area 1), mid (area 2), and lower (area 3) San Antonio Bay and Espiritu Santo Bay (area 4) by TPWD Coastal Fisheries Monitoring Program.

Area	Scientific Name	Common Name	Mean (M), standard error (SE), and sample size (n)												Percent of samples			
			Gillnet			Trawl			Bag seine			Oyster dredge			Gill-net	Trawl	Bag seine	Oyster dredge
			M	SE	n	M	SE	n	M	SE	n	M	SE	n				
3	*Syngnathus louisianae*	Chain pipefish	.	.	.	0.00	0.03	825	0.01	0.19	456	.	.	.	.	0.12	0.22	.
4	*Syngnathus louisianae*	Chain pipefish	.	.	.	.	.	.	0.01	0.01	544	.	.	.	.	.	0.92	.
2	*Syngnathus scovelli*	Gulf pipefish	.	.	.	0.02	0.01	402	0.06	0.03	78	0.01	0.11	349	.	1.00	5.13	0.29
3	*Syngnathus scovelli*	Gulf pipefish	.	.	.	0.01	0.00	825	0.06	0.02	456	.	.	.	.	0.61	4.82	.
4	*Syngnathus scovelli*	Gulf pipefish	.	.	.	.	.	.	0.09	0.02	544	.	.	.	.	.	5.51	.
3	*Synodus foetens*	Inshore lizardfish	.	.	.	0.01	0.00	825	0.02	0.01	456	.	.	.	.	1.09	1.32	.
4	*Synodus foetens*	Inshore lizardfish	.	.	.	0.01	0.01	348	0.01	0.00	544	.	.	.	.	1.15	0.92	.
2	*Thais haemastoma*	Florida rock shell	.	.	.	0.00	0.05	402	.	.	.	.	.	.	.	0.25	.	.
4	*Thais haemastoma*	Florida rock shell	.	.	.	0.01	0.01	348	.	.	.	0.01	0.10	103	.	1.15	.	0.97
3	*Tozeuma carolinense*	Arrow shrimp	.	.	.	.	.	.	0.00	0.05	456	.	.	.	.	.	0.22	.
2	*Trachinotus carolinus*	Florida pompano	0.02	0.02	262	.	.	.	.	.	.	.	.	.	0.76	.	.	.
3	*Trachinotus carolinus*	Florida pompano	0.03	0.01	298	.	.	.	.	.	.	.	.	.	2.01	.	.	.
4	*Trachinotus carolinus*	Florida pompano	0.08	0.02	313	.	.	.	0.02	0.01	544	.	.	.	5.43	.	1.10	.
4	*Trachinotus falcatus*	Permit	0.02	0.01	313	.	.	.	.	.	.	.	.	.	0.96	.	.	.
3	*Trachinotus goodei*	Palometa	0.00	0.06	298	.	.	.	.	.	.	.	.	.	0.34	.	.	.
3	*Trachypenaeus similis*	Yellow rough-necked shrimp	.	.	.	0.00	0.03	825	.	.	.	.	.	.	.	0.12	.	.
3	*Trachypenaeus* sp.	Trachypeneid	.	.	.	0.00	0.03	825	.	.	.	.	.	.	.	0.12	.	.
2	*Trichiurus lepturus*	Atlantic cutlassfish	.	.	.	0.00	0.05	402	.	.	.	.	.	.	.	0.25	.	.
3	*Trichiurus lepturus*	Atlantic cutlassfish	.	.	.	0.01	0.00	825	.	.	.	.	.	.	.	0.97	.	.
4	*Trichiurus lepturus*	Atlantic cutlassfish	.	.	.	0.08	0.02	348	.	.	.	.	.	.	.	4.60	.	.
1	*Trinectes maculatus*	Hogchoker	0.04	0.02	69	0.11	0.07	66	.	.	.	.	.	.	4.35	4.55	.	.
2	*Trinectes maculatus*	Hogchoker	0.03	0.01	262	0.11	0.02	402	0.06	0.04	78	.	.	.	2.29	6.97	3.85	.
3	*Trinectes maculatus*	Hogchoker	0.06	0.01	298	0.06	0.01	825	0.01	0.01	456	.	.	.	6.04	4.36	0.66	.
4	*Trinectes maculatus*	Hogchoker	0.04	0.01	313	0.03	0.01	348	0.01	0.01	544	.	.	.	2.88	2.30	1.10	.
1	*Tursiops truncatus*	Atlantic bottlenose dolphin	0.01	0.12	69	.	.	.	.	.	.	.	.	.	1.45	.	.	.
3	*Urophycis floridana*	Southern hake	.	.	.	0.01	0.00	825	.	.	.	.	.	.	.	1.21	.	.
4	*Urophycis floridana*	Southern hake	.	.	.	0.03	0.01	348	.	.	.	.	.	.	.	2.01	.	.

Table 7.6.7. Species and life stages which occurred in at least 20% of the samples within a gear type for each of the areas of the Guadalupe Estuary. Life stages are: A-adult, S-subadult, and J-juvenile; asterisks (*) indicate areas of greatest abundance according to a criterion based on mean catches and standard errors[a]; original data compilation is given in Table 7.6.6.

Common name	Area I			Area II			Area III			Area IV			Salinity preference[b]	
	A	S	J	A	S	J	A	S	J	A	S	J	Stage	Range
Alligator gar	A*	S*		A*	S*		A	S					AS	(0 to 20)^{1}
Atlantic croaker	A	S*	J*		S*		A*	S	J*	A*	S	J*	A	(15 to 35)^{1}
													S	(10 to 20)^{1}
													J	(0 to 35)^{1}
													R	(15 to 20)^{1}
Bay anchovy	A	S		A	S		A*	S*	J*	A	S	J*	AS	(15 to 25)^{1}
													J	(15 to 30)^{1}
Black drum	A*	S*		A*	S*		A*	S*		A*	S*		AS	(0 to 35)^{1}
													A	(20 to 30)^{2}
Blue catfish	A*												A	(0 to 10)^{1}
Blue crab	A*	S*	J	A*	S*	J	A*	S	J	A*	S	J*	A	(0 to 35)^{1}
													S	(0 to 20)^{1}
													J	(25 to 35)^{1}
Brief squid										A*			A	(25 to 35)^{1}
Brown shrimp		S*	J		S*	J		S*	J		S	J	S	(0 to 25)^{1}
Bull shark	A	S		A	S		A*	S*		A*	S*		AS	(15 to 35)^{2}
Eastern oyster	A			A*			A*			A			A	(10 to 25)^{1,2}
													R	(15 to 25)^{2}
													P	(<10)^{2}
Gafftopsail catfish	A			A*			A			A*			A	(10 to 35)^{1}
Gizzard shad	A			A*			A			A			A	(10 to 20)^{1}
Grass shrimp	A			A			A*			A*			A	(15 to 35)^{1}
Gulf killifish	A*	S*	J*	A*	S*	J*	A*	S*	J*	A	S	J	ASJ	(0 to 25)^{1}
Gulf menhaden		S	J*	A*	S*	J*	A*	S*	J*	A*	S	J*	A	(15 to 35)^{1}
													S	(10 to 25)^{1}
													J	(0 to 35)^{1}
Silversides	A	S	J				A*	S*	J*	A	S	J	A	(15 to 25)^{1}
Ladyfish							A*			A*			A	(15 to 35)^{1}
Longnose gar	A*												A	(0 to 10)^{1}
Longnose killifish				A	S	J	A*	S*	J*	A*	S*	J*	ASJ	(15 to 35)^{1}
Pinfish				A	S		A	S		A*	S*	J*	ASJ	(25 to 35)^{1}
Red drum		S*	J*		S			S	J*		S*	J*	SJ	(0 to 35)^{1}
Sea catfish	A	S	J*	A	S	J*	A*	S*	J*	A*	S*	J*	AS	(15 to 25)^{1}
													J	(0 to 35)^{1}
Sheepshead	A			A			A			A*			A	(25 to 35)^{1}
Sheepshead minnow	A*	S*	J*	A*	S*	J*	A*	S*	J*	A*	S*	J*	ASJ	(0 to 35)^{1}
Southern flounder	A			A			A			A*			A	(25 to 35)^{1}
Spot					S	J	A	S*	J*	A*	S*	J	A	(25 to 35)^{1}
													S	(15 to 35)^{1}
													J	(15 to 25)^{1}
Spotted gar	A*			A*									A	(0 to 20)^{1}
Spotted seatrout	A			A			A			A*			A	(20 to 35)^{1,2}
													R	(25 to 35)^{2}
Striped mullet	A		J	A		J	A		J*	A		J*	A	(0 to 35)^{1}
													J	(15 to 35)^{1}
White shrimp		S	J		S*	J		S	J*			J*	S	(10 to 20)^{1,2}
													J	(15 to 25)^{1,2}

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[a] The letter representing a life stage indicates that the species and life stage occurred in at least 20% of the samples for a particular gear; the areas of greatest abundance noted by asterisks (*) include the area with the highest mean catch rate (X_j) and other areas where the difference between the mean catch rate (X_i) and the highest mean catch rate was less than or equal to the sum of the respective standard errors ($SE_j + SE_i$) of both areas [i.e., $(X_j - X_i) \leq (SE_j + SE_i)$].

[b] The superscript indicates source of information: 1—indicates salinity range associated with the occurrence and abundance information of Table 7.6.6 and 2—indicates salinity information is from Table 6.7.2. Information beginning with R indicates reproductive information based on published information discussed in Chapter 6; information about survival limits for natural predators or disease organism (P) is based on published information discussed in Chapter 6.

vegetation (Childress et al. 1975). At Saluria Bayou, shoalgrass, widgeongrass, and clovergrass (*Halophila engelmanni*) were the dominant submerged vegetation, while sea blite (*Suaeda maritima*) and black mangrove (*Avicennia germinans*) dominated the emergent vegetation (Childress et al. 1975).

Pulich (1990) reported large expanses of seagrasses were present in Shoalwater Bay and Pringle Lake during his 1989-90 survey. In addition, saltwater marshes were common in Pringle Lake, Shoalwater Bay, and the east end of Espiritu Santo Bay. Pulich also reported smooth cordgrass, perennial glasswort, saltwort, and salt-flat grass from the Saluria Bayou area and marsh aster, saltgrass, Carolina wolfberry, sea oxeye daisy, marsh elder (*Iva frutescens*), and saltmeadow cordgrass (*Spartina patens*) were also common in the high-marsh zones of this area. These high-marsh zones graded into upland communities typically containing Gulf and saltmeadow cordgrass and seacoast bluestem (*Shizachyrium* sp.) (Table 7.6.1).

Benthos in polyhaline-to-euhaline Area IV. White et al. (1989) conducted the only significant investigation of the benthic community in this area. They reported characteristic species for different ecotypes (benthic assemblages including open bay center, bay margin, inlet-influenced, river-influenced, grassflat, and oyster reef). The site at the northeast end of Area IV was described as primarily inlet influenced (Saluria Bayou) and was characterized by polychaetes and mollusks. The central and southwest part of the area was characterized as mainly an open-bay-center assemblage dominated by polychaetes. In this bay, the seagrass meadows contained the highest average number of both species and individuals per station, while the inlet-influenced sites were second, and non-vegetated open bay sites were third.

White et al. (1989) did not report number (abundance) of each benthic species collected at each station; therefore, it was not possible to compare dominant species found by them with other studies in the estuary. However, species which were mentioned as generally abundant were identified and included the polychaetes *S. benedicti*, *M. californiensis*, and *Clymenella torquata* and the crustaceans *Ampelisca abdita*, *Grandidierella bonnieroides*, *Cerapus tubularis*, *Corophium acherusicum*, and *Lepidactylus* sp. The polychaetes *M. californiensis* and *S. benedicti* were also common in the other three areas.

Benthic species diversity in this area varied with regard to location in the bay. In the northeast portion of Area IV, species diversity was high with one station having the highest diversity of any in the estuary. Species diversity in the central and southwest portion was generally low; it was similar to that found in areas II and III and Hynes Bay.

In general, the number of species per site was higher in this area than in any of the other three areas. This apparent positive relationship between salinity and number of taxa per site was also noted by other investigators in upper, mid-, and lower San Antonio Bay (Harper 1973; Matthews et al. 1974; Montagna and Kalke 1992). Consequently, Area IV had the largest characteristic benthic assemblage, containing 37 species. Although species diversity is greater in high salinity areas, the productivity may be greater in oligohaline or mesohaline areas (Harper 1973; Matthews et al. 1974). This could provide an explanation of why there are higher biomasses of finfish-shellfish species in mesohaline areas (Zimmerman et al. 1990b; Section 6.3).

Finfish-shellfish in polyhaline-to-euhaline Area IV. Twenty-six finfish-shellfish species were identified as characteristic of this area. The species with the highest catch rates for a portion of their life included juvenile sheepshead, juvenile blue crab, pinfish, juvenile brown and white shrimp, and adult spotted seatrout. Species with high catch rates for a portion of their life in Area IV and at least one other area included adult sheepshead, all stages of gafftopsail catfish, adult Gulf menhaden, adult ladyfish, juvenile spotted seatrout, longnose killifish, adult and subadult spot, silversides, adult Atlantic croaker, and juvenile red drum.

General Salinity Requirements for Each Area

General considerations. Several environmental variables that depend on freshwater inflow may affect species occurrence patterns within an estuary. These include nutrient and sediment load, salinity, temperature, delta inundation frequency, and current. Of these, salinity is the most often studied in surveys on estuaries. In addition, there are more laboratory data on animals dealing with salinity and temperature than other variables. For these reasons, the following discussion will concentrate on salinity requirements of the Guadalupe Estuary biotic community. From nutrient budget and minimum loading information in sections 7.3 and 7.4, it appears that providing for the correct salinity conditions also provides the necessary loading of nutrients. Information on requirements of other variables will be introduced where it exists and is appropriate.

To satisfy the requirements of the biotic community, salinity limits must be identified and the timing of historical annual cycles must be maintained since the organisms have adapted to these general conditions. In the Guadalupe Estuary, these cycles have been determined largely by riverine inflow (Chapter 4 and Section 7.1). Briefly, intermediate inflows occur during January-March, July, and November-December; high inflows occur during May-June and September-October; and the lowest inflows occur during August. The historical high and low inflow periods typically take place just before major salinity shifts in this estuary.

Analytical methods for salinity requirements. To identify the biotic community salinity requirements for each region, the salinity requirements for each inhabitant of the region should be considered. However, they are known for only a few species. For the vast majority of species, these requirements can only be extrapolated from the data indicating environmental conditions under which they have been most abundant or most frequently found. Because of the dynamic nature of estuarine environments, it is likely that data contained in some published surveys were collected under conditions which represented transitional states. Thus, extrapolation of salinity preference information from these surveys is not necessarily precise.

For the important estuarine plant species occurring along the Texas coast, Pulich (1990) compiled and reviewed optimum (preferred) and survival range salinities and occurrence elevations (relative to mean sea level). Much of this information is contained in Table 5.2.2 and was used to identify salinity requirements for a portion of the vegetative communities in each area (Table 7.6.1).

For the benthos and finfish-shellfish communities, preferred salinity limits of individual species were extracted from the literature, Table 6.7.3, or TPWD data (Table 7.6.6). As discussed previously, much of the benthos data contained in the surveys for this estuary was collected during high inflow conditions and is probably biased toward lower-than-normal salinities. The finfish-shellfish community data does not have this problem since it was collected continuously for 12 years and has included low and high inflow periods.

Vegetation requirements in Area I. The salinity regime of this region is very dynamic, which results in a variable biotic community harboring both fresh and brackish-water species. Section 7.1 describes in detail the salinity regime variations of the Guadalupe Estuary under low (less than 25th percentile), intermediate (25th to 75th percentile) and high inflow (greater than 75th percentile) regimes (Figure 7.1.7).

Should a long-term increase in salinity occur in this area, the response by plant species in the community will likely be dramatic. Annuals may be affected immediately, but some long-lived plants will be slow to exhibit a population response. According to salinity tolerance information for freshwater species in the characteristic vegetation assemblage (Table 7.6.1), salinities of more than 5‰ will have a detrimental impact on the submerged water nymph, the emergent cattail and arrowhead, and the floating water hyacinth. Salinities of more than 10‰ would stress at least three more species: bulrush, common reed, and saltmeadow cordgrass. Conversely, these higher salinities would favor the shift to more characteristic saltwater flora. Widgeongrass would replace water nymph, sea oxeye daisy, and saltgrass would replace common reed and saltmeadow cordgrass, and smooth cordgrass would replace cattails and bulrushes. In summary, the freshwater community would be changed to a saltwater community.

In addition to salinity, two other riverine inflow-related variables also affect this community: sediment deposition and inundation frequency (Pulich 1990). Species on the delta which require periodic inundation by flood waters may die due to desiccation from decreased inundation frequency (Pulich 1990). Over the 47-year period 1941 through 1987, the upper delta was inundated, on the average, five to seven times per year (Section 7.5). Reduced inundation frequency may lead to reduced sedimentation on the delta and hasten the development of a more haline-type community, by increasing soil salinity.

Benthos requirements in Area I. Salinity tolerances of benthic animals were not so well documented and must be inferred from salinities in which they have been found. Three taxa were identified as being restricted to this region: *H. florida*, *P. ocularis*, and chironomid larvae. Because this entire region (as well as much of the mid-bay region) was oligohaline during a majority of the published surveys, it is likely that these three organisms are restricted to a very low salinity environment (less than 5‰). Inflows producing salinities of 10‰ over most of the region may severely restrict their occurrence or eliminate these organisms from the estuary. The remainder of the benthos in this region may be spatially restricted but not likely eliminated under increased salinities. The standing crop (total number of organisms/sample) of benthic animals has been shown to be negatively related to salinity in this estuary (Harper 1973; Matthews et al. 1974; Montagna and Kalke 1992). Therefore, a salinity increase may result in a reduction of the total number or biomass of benthic animals in this area.

Finfish-shellfish requirements in Area I. While a number of salinity studies are available for white shrimp, oysters, and crabs (Section 6.2), the salinity tolerance for many of the finfish-shellfish species must be inferred from salinities in which they were collected. The density of several life stages of finfish-shellfish species were high in this area (Table 7.6.7). The freshwater fishes, blue catfish, and gars (alligator, longnose, and spotted) were common. These fishes could be restricted to the Guadalupe River if salinities greater than 10‰ prevailed in Mission Lake, Guadalupe, and upper San Antonio bays. In addition, several species which concentrate in this region during a portion of their life may require an oligohaline or low mesohaline environment to complete their life cycles. Salinities up to 15‰ for periods of less than six months are not likely to eliminate any of these species, with one exception. These conditions may be less than optimal for adult male blue crabs, which inhabit and mate in the oligohaline areas (figures 6.2.1, 6.2.2, 6.2.3, and 6.2.4).

Prolonged salinities as high as 10 to 15‰ in this region will likely have a negative impact on at least eight plant, three benthos, and five finfish-shellfish species as well as many terrestrial species that use the wetlands in this area (TPWD 1991).

Special considerations. A permanent 5‰ increase in salinity in the Mission Lake and Guadalupe Bay waters could result in an increase in the production of many common aquatic estuarine fish and shellfish in Area I. However, there will also be a major change in the emergent marsh vegetation and terrestrial wildlife associated with the existing freshwater marshes. Changing freshwater or oligohaline water marsh habitat to mesohaline marsh habitat will result in the loss of most, if not all, amphibians, a reduction in biodiversity of reptiles and mammals, and a change in the species composition of the birds that use the area (Chabreck 1988).

The Guadalupe Delta Wildlife Management Area, operated by the TPWD, is located on the upper end of Hynes Bay and the northern shore of Mission Lake (TPWD 1991). Vegetation that could be affected by saltwater intrusion if salinities in Area I increased includes switch grass, seashore paspalum, longtom, sandspike sedge, carex, big and little duckweed, mosquito fern, and sandbar willow. Other animals, in addition to those already mentioned, that use the wildlife management area and might be adversely affected by increasing salinity include the American alligator, mottled duck, wood duck, and migratory songbirds that use the site as a staging or fallout area. There may be other species that could be affected which are not known; TPWD biologists are currently working on an inventory of all the terrestrial plants and wildlife inhabiting this area (D. Mabie, TPWD; pers. comm.).

Recovery and recommended bounds for Area I. Recovery time for the biotic community in Area I may vary from as little as a few days or weeks, in response to short-term (less than one month) salinity perturbations, to a year or longer after long-term periods (several months) of low inflow, which result in salinity increases of more than 10‰. Salinities greater than 15‰ occurring for extended periods will restrict or eliminate the use of this area by many current inhabitants and certainly reduce their overall abundance. Some plant species may not reestablish themselves until the following year. At least some of the benthos species have been shown to respond within one month to flood conditions (Harper 1973; Matthews et al. 1974). Most finfish-shellfish populations will likely recover to pre-event status within days or weeks after short-term events, while some long-lived species could require a year or more for populations to recover.

A review of the available information indicates that the salinities in Mission Lake and Guadalupe Bay should remain below 5‰ for 70 to 90% of the time to keep the vegetation on the Guadalupe Delta and the Alligator Slide area from changing from a fresh-brackish community to a salt marsh community.

Vegetation requirements in Area II. The large expanse of Area II ranges from oligohaline to primarily mesohaline depending on freshwater inflows. Under intermediate inflow conditions, it is mesohaline, ranging from just less than 10‰ near McDowell Point on the west shore to 15‰ near the Aransas National Wildlife Refuge and Welder Flats.

Most of the plant species in this region withstand salinities from 10 to 30‰, with only a population density change at extreme salinities. Exceptions are water nymph, Gulf cordgrass, common reed, and salt-meadow grass; published information indicates these species occur in salinities of less than 10‰. The latter species may inhabit small sites having oligohaline or near-oligohaline salinities, or they may tolerate higher salinities than have been documented.

The lower salinity bounds for this plant community which would not result in species elimination appears to be about 10‰. The upper bounds are more difficult to define because of the characteristic mesohaline nature of the region. Some of the oligohaline to low mesohaline species (in particular, common reed) may be eliminated from the region if salinities go permanently above 20‰ in Area II. For the remainder of the characteristic plant assemblage, salinities would need to exceed 30‰ for an extended period (at least several months) to cause significant change. Variations within this 10 to 30‰ range would probably only cause fluctuations in population densities.

Benthos requirements in Area II. From historical salinity levels and benthic species occurrence patterns, the low and high salinity bounds are 10 to 20‰. At least three species from the characteristic benthic assemblage occur in this region and in more saline areas, but not in the upper bay region where lower salinities prevail. These three species may be eliminated from Area II if long-term salinity levels declined by 5‰. At least five taxa from the characteristic benthos assemblage list occur in this and the upper area, but are absent from the higher salinity areas. These species could be eliminated from this area by a permanent salinity increase of 5‰.

Finfish-shellfish requirements of Area II. The finfish-shellfish community of this region is typical of an estuarine environment. Spotted gar has the lowest tolerance for marine salinities, tolerating salinities that are generally less than 15‰ (from TPWD data), and would likely lose the use of this area entirely if salinities increased 5 to 10‰ above historical levels. Other species which appear to be limited by the higher salinities of lower regions are juveniles of Gulf menhaden and Atlantic croaker, adults of alligator gar and gizzard shad, subadult blue crab, and all sizes of grass shrimp, sheepshead minnow, and white shrimp. An increase in salinity to levels common in the lower areas (15 to 25‰) may restrict the use of this area by these species. Eastern oysters, which occur in large reefs in this area, are negatively affected by salinities of less than 5 or greater than 30‰, if these conditions persist for more than a few weeks. Spat set is suboptimal at salinities outside the 15 to 25‰ range. The remainder of the finfish-shellfish community tolerates a wide range of salinities and can probably withstand levels from 10 to 25‰ without significant changes.

Recovery and recommended bounds for Area II. Complete recovery of the biotic assemblage of this region from salinities outside the normal bounds will vary among the three communities discussed. The plant communities may return to normal within a few months if normal salinities are restored in winter or spring. If normal conditions return during late fall to winter, complete recovery may not occur until the following year.

Recovery of benthic populations to pre-event status can be inferred from responses to flood conditions experienced during published surveys. Those data indicate a rapid response to environmental changes. The benthos may recover within a few months after return to normal salinities, if reproductive conditions are prevalent.

Harper (1973) and Matthews et al. (1974) demonstrated that the benthic community of San Antonio Bay responded quickly (approximately one month) to a dramatic inflow increase by increasing the density of a few species (*H. florida*, *T. sphinctostoma*, *R. cuneata*). With the apparent ability of many of the macroinvertebrates in this region to adapt to altered salinity regimes, it is unlikely that temporary changes in these regimes will result in permanent population changes. However, permanent changes to inflow quantity or patterns will produce permanent community changes. It may eliminate some species which exist only in this and the upper region such as *Capitella capitata*, *R. cuneata*, *R. flexuosa*, and some oligochaetes. But several other species characteristic of higher salinity parts of the estuary might establish themselves as abundant, permanent residents of Area II including several species of polychaetes, amphipods, cumaceans, isopods, taraids, and bivalves.

The floods of 1987 provided an opportunity to observe the effects of low salinities on eastern oysters and the time for their recovery from such an event. The record inflows of June 1987 killed almost all of the oysters in areas I, II, and III of the estuary as salinities fell to less than 5‰ for several months. The only live oysters found from July 1987 to March 1988 were located in Ayres Bay near Cedar Bayou and in the extreme east end of Espiritu Santo Bay near Saluria and Big bayous. Both sites (one in Area III and one in Area IV) received water from the Gulf and remained in a mesohaline-polyhaline condition throughout the flood. TPWD surveys indicated that oyster communities had recovered by 1989. Interestingly, oyster landings in the adjacent Mission-Aransas Estuary increased in 1988 and 1989 (Johns 1990), probably as a response to the 1987 inflows from the Guadalupe River.

The remainder of the finfish-shellfish community is not noticeably affected by high inflow conditions, but may be negatively affected by conditions resulting in high salinities. If year-classes are not lost as a result of high salinities, the mobile finfish-shellfish community may recover from high salinity events within days or weeks after salinities return to normal. If year-classes are lost, those species may require a year or longer to recover.

Salinities in this area should remain between 10 and 20‰ for approximately 60 to 80% of the time to maintain high productivity of blue crab, white shrimp, Gulf menhaden, and brown shrimp. Higher salinities (above 20‰) will favor brown shrimp and oysters, and lower salinities (about 10‰) will favor blue crabs and white shrimp (Table 7.6.7).

Vegetation requirements in Area III. During intermediate inflow conditions, this region is a mesohaline-polyhaline environment ranging from 15‰ at the Intracoastal Waterway, to 15 to 20‰ in Mesquite Bay, to 20 to 25‰ at South Pass. During high inflow periods, the salinity decreases about 5‰, and during low inflow conditions, salinity will increase about 5 to 7‰.

The plant community requirements of this region are similar to the mid-bay region with the exception that there does not appear to be any species characteristic of an oligohaline environment. Most of the species occurring here can tolerate a salinity range of 10 to 35‰ with changes only in population densities. Long-term salinity changes outside these bounds will probably show greater changes in the plant and animal abundances comprising the community in this area.

Benthos requirements in Area III. The benthic community will likely exhibit larger changes in response to a smaller salinity change than the plant community. Long-term salinities beyond the 15 to 25‰ range may decrease the density of some populations or even eliminate some species.

Finfish-shellfish requirements in Area III. The salinity requirements of the finfish-shellfish community of this region are similar to those of the mid-bay region. The species which use this region in disproportionately high numbers are very mobile with the exception of eastern oysters. The species within the finfish-shellfish assemblage prefer salinities between 10 and 30‰ (Table 7.6.7). A few of the species can tolerate salinities as low as 5‰ and a few as high as 35‰, but extended times outside this range will significantly reduce or eliminate them from using this area.

Recovery and recommended bounds for Area III. Recovery times will be similar in this area to those of Area II. Oysters and plants will require the longest recovery time, ranging from several months, depending on timing, extent, and duration of the event, to a year or more. The finfish-shellfish community, with the exception of oysters, will likely require only a few weeks, at most, to regain its pre-event status.

The salinity regime in this area should be between 15 and 30‰ for 60 to 80% of the time. Significant increases in salinity to more than 25‰ will likely reduce production of blue crabs, white shrimp, brown shrimp, and Gulf menhaden.

Vegetation requirements in Area IV. This area is comprised of Espiritu Santo Bay and is typically polyhaline (18 to 30‰), with the southwest area ranging from 20 to 25‰, and the northeast area more than 25‰.

Only under extreme high or low river inflow conditions will salinities be altered enough to produce detectable changes in the plant community. Salinity limits for this community are roughly 10 to 35‰. Several species occurring here have published optimal upper salinity limits of less than 25‰ even though salinities frequently reach 30‰ (Table 7.6.1). These species tolerate salinities of more than 30‰ for short periods, although prolonged exposures may stress the plants beyond their ability to adjust. Salinities outside the 10 to 35‰ range will cause a density change in some of the species.

Benthos requirements for Area IV. The benthic assemblage of Area IV is strongly influenced by Gulf exchange through Pass Cavallo and contains many species not reported from the other regions. Presumably, these are high-salinity tolerant species. At least one of these, *Ampelisca abdita*, also occurs on the continental shelf on the Gulf side of Matagorda Island. This indicates that the salinity tolerance limits of this community are roughly 20 to 35‰, with some species capable of withstanding lower and some higher salinities. Extended periods beyond either of these extremes will probably eliminate some of these species from the area.

Finfish-shellfish requirements for Area IV. Similarly, the finfish-shellfish assemblage in this area contains species characteristic of both estuarine and marine environments. Salinities of 20 to 30‰ are required to provide for the needs of most of the species comprising this community. Extended periods beyond these extremes will likely restrict the use of the area for some species.

Recovery and recommended bounds for Area IV. For the finfish-shellfish community, recovery from major perturbations will occur within weeks or months. The plant community will require months to recover if only densities are affected, and a year or more if species are eliminated. Benthos recovery will be more rapid than that of the plant community if only densities are changed. Otherwise, their recovery times will be similar.

The salinity regime for this region should remain between 20 to 30‰ for more than 80% of the time. In general, this area of the estuary should be fairly stable, and the communities will be very resistant to short-term changes caused by salinities higher or lower than these limits.

7.7 CONCLUSIONS

Chapter 7 has taken a close look at the Guadalupe Estuary to develop the specific information needed to apply the analytical tools that are presented in Chapter 8. With this information, it is possible to estimate the freshwater inflow required to maintain the ecological health and productivity that has been historically characteristic of this estuary.

Hydrology and salinity patterns. Section 7.1 reviewed the historical inflows by month and showed general salinity patterns within the estuary based on low, intermediate, and high inflow conditions. Monthly median inflows ranged from the lowest amounts during July (95,200 acre-ft) to the highest during May (222,627 acre-ft) in any given year.

Nutrient loading requirement for nitrogen. A minimum nitrogen loading requirement was developed in Section 7.4 using dissolved nutrient estimates from TWC and TWDB monitoring data (Section 7.2) and a nutrient budget (Section 7.3). Based on this analysis, an annual combined freshwater inflow of 286,000 acre-ft is required to replace all nitrogen lost to the system each year. Since many assumptions were made in making the budget, the estimate should be viewed as an absolute minimum. The estuarine system is driven by available nutrients and failure to deliver these nutrients in sufficient quantities could result in long-term declines similar to the one documented for shrimp and sardines off the mouth of the Nile River (Wadie and Razek 1985). Because this estimate is so important and several assumptions involving key estimates for intermediate values were made, the estimate should be revised and refined as new information becomes available.

Sediment loading requirement. In Section 7.5, it was estimated that 355,235 acre-ft of freshwater inflow was required to deliver 132 acre-ft of sediment per year to maintain the bathymetry of Mission Lake and accretion on the Traylor Cut subdelta.

Salinity requirements of characteristic species and life stages in four areas of the Guadalupe Estuary. Section 7.6 reviewed the distribution of marsh vegetation, benthic animals, and fish and shellfish that occur within four large areas of the Guadalupe Estuary having different salinity and nutrient conditions. Information for the comparisons came from previously published studies, recent studies done as a result of the 1985 legislation, and the TPWD Coastal Fisheries Monitoring Program. These studies showed that the plant and animal species composition varied among the areas based on salinity tolerances and the general salinity regime in a bay area. Benthic studies may be biased since they were done at times of high inflows; some species were unique to the fresh-to-oligohaline or oligohaline-to-mesohaline areas (i.e., *R. cuneata*, *R. flexuosa*, and chironomid larvae).

Most common estuarine fish and shellfish, including red drum, spotted seatrout, Atlantic croaker, brown and white shrimp, and eastern oysters, were found in each of the four areas. In many cases, however, the catch rates for a given life stage were much higher in one or two areas than they were in other areas. This suggests that some species use areas within the estuary differentially, and distinct age groups of the same species (based on animal size) preferentially using different parts of the estuary.

Section 7.6 also recommends general salinity ranges that should be provided in different estuarine areas to sustain the existing populations. Considering that various species and distinct age groups of the same species were associated with different salinity regimes in the Guadalupe Estuary, it is necessary to provide enough fresh water to maintain the salinity regimes in these areas to ensure that estuarine species have all of the environmental conditions they need to complete their life cycles.

CHAPTER 8: GUADALUPE ESTUARY EXAMPLE ANALYSIS

8.0 INTRODUCTION

Purpose

The purpose of this chapter is to introduce two computer models for analyzing freshwater inflows, and to demonstrate their use as applied to the Guadalupe Estuary. The first is an optimization model, the Texas Estuarine Mathematical Programming (TXEMP) Model, that is used to determine the quantity and timing of freshwater inflows that fulfill some stated management objectives. The second is a hydrodynamic and conservative transport model, TXBLEND, that is used to demonstrate circulation and salinity patterns given a particular regime of tides, wind, rainfall, and freshwater inflow.

In the following sections, a mathematical description of the TXEMP Model, the specific equations and the conditions used in the model, and the results of the model application to the Guadalupe Estuary are presented. In addition, simulated water movements and salinities from the TXBLEND Model are shown. These results serve as a way of highlighting the various aspects of the modeling process applied to the problem of determining freshwater inflow requirements.

Optimization Model

The TXEMP Model was developed as a tool to study the effect of freshwater inflow to the bays and estuaries in Texas and to help establish long-term objectives of maintaining and enhancing an ecologically sound environment for the estuarine systems. The optimization model is the culmination of the analytical method used to determine inflows necessary to maintain productivity. Information including equations that relate salinity or harvest with inflow, critical limits involving salinity for viability or inflow for nutrient and sediment loading, and various objectives for management provide the data needed to run the model. The model calculates the optimum seasonal or monthly inflows that fulfill the objectives while still satisfying the critical limits for inflow or salinity.

Background of the model. The model is a nonlinear, stochastic, multiobjective mathematical programming model (or optimization model). It is a completely new second generation model building upon the experience of the first Estuarine Linear Programming Model developed by the Texas Department of Water Resources (TDWR 1980a, 1980b; Martin 1987). The TDWR model was a pioneering effort in the application of operations research techniques to the estuarine management problem. The new TXEMP Model expanded on the first model through its treatment of nonlinear equations, incorporation of chance constraints, and analysis of the problem using the multiobjective approach. Initial development of the new model was performed by Bao et al. (1989) at the Center for Research in Water Resources, University of Texas at Austin.

Objective functions and constraints. There are six objective functions that can be activated one at a time in the model. They are the minimization or maximization of annual inflow, the minimization or maximization of fishery harvest, and the maximization of salinity probability or harvest probability. The constraint set includes salinity conditions, minimum and maximum allowable inflows, and targets for fishery harvests. Salinities and fishery harvests are related to freshwater inflows through regression equations. Salinity and fishery harvest constraints are treated as chance constraints. The constraint set also includes critical limits on sediment and nutrient loading that are consistent with water rights permitting legislation.

A performance or tradeoff curve between fishery harvest and freshwater inflow can be generated to examine how the harvest changes as the inflow varies from minimum to maximum.

Hydrodynamic and Conservative Transport Model

The TXBLEND Model. TXBLEND is another computer model developed for the Bay and Estuary Studies. The model is a depth-averaged, two dimensional finite element model that simulates water circulation and salinity patterns in estuaries. To demonstrate the use of the model, one set of optimal inflows found by the TXEMP Model is used as input inflows to TXBLEND, and the response of the estuarine system to the new conditions is examined.

Use of the model in inflow analysis. The hydrodynamic and conservative transport model is useful in assessing the changes in circulation and salinity resulting from alterations in inflow, bathymetry, or inlet configuration. For this study, however, this model is used as a check on the calculations from the TXEMP Model. Once the monthly inflows needed to fulfill management objectives are calculated, the hydrodynamic and conservative transport model is run to determine the salinity distribution throughout the bay for each month. Isohalines from this model can be compared to known salinity requirements for the species of interest to confirm that salinities do not exceed the critical levels specified in the TXEMP Model.

8.1 DESCRIPTION OF THE TXEMP MODEL FOR THE GUADALUPE ESTUARY

Background

Assumptions. There are two basic assumptions in the TXEMP Model. One is the consistency of the estuarine system. All the biological and hydrological processes are assumed to occur in the same way year after year, at least in the average sense. The other assumption is the controllability of the inflows. We assume inflows to the estuary can be controlled 100% of the time as to their amount and timing.

Terminology. An objective function mathematically defines the criterion for evaluating the system's performance. It is the objective for the system to strive for, and, as such, it guides the search for a better solution.

A constraint set is a collection of conditions imposed on a particular problem. For instance, budgetary restrictions or natural and human resource restrictions are common conditions imposed on economic optimization problems. A constraint set defines a feasible region of solutions; any solution within a feasible region satisfies all the conditions. But, the quality of the solution depends on the objective function.

Objective Functions

MinQ and MaxQ. One of the most frequently asked questions for any water resource system is: what is the minimum inflow (MinQ) needed to satisfy all of the requirements without violating any constraints? For such a case, the objective function is:

$$\min \sum_{j=1}^{12} Q_j \qquad (8.1.1)$$

where Q_j represents the inflow for the jth month. MinQ, therefore, is the minimum of the sum of the flows for all 12 months.

If we desire to know the maximum inflow, MaxQ, needed to satisfy all of the requirements, the objective function is:

$$\max \sum_{j=1}^{12} Q_j \qquad (8.1.2)$$

The MinQ and MaxQ solutions define the feasible range of annual inflow. In other words, any inflow above MaxQ or below MinQ cannot meet one or more of the constraints.

MinH and MaxH. Optimization methods for problems in natural resource management often involve maximizing the yield or harvest of a living population. The objective function for maximizing fishery harvest is:

$$\max \sum_{k=1}^{7} H_k \qquad (8.1.3)$$

where H_k represents the fishery harvest of the kth species. This study presents harvest equations for seven species.

When a particular species or a set of selected species are to be optimized, the objective function can be expressed as:

$$\max \sum_{k=1}^{7} p_k \cdot H_k \qquad (8.1.4)$$

where p is a preference vector and $p_k = 1$ if the kth species is to be included, 0 otherwise. Another way of using p is as a profit vector where p_k represents a unit profit or a relative weight assigned to the kth species.

Although we are mainly interested in MaxH solutions, it is possible to determine the minimum harvest for a given set of conditions. Then the objective function becomes:

$$\min \sum_{k=1}^{7} H_k \qquad (8.1.5)$$

MaxSalP and MaxHarP. It may be useful to determine the maximum attainable probability of not violating the salinity constraints. In such a case, the objective function is:

$$\max \ SalP \qquad (8.1.6)$$

where *SalP* represents the salinity probability.

Similarly, it may be of interest to know the maximum attainable probability of not violating the fishery harvest constraints. In this instance,

$$\max \ HarP \qquad (8.1.7)$$

is the appropriate objective function, where *HarP* represents the harvest probability.

Constraints

Salinity constraints. A deterministic form of the salinity constraint is:

$$SLB_{ij} \le S_{ij} \le SUB_{ij} \qquad (8.1.8)$$

for $i = 1, 2, 3$, and $j = 1, \ldots, 12$. S_{ij} represents the salinity in parts per thousand (‰) at location i for month j, and SLB_{ij} and SUB_{ij} are the lower and upper bounds for the salinity, respectively. For the Guadalupe Estuary, $i = 1$ for upper San Antonio Bay, 2 for lower San Antonio Bay, and 3 for Espiritu Santo Bay.

Two forms of salinity regression equations can be used in the model. One is a simple regression equation relating the salinity directly to the freshwater inflows:

$$S_{ij} = a_i Q_j^{b_i} \qquad (8.1.9)$$

Here, S_{ij} is the calculated salinity, Q_j is the monthly combined inflow, and a_i and b_i are regression constants. The second form is a multiple regression equation relating the salinity to the antecedent salinity from the previous month and the freshwater inflows:

$$S_{ij} = a_i Q_j^{b_i} AS_{ij}^{c_i} \qquad (8.1.10)$$

In this equation, S_{ij} is the calculated salinity, Q_j is the monthly combined inflow, AS_i represents the antecedent salinity, and a_i, b_i, and c_i are the regression constants.

While a deterministic constraint such as (8.1.8) requires that the constraint be satisfied without any violation, a chance constraint may allow the constraint to be violated to some degree. Usually, the degree of enforcement is expressed in terms of probability. Let $SalP_i$ represent the probability level at which we want the salinity constraint (8.1.8) to be satisfied. Then the salinity chance constraint is written as:

$$\text{Prob}\ \{ SLB_{ij} \le S_{ij} \le SUB_{ij} \} \ge SalP_i \qquad (8.1.11)$$

where Prob is the abbreviation for probability. In other words, the chance constraint (8.1.11) means that the probability of satisfying both the salinity lower bound and the upper bound must be $SalP_i$ or greater for site i for any month (although a different month may have different bounds).

Constraint (8.1.11) may be a natural way to express the salinity chance constraint, but in initial testing, we found that the constraint was too sharply dependent on the bounds *SLB* and *SUB*, and that it is difficult to treat the salinity probability uniformly among different locations. Therefore, this constraint was changed to treat the bounds separately. The chance constraints for the lower and upper salinity bounds with $i = 1, 2, 3$ and $j = 1, \ldots, 12$ are:

$$\text{Prob}\ \{ S_{ij} \ge SLB_{ij} \} \ge SalP_i \qquad (8.1.12)$$

$$\text{Prob}\ \{ S_{ij} \le SUB_{j} \} \ge SalP_i \qquad (8.1.13)$$

Figure 8.1.1 illustrates the salinity chance constraint with 50% salinity probability. The distribution of salinity for a given inflow is assumed to follow Student's t-distribution. The probability level (*SalP*) is the area under the t-distribution and above the salinity lower bound (*SLB*) for salinity chance constraint (8.1.12), and the area below the salinity upper bound (*SUB*) is for salinity chance constraint (8.1.13).

That the 50% probability case corresponds to the deterministic case can be explained as follows. Because the regression line consists of the points of expected values, i.e., the points which will be exceeded 50% of the time if the regression equation is used to find the inflows corresponding to the *SLB* and *SUB*, the deterministic salinity constraint (8.1.8) becomes equivalent to the 50% salinity probability case.

Figure 8.1.2 shows the effect of salinity probability on the feasible inflow range. As the probability assigned to *SalP* is increased, the limits of the feasible inflow range move toward the midpoint because inflows at or near the midpoint of the feasible inflow range have the highest probability of satisfying both the lower and upper salinity bounds. Thus, as the salinity probability level is increased, the feasible inflow range becomes narrower.

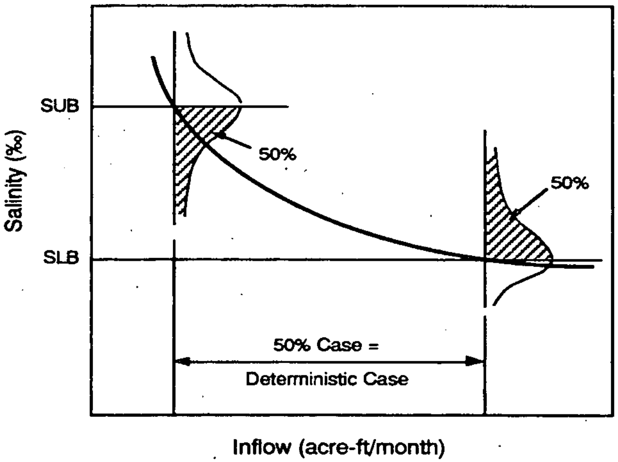

Figure 8.1.1. Feasible range of inflow for 50% salinity probability.

Harvest constraints. The harvest chance constraint is:

$$\text{Prob}\{H_k \geq T_k\} \geq HarP_k \qquad (8.1.14)$$

for $k=1,\ldots,7$ for the species for which harvest equations are available. For the *k*th species, H_k is the calculated level of harvest, T_k represents the harvest target, and $HarP_k$ is the probability level at which we desire H_k to meet the target T_k. In other words, the probability of the harvest meeting or exceeding the target shall be no less than $HarP_k$.

In the TXEMP Model applied to the Guadalupe Estuary, seven fishery species are included: blue crab, oyster, brown shrimp, and white shrimp for the shellfish; and red drum, black drum, and spotted seatrout for the fish. Harvests are estimated using the following form of the regression equation, which mainly relates fishery harvest to seasonal inflows:

$$H_k = a_k\left(\prod_{s=1}^{6} W_s^{b_{ks}} T_s^{c_{ks}}\right) E_k^{d_k} \qquad (8.1.15)$$

In this equation, a_k, b_{ks}, c_{ks}, and d_k are the regression constants, s is an index for seasons, W_s is the seasonal inflow, T_s is the seasonal air temperature, and E_k is the fishing effort in terms of number of trips. In this model, seasonal inflows are defined as the following bimonthly inflows:

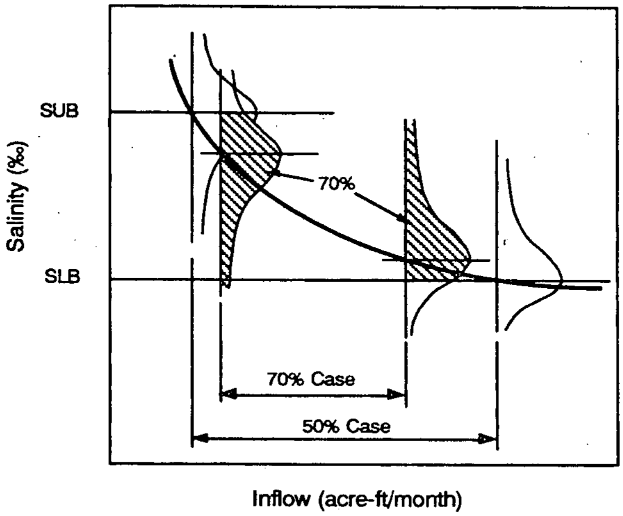

Figure 8.1.2. Feasible range of inflow for 70% salinity probability.

$$\begin{aligned} W_1 &= Q_{JF} = Q(\text{Jan}) + Q(\text{Feb}) \\ W_2 &= Q_{MA} = Q(\text{Mar}) + Q(\text{Apr}) \\ W_3 &= Q_{MJ} = Q(\text{May}) + Q(\text{Jun}) \\ W_4 &= Q_{JA} = Q(\text{Jul}) + Q(\text{Aug}) \\ W_5 &= Q_{SO} = Q(\text{Sep}) + Q(\text{Oct}) \\ W_6 &= Q_{ND} = Q(\text{Nov}) + Q(\text{Dec}) \end{aligned} \qquad (8.1.16)$$

Inflow constraints. Inflow constraints can be of three types: monthly, seasonal, and annual. Monthly inflows are both lower and upper bounded by:

$$QLB_j \leq Q_j \leq QUB_j \qquad (8.1.17)$$

for $j=1,\ldots,12$, where QLB_j is the lower bound and QUB_j is the upper bound for *j*th month. Upper bounds may be the median or mean inflow, while lower bounds are typically some percent of the monthly inflow.

Seasonal inflows (where seasons are two-month periods) are bounded by:

$$WLB_s \leq W_s \leq WUB_s \qquad (8.1.18)$$

for $s = 1, \ldots, 6$, where WLB_s is the lower bound and WUB_s is the upper bound for sth season.

Total or annual inflow can be bounded by:

$$TQLB \leq TQ \leq TQUB \qquad (8.1.19)$$

where TQ is the total inflow, $TQLB$ is the lower bound, and $TQUB$ is the upper bound for the annual inflow.

Sediment constraint. The sediment constraint is designed to maintain the bathymetry of Mission Lake by providing enough suspended sediment to offset subsidence and eustatic sea level rise in the lake. The constraint has the form:

$$SD \geq SDLB \qquad (8.1.20)$$

where SD is the yearly sediment load delivered to the lake and $SDLB$ is the minimum annual sediment load required.

Nutrient constraint. The nutrient constraint is included to avoid nitrogen depletion in the estuary through losses to burial, denitrification, export, and harvest. The constraint is expressed as:

$$NR \geq NRLB \qquad (8.1.21)$$

where NR is the nitrogen remaining and $NRLB$ is the minimum requirement.

The sediment load and nitrogen remaining are related to inflows by regression equations, whose specific forms are presented in Section 8.3.

Other TXEMP Model Features

Multiobjective programming. By combining any one of the above objective functions and the constraints, a mathematical optimization problem is formulated. For instance, a MinQ problem (minimum inflow needed) is formulated with the objective function (8.1.1) and constraints for upper and lower salinity probability (8.1.12 and 8.1.13); harvest (8.1.14); monthly, seasonal, and annual inflows (8.1.17, 8.1.18, and 8.1.19); minimum annual sediment (8.1.20); and minimum nutrient requirement (8.1.21).

The purpose of multiobjective programming or multiobjective analysis is to illuminate the interactions among competing objectives and their consequences. One way to do this is to examine a whole range of solutions from one extreme to the other extreme. A MinQ solution is the minimum inflow needed to satisfy all the constraints and it is at the lower end of the feasible range for the total (or annual) inflow, while a MaxQ solution is at the higher end of the range. In between, there are an infinite number of possible solutions which satisfy all the constraints. Among them is a special solution set called a noninferior set, which is a collection of optimal solutions for different conditions. We can generate the noninferior set as outlined by Cohon (1978) by solving a series of MaxH problems in which only the limit on the total inflow is varied from MinQ to MaxQ.

Stochastic programming. Since the salinity-inflow equations and the fishery harvest equations are statistically derived, their relationships are stochastic. The element of uncertainty is represented by the random component in the equations. How this uncertainty affects the solutions may be seen by varying the salinity probability or the harvest probability from 50% to MaxSalP or MaxHarP. A probability of less than 50% can be assigned, but since a 50% case corresponds to the deterministic case, it is a good reference point to keep.

Additional details concerning the formulation, operation, and testing of this model will be presented in a separate publication.

8.2 SETUP OF THE TXEMP MODEL

Equations Required for the Guadalupe Estuary MP Model

Salinity-inflow regression equations. Two data sets relating observed salinity with freshwater inflow were prepared. One set contained salinity data from the TWDB Coastal Data System, TWDB datasondes, TNRCC Statewide Monitoring Network, and Texas Department of Health Shellfish Sanitation Monitoring Program; the other set contained salinity data from the TPWD Standardized Fishery-Independent Monitoring Program.

Data for three areas were chosen for regression equations, upper San Antonio Bay, lower San Antonio Bay, and Espiritu Santo Bay. For each site, the period of record was divided into consecutive, non-overlapping, seven-day periods. These seven-day periods of salinity measurements were chosen to maximize inclusion of salinity data while minimizing overlapping flow periods. An average salinity was computed for each period for which data was available. This step was taken to minimize the influence on the regression of records representing intensive data collection efforts. (The three areas from which data was selected are illustrated in Figure 8.4.1 and later figures in Section 8.4 of this chapter.)

For freshwater inflow, the combined inflow, consisting of the gaged, ungaged, diversion, and return flows was used. The monthly combined inflow was computed by summing the daily combined inflows into sliding, non-calendar, 30-day periods. These 30-day inflow periods were chosen specifically to fit the monthly time span of the mathematical programming model. For the simple regression, the salinity associated with the mid-date of a seven-day period is assumed to be the result of the monthly combined inflow accumulated during the previous 30 days.

For the multiple regression, the salinity-inflow data set generated for the simple regression was searched for salinity pairs approximately one month apart, which greatly reduced the number of observed data (Table 8.2.1). The salinity for the multiple regression is assumed to be correlated to the 30-day antecedent salinity and the monthly combined inflow. The salinities calculated by the salinity-inflow equations provide estimates of average salinities that would result from relatively constant flows during the previous 30 days. If steady-state flow conditions ensued for 30 additional days, the same salinity levels would be expected and would be considered the average monthly salinity.

Table 8.2.1 lists the regression statistics for the two forms of the equations. In general, the multiple regression form of the equation has a better correlation in terms of the coefficient of determination, or r^2. In the initial testing, however, we found that the salinities resulting from the minimum inflow solutions are near or at the upper salinity bounds, and the equations with antecedent salinity terms tend to extrapolate computed salinity values beyond the upper range of salinities used to compute the regressions. This was due to the scarcity of data at the high end of the salinity range for the smaller multiple regression data sets. To avoid the extrapolation, we adopted the simple regression equations based on the data set that contains the most data in the higher end of the observed salinities.

The coefficients of determination of the adopted equations account for 27 to 50% of the variance in salinity. Equations for zones closer to the river mouth account for a greater proportion of the variance than those farther away. Although the proportion of the variance in salinity accounted for by freshwater inflow seems modest, it should be noted that salinity also depends on wind direction, duration, and velocity, precipitation falling directly on the estuary, and probably some aspect of the annual tidal cycle in the bay.

Table 8.2.1. Salinity-inflow regression statistics for the Guadalupe Estuary. The data source noted as "TWDB" includes data from the TWDB Coastal Data System and Datasonde programs, TNRCC Statewide Monitoring Network, and TDH Shellfish Sanitation Monitoring Program. The data source termed "TWDB + TPWD" includes the TWDB data and surface readings from the TPWD Standardized Fishery-Independent Montoring Program. Equation type 1 is a simple regression of salinity on freshwater inflow, while equation type 2 is a multiple regression of salinity on inflow and 30-day antecedent salinity. The number of data points used in the regression is denoted by N, while R^2 is the coefficient of determination, and S.E. is the standard error of the regression.

Site	Data source	Equation type	N	R^2	S.E.	Salinity range (‰)
Upper San Antonio Bay	TWDB	1	225	0.54	1.03	0.01 to 19.94
	TWDB+TPWD[a]	1	359	0.50	1.16	0.01 to 32.20
	TWDB	2	135	0.72	0.86	0.01 to 19.43
	TWDB+TPWD	2	198	0.61	1.01	0.01 to 26.60
Lower San Antonio Bay	TWDB	1	99	0.58	0.56	0.44 to 32.63
	TWDB+TPWD[a]	1	215	0.48	0.66	0.01 to 35.25
	TWDB	2	25	0.70	0.54	0.44 to 22.18
	TWDB+TPWD	2	84	0.46	0.80	0.01 to 27.20
Espiritu Santo Bay	TWDB	1	126	0.27	0.33	4.91 to 47.06
	TWDB+TPWD[a]	1	273	0.27	0.38	0.60 to 47.06
	TWDB	2	10	0.38	0.47	6.20 to 33.65
	TWDB+TPWD	2	92	0.28	0.42	0.60 to 35.00

[a] Indicates equations adopted in the TXEMP Model for the Guadalupe Estuary.

Since there is some human control over freshwater inflow but none for weather conditions, direct rainfall, or tidal effects, the equations are presented only in terms of the effect of inflow on salinity. It is not surprising, therefore, that the explained variance is not higher. Nevertheless, we think these regression equations capture the essential element of the mixing process well enough to be used in the TXEMP Model.

Three sites were chosen for developing salinity-inflow regression equations to ensure retention of a salinity gradient along the major axis of the estuary. Figure 8.2.1 illustrates the salinity-inflow regression equation for upper San Antonio Bay in which the regression is:

$$S = 4366.1 \cdot Q^{-1.3702} \tag{8.2.1}$$

where S is the salinity in ‰ and Q is the monthly inflow in 1,000 acre-ft. Figure 8.2.1 clearly depicts the scatter in the observed data. This high variability is reflected in the large standard error. As a result, the confidence interval on prediction has a wide range. The narrow confidence interval on the regression line is due to a small standard error of the mean because the number of observations is large.

Figures 8.2.2 and 8.2.3 illustrate the regression lines and the confidence intervals for lower San Antonio Bay, for which the regression is:

$$S = 658.63 \cdot Q^{-0.7693} \tag{8.2.2}$$

and for Espiritu Santo Bay:

$$S = 85.21 \cdot Q^{-0.2675} \tag{8.2.3}$$

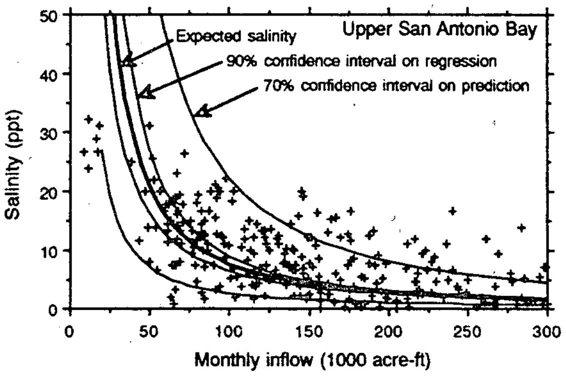

Figure 8.2.1. Salinity-inflow regression and confidence interval for upper San Antonio Bay.

For comparison, Figure 8.2.4 shows all three regression lines together. Roughly speaking, these salinity equations are 10‰ apart, in the normal range of inflow of 100,000 to 250,000 acre-ft/month.

Fishery harvest equations. Seven fishery harvest equations from Section 6.9 are used in the application of the TXEMP Model to the Guadalupe Estuary and are listed in Table 8.2.2. Some of the equations include variables for temperature and fishing effort that are not decision variables. Since they play no role in the optimization process, the non-decision variables were set to their average values over the last five years of the study period. In this way, the solutions to the harvest equations should reflect recent conditions, rather than those that existed several decades ago.

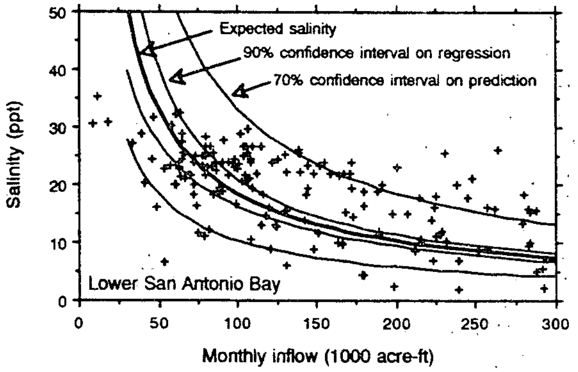

Figure 8.2.2. Salinity-inflow regression and confidence interval for lower San Antonio Bay.

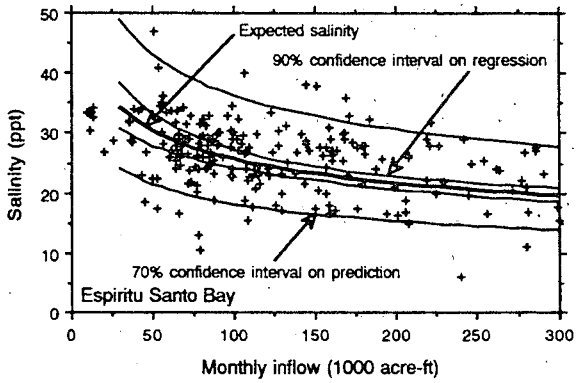

Figure 8.2.3. Salinity-inflow regression and confidence interval for Espiritu Santo Bay.

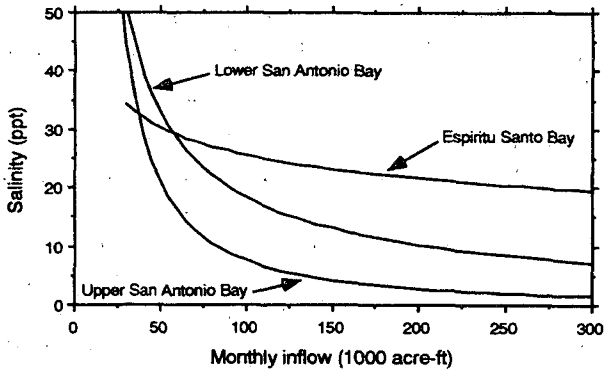

Figure 8.2.4. Salinity-inflow regressions for the Guadalupe Estuary.

Objective Functions

Ten cases were run to test the TXEMP Model. The weightings used in objective function 8.1.4 for the ten cases are listed in Table 8.2.3. Of the ten cases, seven are single species optimizations and three apply different weightings on various species. In Table 8.2.3, all the weights sum to one, which is perfectly valid but not necessary; any reasonable value can be used to weight the relative importance of each species. The selection of these species and weighting coefficients are presented here as an example. In actual practice, the final selection of these constraints is left up to the user and may be modified to suit policy needs.

Constraints

Harvest targets. Table 8.2.4 lists the mean, minimum, and maximum historical harvests for the period 1962 through 1987. For initial testing, targets for fishery harvests were set to the historical mean levels for the 50% harvest probability level. This implies that we desire to harvest at least at the historical mean levels 50% of the time. In other words, there will be no harvest reduction of any species to enhance others.

Testing showed that requiring mean harvests for all species was too restrictive a goal; consequently, targets were set to 80% of the historical mean for the example analyses. This increases maneuverability of the mathematical programming model to search for an optimal solution because as the target level is reduced, the feasible region expands.

All the harvest probabilities were set to 50% except for black drum which was set at 0%. During initial testing, it was noticed that the black drum equation behaves differently from other equations. It requires much more inflow than other species, which may or may not reflect the true relationship. Inclusion of black drum at 50% harvest probability made the optimization problem infeasible. Several fishery experts suggested that the black drum data was of low quality. Others pointed out the low historical harvest and a relatively low economic value for the species. Consequently, the harvest probability for black drum was set to 0%. This means the black drum equation does not influence the harvest of other species, but the harvest of black drum is computed according to the equation.

Monthly inflow bounds. Table 8.2.5 lists the mean, median, 1984, and 10th percentile monthly inflows. To set the lower bounds on the monthly inflows, several test runs were made using various values such as the historically lowest inflows and a percentage of mean inflows. As a result of these test runs, we set the lower bound to the 10th percentile inflows. However, any regulatory minimum streamflow requirement could be set as the lower bound.

The upper bounds on monthly inflows were set to the historical median monthly inflows. The upper bounds could be set to other values such as historical mean inflows or some percentage of them. In this instance, the median was substantially smaller than the mean: 1.53 million acre-ft versus 2.35 million acre-ft on an annual basis. The difference between the median and the mean shows that inflows are not normally distributed, probably because of a few very large inflows. In this instance, the median is a better measure of central tendency than the mean.

The upper bounds on the monthly inflows are of crucial importance because they directly influence the higher end of the total inflow requirement or the total allowable inflow. Similarly, the lower bounds are critical in determining the lower end of the total inflow needed.

Seasonal inflow bounds. Table 8.2.6 lists the bounds on the seasonal inflows. They are set to roughly reflect the lower and upper bounds of the seasonal historical values. These bounds prevent extrapolating harvests from the fishery equations beyond the inflow levels over which the equations are valid. The upper bounds on seasonal inflows in Table 8.2.6 are much higher than the sum of monthly upper bounds for the corresponding periods. Therefore, the upper bounds on seasonal inflows do not affect the higher side of the inflows. The lower bounds on the seasonal inflows, however, are greater than the sum of the monthly lower bounds. Thus, the lower side of monthly inflows will be affected by the lower bounds on the seasonal inflows.

Salinity bounds. Salinity bounds are another important factor that greatly influence the outcome of TXEMP. Salinity bounds are another element of TXEMP that may be changed by the user and revised as policy needs change. Table 8.2.7 lists example salinity viability limits adopted for upper San Antonio Bay in the previous study (TDWR 1980b) and by Espey, Huston & Associates, Inc. (EH&A

Table 8.2.2. Harvest equations for the Guadalupe TXEMP Model: H is harvest in pounds; Q_{ij} is the seasonal inflow for months i and j combined; T_{ij} is the temperature for months i and j; E is annual fishing effort; and the abbreviations for months are: JF = January + February, MA = March + April, MJ = May + June, JA = July + August, SO = September + October, and ND = November + December.

Species	Equation
Oyster	$\ln H = 38.148 - 1.200 \cdot \ln Q_{JF} + 1.354 \cdot \ln Q_{MA} - 1.073 \cdot \ln Q_{MJ} - 0.3407 \cdot T_{JA}$
Brown shrimp	$\ln H = 5.9168 + 0.2126 \cdot \ln Q_{MJ} + 0.000793 \cdot E - 0.06126 \cdot T_{JF}$
White shrimp	$H = -233.56 + 174.3 \cdot \ln Q_{JF} + 83.53 \cdot \ln Q_{MJ} - 180.4 \cdot \ln Q_{ND} + 0.1314 \cdot E$
Blue crab	$\ln H = 14.413 - 0.9076 \cdot \ln Q_{MA} + 1.884 \cdot \ln Q_{JA} - 1.167 \cdot \ln Q_{SO} - 0.1182 \cdot T_{JF}$
Red drum	$\ln H = 4.1138 + 1.358 \cdot \ln Q_{JA} - 1.249 \cdot \ln Q_{ND}$
Spotted seatrout	$\ln H = 2.5900 + 1.475 \cdot \ln Q_{MJ} - 1.366 \cdot \ln Q_{ND}$
Black drum	$H = -947.81 + 58.96 \cdot \ln Q_{JF} - 52.20 \cdot \ln Q_{MA} + 60.56 \cdot \ln Q_{ND} + 8.482 \cdot T_{SO}$

Table 8.2.3. Relative weights of species for ten different harvest objective cases. In case 1, the harvest of all species are weighted equally; cases 2 through 8 are single species harvest objectives; case 9 has equal weights for oyster, brown shrimp, red drum, and spotted seatrout; in case 10, the weights are based on the value of the commercial catch of each species in the estuary.

Species	Case									
	1	2	3	4	5	6	7	8	9	10
Oyster	0.14	1.0	0.0	0.0	0.0	0.0	0.0	0.0	0.25	0.02
Brown shrimp	0.14	0.0	1.0	0.0	0.0	0.0	0.0	0.0	0.25	0.27
White shrimp	0.14	0.0	0.0	1.0	0.0	0.0	0.0	0.0	0.00	0.08
Blue crab	0.14	0.0	0.0	0.0	1.0	0.0	0.0	0.0	0.00	0.01
Red drum	0.14	0.0	0.0	0.0	0.0	1.0	0.0	0.0	0.25	0.08
Spotted seatrout	0.14	0.0	0.0	0.0	0.0	0.0	1.0	0.0	0.25	0.49
Black drum	0.14	0.0	0.0	0.0	0.0	0.0	0.0	1.0	0.00	0.05

1986), along with the proposed bounds by the Texas Parks and Wildlife Department (TPWD) for the current study. Table 8.2.8 lists the viability limits and proposed bounds for lower San Antonio Bay.

In addition to the two sites in San Antonio Bay, the third control point in Espiritu Santo Bay was selected by the TPWD and the salinity bounds were set to 10 to 40‰. The proposed TPWD salinity ranges do not differ from month to month although there would be no problem allowing them to vary monthly. The results presented in Section 8.3 are based on the TPWD salinity range. These bounds were intentionally selected to represent wide salinity zones and not limit TXEMP's ability to find a feasible solution. Salinity requirements for individual species as discussed in Section 6.2 are useful for refining salinity bounds that will be used in future analyses.

Table 8.2.4. Historical annual fishery harvests (1,000 lbs) from 1962 through 1987.

Species	Min	Max	Mean	80% of mean
Oyster	53.9	1,937.2	276.7	221.4
Brown shrimp	67.1	1,830.2	350.6	280.5
White shrimp	241.8	1,415.0	737.5	590.0
Blue crab	276.1	4,775.7	1,336.4	1,069.1
Red drum	24.4	179.2	66.4	53.1
Spotted seatrout	12.2	114.8	46.7	37.4
Black drum	0.4	131.0	40.5	32.4

Table 8.2.5. Mean, median, 1984, and 10th percentile monthly combined inflows (1,000 acre-ft) for the Guadalupe Estuary.

Month	Mean	Median	1984	10th percentile
Jan	160.8	111.2	90.3	33.9
Feb	177.8	124.2	58.5	46.2
Mar	152.9	117.5	90.0	43.0
Apr	200.4	110.5	37.7	42.2
May	294.9	222.6	39.7	61.0
Jun	295.5	162.7	20.0	36.2
Jul	158.5	95.2	9.1	22.3
Aug	118.5	94.9	22.9	18.5
Sep	241.4	139.4	11.6	38.5
Oct	226.3	138.4	147.9	42.7
Nov	168.9	116.1	89.6	33.1
Dec	149.7	104.1	82.5	38.5
Total	2,345.6	1,536.8	699.8	456.1

Table 8.2.6. Bounds on seasonal inflows (1,000 acre-ft). Sum of monthly lower bounds (LBs) is the bimonthly sum of the 10th percentile flows from Table 8.2.5. Sum of the monthly upper bounds (UBs) is the bimonthly sum of the median flows from Table 8.2.5.

Period	Seasonal lower bound	Seasonal upper bound	Sum of monthly LBs	Sum of monthly UBs
Jan-Feb	100	500	80.1	235.4
Mar-Apr	100	600	85.2	228.0
May-Jun	120	800	97.2	385.3
Jul-Aug	80	500	40.8	190.1
Sep-Oct	100	600	81.2	277.8
Nov-Dec	100	500	71.6	220.2

Sediment constraint. The equation relating sediment load to inflow is:

$$SD = 0.78742 \cdot TQ_g^{0.96655} \qquad (8.2.4)$$

where SD is the sediment load in tons/yr and TQ_g is the gaged inflow in acre-ft/yr. See Section 7.5 for its derivation.

The inflow in the TXEMP Model represents the combination of gaged and ungaged inflows. Gaged inflow (TQ_g) on average represents 80.85% of the combined inflow (TQ) into the Guadalupe Estuary ($TQ_g = 0.8085 \cdot TQ$). Therefore, equation (8.2.4) must be modified as follows:

$$SD = 0.78742 \cdot (0.8085 \cdot TQ)^{0.96655}$$

Table 8.2.7. Salinity bounds for upper San Antonio Bay (‰). The lower viability limits (LVL) and upper viability limits (UVL) recommended by the TDWR come from TDWR (1980b). The Espey, Huston & Associates salinity upper bound (SUB) estimate comes from EH&A (1986), and the TPWD salinity lower bound (SLB) and salinity upper bound (SUB) recommendations were made by representatives of the Coastal Fisheries and Resource Protection divisions of the TPWD.

Month	TDWR		EH&A		TPWD	
	LVL	UVL	SLB	SUB	SLB	SUB
Jan	10	20	-	15	1	20
Feb	10	20	-	15	1	20
Mar	10	20	-	15	1	20
Apr	5	15	-	15	1	20
May	1	15	-	15	1	20
Jun	1	15	-	15	1	20
Jul	10	20	-	20	1	20
Aug	10	20	-	20	1	20
Sep	5	15	-	15	1	20
Oct	5	15	-	15	1	20
Nov	10	20	-	15	1	20
Dec	10	20	-	15	1	20

Table 8.2.8. Salinity bounds for lower San Antonio Bay (‰). The lower viability limits (LVL) and upper viability limits (UVL) recommended by the TDWR come from TDWR (1980b). The Espey, Huston & Associates salinity upper bound (SUB) estimate comes from EH&A (1986), and the TPWD salinity lower bound (SLB) and salinity upper bound (SUB) recommendations were made by representatives of the Coastal Fisheries and Resource Protection divisions of the TPWD.

Month	TDWR		EH&A		TPWD	
	LVL	UVL	SLB	SUB	SLB	SUB
Jan	10	30	-	25	5	25
Feb	10	30	-	25	5	25
Mar	10	25	-	25	5	25
Apr	5	20	-	25	5	25
May	5	20	-	25	5	25
Jun	5	20	-	30	5	25
Jul	10	25	-	30	5	25
Aug	10	25	-	30	5	25
Sep	5	20	-	25	5	25
Oct	5	20	-	25	5	25
Nov	10	30	-	25	5	25
Dec	10	30	-	25	5	25

$$SD = 0.64117 \cdot TQ^{0.96655} \qquad (8.2.5)$$

From Section 7.5, the minimum requirement for the sediment is 182,415 metric tons, so the sediment constraint is:

$$SD \geq 182{,}415 \qquad (8.2.6)$$

Nutrient constraint. The relationship between the quantity of nitrogen remaining in the Guadalupe Estuary in excess of losses and the inflow is given by:

$$NR = -490.79 + 0.0017148 \cdot TQ \quad (8.2.7)$$

where NR represents the nitrogen remaining in millions of grams (see Section 7.4 for the derivation) and TQ is the total inflow in acre-ft/yr. The nutrient constraint to avoid the depletion of nitrogen remaining in the estuary is:

$$NR \geq 0 \quad (8.2.8)$$

8.3 RESULTS OF MODEL APPLICATION TO THE GUADALUPE ESTUARY

Example Cases

The TXEMP Model was executed for the maximization of the salinity probability, and the maximum salinity probability (MaxSalP) was found to be 63.6%. Based on this information, two salinity probabilities were selected to examine the effect of salinity probability on the inflows and the harvests. One is 50%, which corresponds to the deterministic case and the other is 57%, which is a mid-point between 50% and the MaxSalP.

Results for single species models. Tables 8.3.1 and 8.3.2 list the results of the ten cases for 50% and 57% salinity probabilities, respectively. Additionally, MinQ and MaxQ bounds on monthly inflows, and historical minimum and maximum harvests are listed for comparison. It may be seen from cases 2 through 8 (single species optimizations) that the model sets the inflows as high as possible for the months (or seasons) with positive coefficients in the harvest equation, and as low as possible for the months with negative coefficients. For those months with no variable in the harvest equation, inflows can be any value as long as they do not cause violations of any constraint. For some cases, the inflow for the May-June season is set so that the target for seatrout harvest is satisfied. When there are alternative optimal solutions, i.e., same objective values but different inflows, those solutions listed in Tables 8.3.1 and 8.3.2 are the ones that require the least inflow among the solutions found by the model.

Results for multiple species models. Cases 1, 9, and 10 are multiple species optimizations. For some species, periods of positive correlation overlap, but, in general, they compete with each other for inflows. How the inflows are allocated depends on the objective function: usually, the most valuable species gets the first allocation. Case 1 is an instance of uniform weighting. The model will first choose the species with the largest harvest potential in terms of weight, such as blue crab, because it will contribute most to the objective function. Case 9 focuses on four species: oyster, brown shrimp, red drum, and seatrout. The model predicts the harvest of oyster and red drum at the same high level as in the single species optimization case, seatrout at the target level, and brown shrimp at a relatively high level. Case 10 is a weighted scheme emphasizing seatrout and brown shrimp harvest. The model predicts the harvest of seatrout and brown shrimp at the same high level as in the single species optimization case.

By comparing the solutions for the same cases in Tables 8.3.1 and 8.3.2, the effect of salinity probability can be seen. In general, the higher the salinity probability, the lower the harvest. Of course, as the salinity probability increases, there is more assurance that the salinity constraint will be met.

The value 70.3 (70,300 acre-ft) in Table 8.3.1 and the value 81.8 (81,800 acre-ft) in Table 8.3.2 occur several times in the tables. These amounts represent the minimum monthly inflows needed to satisfy the salinity constraint for lower San Antonio Bay for the 50% and 57% salinity probability levels, respectively. During other months, the harvest equations or other constraints determine the inflow requirement.

Estuarine Performance Curves for the Case 1 Example

Figure 8.3.1 illustrates the performance curves of total fishery harvest against annual inflow for the uniform weighting case (Case 1), for the 50 and 57% salinity probabilities, and MaxSalP. The same type of family of curves could be generated by varying the probability of harvest, HarP, to the maximum, MaxHarP. These curves are the envelopes of the maximum harvest points (or MaxH solutions) for bounded inflows, and they are the "noninferior sets" in the sense that for a given set of conditions, there is no better solution than the one on the envelopes. They are also the tradeoff curves from the point of view that we trade freshwater inflow for fishery harvest. The end points of these curves are the MinQ and MaxQ points. The irregularity of the curves is due to the nonlinearity of the problem.

Interpreting the curves. One of the distinctive features of these curves is that they have peaks not at the end points, but between those points. The left side of the curve represents the condition of too little inflow, or "inflow deficit," where the harvest is decreased as the inflow decreases away from the peak. Similarly, the right side of the curve represents the condition of too much inflow, or "inflow surplus."

Table 8.3.1.- Monthly inflows and harvests for ten cases at 50% salinity probability. Case 1 is equal weighting for all species; cases 2 through 8 are weighted only for the species named; case 9 has equal weights for oyster, brown shrimp, red drum, and spotted seatrout; and case 10 is weighted based on the value of the commercial catch of each species in the estuary.

Case	Uniform weight (1)	Oyster (2)	Brown shrimp (3)	White shrimp (4)	Blue crab (5)	Red drum (6)	Spotted seatrout (7)	Black drum (8)	Four species (9)	Value-weighted (10)	MinQ	MaxQ	Monthly inflow	
													QLB	QUB
Monthly inflow (in acre-ft):														
Jan	70.3	70.3	70.3	111.2	70.3	70.3	70.3	111.2	70.3	111.2	70.3	111.2	33.9	111.2
Feb	70.3	70.3	70.3	124.2	70.3	70.3	70.3	124.2	70.3	124.2	70.3	124.2	46.2	124.2
Mar	85.6	117.5	70.3	70.8	70.3	70.3	70.3	70.3	117.5	85.9	70.3	117.5	43.0	117.5
Apr	85.6	110.5	70.3	70.3	70.3	70.3	70.3	70.3	110.5	85.4	70.3	110.5	42.2	110.5
May	90.1	70.3	222.6	222.6	110.4	110.4	222.6	157.9	123.6	222.6	125.4	222.6	61.0	222.6
Jun	105.6	125.4	162.7	162.7	87.4	87.4	162.7	157.9	72.0	162.7	70.3	162.7	36.2	162.7
Jul	95.2	70.3	70.3	70.3	87.4	95.2	70.3	95.2	95.2	95.2	70.3	95.2	22.3	95.2
Aug	94.9	70.3	70.3	70.3	87.4	94.9	70.3	94.9	94.9	94.9	70.3	94.9	18.5	94.9
Sep	70.3	70.3	70.3	70.3	70.3	70.3	70.3	131.6	70.3	70.3	70.3	139.4	38.5	139.4
Oct	70.3	70.3	70.3	70.3	70.3	70.3	70.3	131.6	70.3	70.3	70.3	138.4	42.7	138.4
Nov	70.3	70.3	70.3	70.3	70.3	70.3	70.3	116.1	70.3	70.3	70.3	116.1	33.1	116.1
Dec	70.3	70.3	70.3	70.3	70.3	70.3	70.3	104.1	70.3	70.3	70.3	104.1	38.5	104.1
Annual inflow	978.8	986.1	1,088.3	1,183.6	935.0	950.3	1,088.3	1,365.3	1,035.5	1,263.3	898.7	1,536.8	456.1	1,536.8
													Historical harvest	
													Min	Max
Annual harvest (in 1,000 lbs):														
Blue crab	4,775.7	2,083.0	3,232.5	3,220.9	4,775.7	4,775.7	3,232.5	2,747.4	3,682.2	4,775.7	3,232.5	1,662.0	276.1	4,775.7
Oyster	1,106.4	1,630.9	409.3	221.4	836.7	836.7	409.3	272.6	1,630.9	287.8	846.5	424.3	53.9	1,937.2
Red drum	157.8	104.7	104.7	104.7	140.8	157.8	104.7	90.0	157.8	157.8	104.7	90.0	24.4	179.2
Black drum	0.4	0.4	0.4	6.6	0.4	0.4	0.4	34.0	0.4	0.4	0.4	8.7	0.4	131.0
Seatrout	37.4	37.4	101.6	101.6	38.0	38.0	101.6	41.0	37.4	101.6	37.4	55.0	12.2	114.8
Brown shrimp	1,179.6	1,179.5	1,362.3	1,362.3	1,182.3	1,182.3	1,362.3	1,305.9	1,179.6	1,362.3	1,179.5	1,362.3	67.1	1,830.2
White shrimp	748.3	748.3	804.9	894.8	749.2	749.2	804.9	797.1	748.3	894.8	748.3	813.8	241.8	1,415.0
Total harvest	8,005.6	5,784.2	6,015.7	5,912.3	7,723.1	7,740.1	6,015.7	5,288.0	7,436.6	7,580.4	6,149.3	4,416.1	675.9	10,383.1

Table 8.3.2. Monthly inflows and harvests for ten cases at 57% salinity probability. Case 1 is equal weighting for all species; cases 2 through 8 are weighted only for the species named; case 9 has equal weights for oyster, brown shrimp, red drum, and spotted seatrout; and case 10 is weighted based on the value of the commercial catch of each species in the estuary.

Case	Uniform weight (1)	Oyster (2)	Brown shrimp (3)	White shrimp (4)	Blue crab (5)	Red drum (6)	Spotted seatrout (7)	Black drum (8)	Four species (9)	Value-weighted (10)	MinQ	MaxQ	Monthly inflow QLB	Monthly inflow QUB
Monthly inflow (in acre-ft):														
Jan	81.8	81.8	96.1	111.2	95.3	96.1	96.1	111.2	81.8	111.2	81.8	111.2	33.9	111.2
Feb	81.8	81.8	96.1	124.2	95.3	96.1	96.1	124.2	81.8	124.2	81.8	124.2	46.2	124.2
Mar	81.8	117.5	96.1	96.1	81.8	96.1	96.1	81.8	117.5	81.8	81.8	117.5	43.0	117.5
Apr	81.8	110.5	96.1	96.1	81.8	96.1	96.1	81.8	110.5	81.8	81.8	110.5	42.2	110.5
May	112.6	112.6	222.6	222.6	114.4	109.1	222.6	157.9	112.6	222.6	112.6	222.6	61.0	222.6
Jun	112.6	112.6	162.7	162.7	114.4	116.2	162.7	157.9	112.6	162.7	112.6	162.7	36.2	162.7
Jul	95.2	86.6	95.2	95.2	95.2	95.2	95.2	95.2	95.2	95.2	81.8	95.2	22.3	95.2
Aug	94.9	86.6	94.9	94.9	94.9	94.9	94.9	94.9	94.9	94.9	81.8	94.9	18.5	94.9
Sep	81.8	86.6	96.1	96.1	81.8	96.1	96.1	131.6	96.1	81.8	81.8	139.4	38.5	139.4
Oct	81.8	86.6	96.1	96.1	81.8	96.1	96.1	131.6	96.1	81.8	81.8	138.4	42.7	138.4
Nov	81.8	81.8	85.9	81.8	81.8	81.8	81.8	116.1	81.8	81.8	81.8	116.1	33.1	116.1
Dec	81.8	81.8	85.9	81.8	81.8	81.8	81.8	104.1	81.8	81.8	81.8	104.1	38.5	104.1
Annual inflow	1,069.7	1,126.8	1,323.8	1,358.8	1,100.3	1,155.6	1,315.6	1,388.3	1,162.7	1,301.6	1,043.2	1,536.8	456.1	1,536.8
													Historical harvest Min	Historical harvest Max
Annual harvest (in 1,000 lbs):														
Blue crab	4,167.1	2,420.2	2,980.2	2,980.2	4,775.7	2,980.2	2,980.2	2,392.9	2,553.2	4,167.1	3,140.2	1,662.0	276.1	4,775.7
Oyster	745.0	1,168.0	429.5	336.8	610.3	763.7	429.5	335.0	1,168.0	270.6	745.0	424.3	53.9	1,937.2
Red drum	130.5	115.0	122.8	130.5	130.5	130.5	130.5	90.0	130.5	130.5	106.4	90.0	24.4	179.2
Black drum	0.4	0.4	0.4	0.4	0.4	0.4	0.4	26.1	0.4	8.1	0.4	8.7	0.4	131.0
Seatrout	37.4	37.4	77.2	82.6	38.3	37.4	82.6	41.0	37.4	82.6	37.4	55.0	12.2	114.8
Brown shrimp	1,215.4	1,215.4	1,362.3	1,362.3	1,219.4	1,215.4	1,362.3	1,305.9	1,215.4	1,362.3	1,215.4	1,362.3	67.1	1,830.2
White shrimp	759.1	759.1	823.3	867.4	786.9	787.3	832.1	797.1	759.1	867.4	759.1	813.8	241.8	1,415.0
Total harvest	7,054.9	5,715.5	5,795.7	5,760.2	7,561.5	5,914.9	5,817.6	4,988.0	5,864.0	6,888.6	6,003.9	4,416.1	675.9	10,383.1

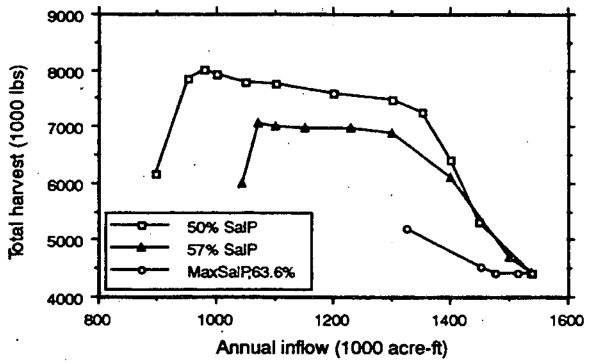

Figure 8.3.1. Total harvest performance curve for the Guadalupe Estuary.

A few more observations can be made from the performance curves. First, these curves have very flat peaks. The uniform case is dominated by the crab harvest because its harvest potential is the largest, four to five times larger than oyster and shrimp. Most of the solutions at or near the peaks include crab at its maximum harvest level. Therefore, these flat peaks may be a strong indication that a relatively wide range of optimal conditions exist for crabs.

Another observation is that the peaks of the 50 and 57% performance curves are located rather near the MinQ points. This could be an indication, at least for this objective function, that the current inflow, 1.5 million acre-ft in terms of the median annual inflow or 2.3 million acre-ft in terms of the mean annual inflow, is too great. According to the performance curves, the inflow could be reduced to 1.0 to 1.1 million acre-ft.

For this particular objective function and constraint set, the effect of the probability level has an impact. The minimum inflow is 0.89 million acre-ft for 50% salinity probability, but increases to 1.04 million acre-ft for 57% salinity probability. The salinity probability has no impact at all on the maximum inflow, which is 1.53 million acre-ft for all salinity probabilities. This is probably the result of a combination of two factors: the upper bounds on the monthly inflows may have been set a little low, and the salinity lower bounds might have been too low. If the salinity lower bounds were raised or if higher upper bounds on monthly inflows were imposed, MaxQ may be different for different salinity probabilities.

As the salinity probability is increased, the feasible range of the inflow is narrowed and the total harvest is decreased. This statement is borne out graphically by the performance curves in Figure 8.3.1, in which a 50% salinity probability case has the widest feasible inflow range and the highest total harvest. At 57% salinity probability, the inflow range and the total harvest are reduced. As the salinity probability is increased further, the feasible inflow range and the total harvest approach the lower limits at MaxSalP, which is 63.6%.

Two Optimal Solutions for Case 1

Comparison of optimal and historical flow patterns. Table 8.3.3 lists the two optimal solutions that are the peak points of the performance curves in Figure 8.3.1. Figure 8.3.2 compares these optimal inflows with the median inflows. The figure shows a reduction in inflows for most periods except for the July-August season. The biggest reduction occurs in the May-June season, followed by the September-October season.

Figure 8.3.3 compares predicted harvests for 50 and 57% salinity probabilities with historical maximum and mean harvests. The dominance of the crab harvest is clearly shown in the figure. The majority of the difference between the 50 and 57% salinity probability solutions is in the crab and oyster harvests.

Comparison of predicted and historical salinities. Figures 8.3.4, 8.3.5, and 8.3.6 illustrate the predicted salinities at three locations in the Guadalupe Estuary (upper and lower San Antonio and Espiritu Santo bays) for 50 and 57% salinity probability cases; the historical average salinities are also shown (Table 8.3.4). These figures show the effect of

Table 8.3.3. Two optimal solutions at the peaks of the estuarine performance curves.

Case	50% SalP	57% SalP
Monthly optimal inflows (1,000 acre-ft):		
Jan	70.3	81.8
Feb	70.3	81.8
Mar	85.6	81.8
Apr	85.6	81.8
May	90.1	112.6
Jun	105.6	112.6
Jul	95.2	95.2
Aug	94.9	94.9
Sep	70.3	81.8
Oct	70.3	81.8
Nov	70.3	81.8
Dec	70.3	81.8
Annual inflow	978.8	1,069.7
Predicted annual harvest (1,000 lbs):		
Crab	4,775.7	4,167.1
Oyster	1,106.4	745.0
Red drum	157.8	130.5
Brown drum	0.4	0.4
Spotted seatrout	37.4	37.4
Brown shrimp	1,179.6	1,215.4
White shrimp	748.3	759.1
Total harvest	8,005.6	7,054.9

increasing the salinity probability on the predicted salinity values. As the salinity probability increases, predicted salinities are less varied within the lower and upper salinity bounds. The figures show that lower salinities would occur during the late spring and summer seasons, reflecting the larger inflows during those periods. They also reflect higher salinities for optimal solutions than is indicated by historical average salinities. This may be reasonable since the annual inflows of the optimal solutions are significantly reduced from the average inflows.

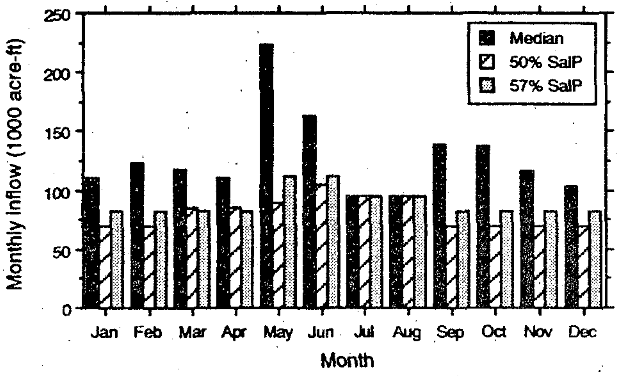

Figure 8.3.2. Optimal and median monthly inflows for the Guadalupe Estuary.

The predicted salinities for upper San Antonio Bay lie in the middle of the lower and upper bounds, as shown in Figure 8.3.4. But for lower San Antonio Bay, the predicted salinities lie at the upper bounds in early spring, fall, and winter seasons, as shown in Figure 8.3.5. In other words, the salinity upper bounds for lower San Antonio Bay are the binding constraints (meaning they are directly influencing the solutions). It should be noted that salinities predicted by TXEMP are meant to be monthly averages, not an instantaneous prediction.

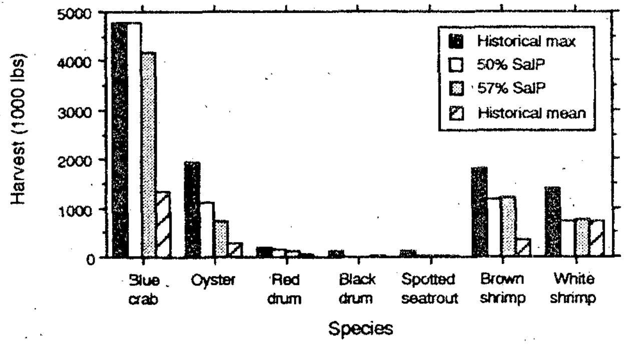

Figure 8.3.3. Predicted and historical fishery harvests for the Guadalupe Estuary.

Figure 8.3.6 for Espiritu Santo Bay manifests a low sensitivity of salinity to the inflows in contrast to Figures 8.3.4 and 8.3.5 that show high sensitivity to inflows in upper and lower San Antonio Bay.

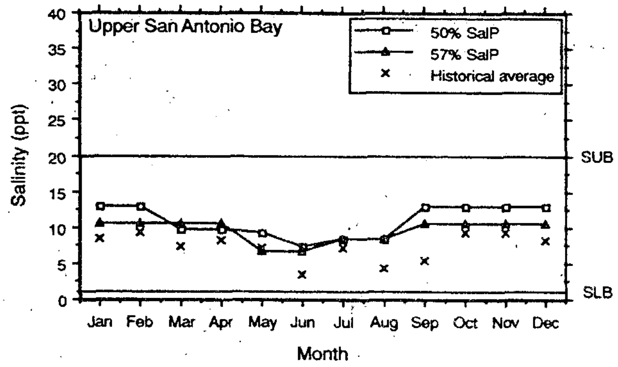

Figure 8.3.4. Predicted salinities in upper San Antonio Bay.

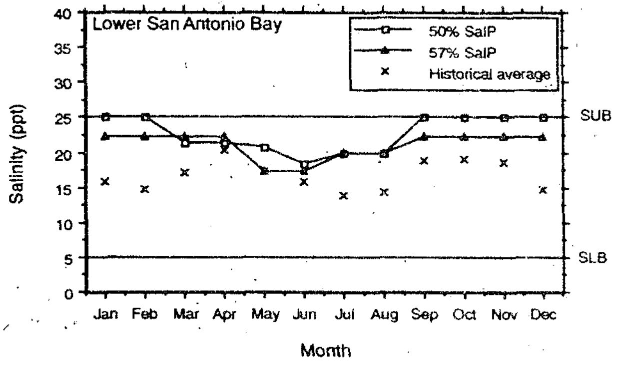

Figure 8.3.5. Predicted salinities in lower San Antonio Bay.

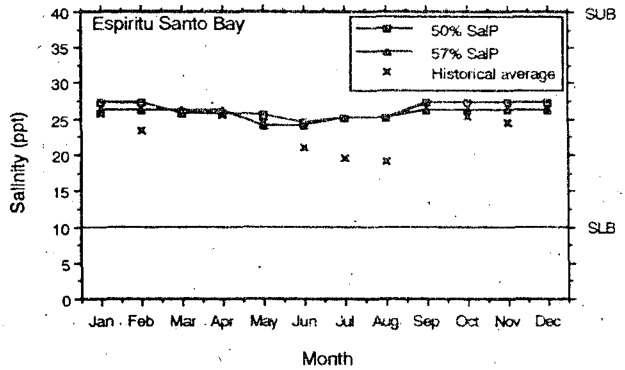

Figure 8.3.6. Predicted salinities in Espiritu Santo Bay.

Sensitivity Studies

The base for comparison for sensitivity studies is Case 1, with uniform weighting among species, 50% salinity probability (*SalP*), 25‰ for the salinity upper bound (*SUB*) in lower San Antonio Bay, and median monthly inflows for the inflow upper bounds (*QUB*s). We have already looked at the effect of *SalP*. In this section, we examine the effects of *SUB*, *QUB*, and the objective function.

Table 8.3.4. Historical average salinities and predicted salinity of the two optimal solutions.

Month	Average	50% SalP	57% SalP
Upper San Antonio Bay:			
Jan	8.4	12.9	10.5
Feb	9.4	12.9	10.5
Mar	7.3	9.8	10.5
Apr	8.3	9.8	10.5
May	7.2	9.2	6.7
Jun	3.4	7.4	6.7
Jul	7.1	8.5	8.5
Aug	4.4	8.5	8.5
Sep	5.3	12.9	10.5
Oct	9.2	12.9	10.5
Nov	9.3	12.9	10.5
Dec	8.3	12.9	10.5
Lower San Antonio Bay:			
Jan	15.7	25.0	22.2
Feb	14.8	25.0	22.2
Mar	17.0	21.5	22.2
Apr	20.4	21.5	22.2
May	17.3	20.7	17.4
Jun	15.7	18.3	17.4
Jul	13.9	19.8	19.8
Aug	14.2	19.8	19.8
Sep	18.9	25.0	22.2
Oct	19.0	25.0	22.2
Nov	18.6	25.0	22.2
Dec	14.7	25.0	22.2
Espiritu Santo Bay:			
Jan	25.9	27.3	26.2
Feb	23.4	27.3	26.2
Mar	26.1	25.9	26.2
Apr	25.6	25.9	26.2
May	24.6	25.6	24.1
Jun	21.1	24.5	24.1
Jul	19.5	25.2	25.2
Aug	19.2	25.2	25.2
Sep	27.3	27.3	26.2
Oct	25.3	27.3	26.2
Nov	24.6	27.3	26.2
Dec	26.4	27.3	26.2

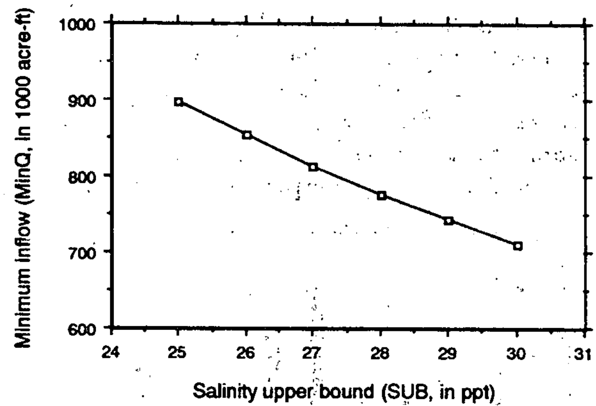

Figure 8.3.7. Minimum inflows (MinQ's) for different salinity upper bounds (*SUB*'s) for lower San Antonio Bay.

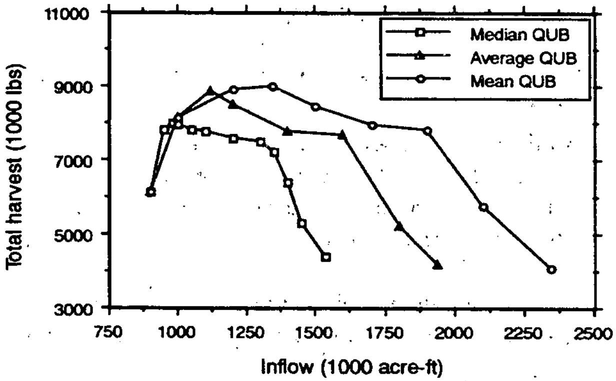

Figure 8.3.8. Total harvest performance curves for different inflow upper bounds (*QUB*'s) for the Guadalupe Estuary.

Effect of the salinity upper bound (SUB). We pointed out earlier that the salinity upper bounds for lower San Antonio Bay is the binding constraint for Case 1. It may be of interest to know how much these constraints influence the solutions, in particular the MinQ solutions. Figure 8.3.7 shows the effect of *SUB* on MinQ, which is almost a linear relationship. MinQ is 0.90 million acre-ft when the *SUB* is 25‰ (base case), but it could be as small as 0.71 million acre-ft if the *SUB* were set to 30‰.

Effect of inflow upper bound (QUB). Figure 8.3.8 shows three performance curves for Case 1 for three different *QUB*'s. The first one is the median *QUB* (base case), the second is the average (of median and mean) *QUB*, and the third is the mean *QUB*. As the *QUB*s are set to larger values, the feasible inflow range is expanded and the total harvest is increased. The peak of the performance curve for median *QUB* occurs at 0.98 million acre-ft of inflow and 8.0 million pounds of total harvest; the peak for the average *QUB* performance curve occurs at 1.12 million acre-ft of inflow and 8.9 million pounds of total harvest; and the peak for the mean *QUB* performance curve occurs at 1.34 million acre-ft of inflow and 9.0 million pounds of total harvest. Note that all three curves start from the same MinQ point because the salinity bounds and the lower bounds on the monthly inflows were held fixed.

Effect of objective function. Figure 8.3.9 shows the performance curve for Case 10 where the objective function is weighted to favor seatrout and brown shrimp harvests. Because the objective function is the sum of the term $p_k H_k$ in which p_k is the weight representing a relative importance of the species and H_k is the harvest, the objective function value can be thought of as an economic or recreational value of the total harvest. The curve depicted in Figure 8.3.9 is

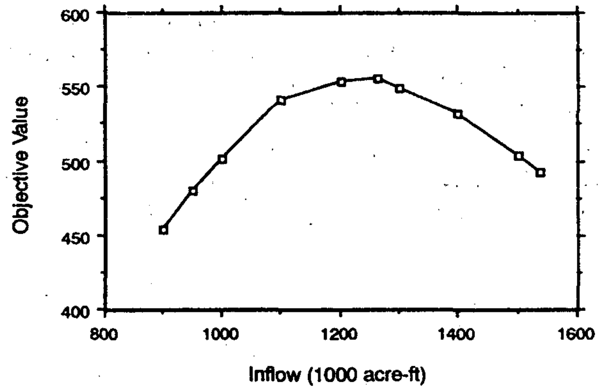

Figure 8.3.9. Objective value performance curve for case 10.

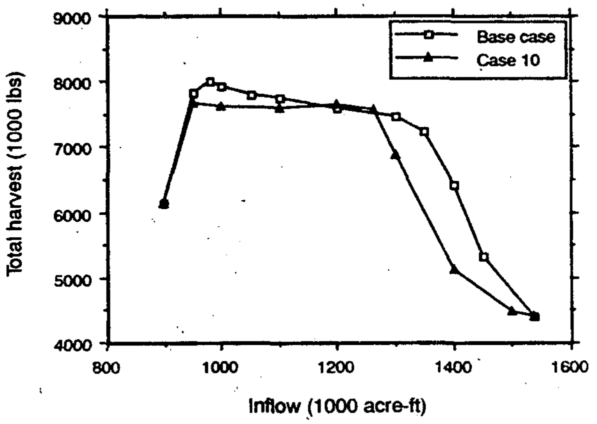

Figure 8.3.10. Total harvest performance curves for the base case and case 10.

smooth and round, and the peak is well defined compared to the performance curves for Case 1 in Figure 8.3.1. The peak occurs at 1.26 million acre-ft and the objective function value is 555 units. Since the weightings are so heavily in favor of brown shrimp (27 times the weighting on blue crab) and spotted seatrout (49 times the weighting on blue crab), this objective function curve shows the effect of inflow on brown shrimp and spotted seatrout that is usually masked by the large blue crab harvest. Figure 8.3.10 shows the total harvest performance curve for Case 10 together with the performance curve for the base case for comparison. These performance curves appear similar but are actually different with regard to individual species harvests (see Table 8.3.1 for maximum harvests of the two cases). The seatrout harvest emphasized in Case 10, which is three times as great as in the base case, does not stand out in the total harvest performance curve because of the dominance (in terms of weight) of the crab harvest.

8.4 SIMULATION OF SALINITY CONDITIONS FOR THE YEAR 1984

TXBLEND Model

An optimization model such as TXEMP is very powerful in examining "what-if" type questions, but it lacks the ability to reflect the estuary's response in detail. A simulation model may be used to show how an estuarine system responds to a selected inflow pattern. TXBLEND is a simulation model developed to study the system's response in terms of water movement and salinity to different inflow, tide, and weather conditions. TXBLEND is an expanded version of the BLEND model developed by Dr. William G. Gray of Notre Dame University. His model was modified by adding input routines for tides, river inflows, winds, evaporation, concentration, and various utilities. BLEND is a depth-averaged, two-dimensional finite element model and employs linear triangular elements.

To demonstrate the capability of the TXBLEND model, San Antonio Bay salinity conditions in 1984 were simulated in real time, after calibrating model parameters using data from other years. 1984 was chosen because most of the input data was readily available, since it had been prepared for the nutrient balance study to represent a very dry year (Section 7.3).

Major input data for TXBLEND includes daily freshwater inflows, bihourly tides, and daily meteorological data including evaporation, precipitation, and wind. Tidal data for 1984 was synthesized by a tide generation program based on 37 harmonic components. Evaporation from the bay was estimated using an empirical equation based on air temperature, dew point temperature, wind speed, and the water surface area (Brandes and Masch 1972). A separate publication will present the TXBLEND model in detail.

Model Calibration

Figure 8.4.1 is a map illustrating the three estuarine systems and the boundaries of the Three-Bay Model which was originally created to study the flow exchanges between the Guadalupe Estuary and its neighboring estuaries, the Lavaca-Colorado Estuary and the Mission-Aransas Estuary. A separate publication will provide details about the calibration of the Three-Bay Model. Figure 8.4.2 is the finite element grid used for the Three-Bay Model. Node 502 represents upper San Antonio Bay, node 523 mid-San Antonio Bay, and node 579 lower San Antonio Bay. Salinities measured and computed at these nodes are presented in figures in this section.

Water movement. Figures 8.4.3 through 8.4.10 are the velocity vectors for every three hours showing how the model simulates the water movements in one tidal cycle on June 28, 1984. These vector plots depict slow movements in open bays and fast movements at flow exchange points with the Gulf of Mexico in such places as the Matagorda Ship Channel at the entrance, Pass Cavallo, Corpus Christi Channel at the entrance, and Lydia Ann Channel. At the eastern boundary of the Guadalupe Estuary, Saluria Bayou and Big Bayou show a fairly strong water movement, as well as at Steamboat Pass and South Pass at the boundary of San Antonio Bay and Espiritu Santo Bay. However, at the western boundary of the estuary, Cedar Dugout and Ayres Dugout do not show as strong water movement.

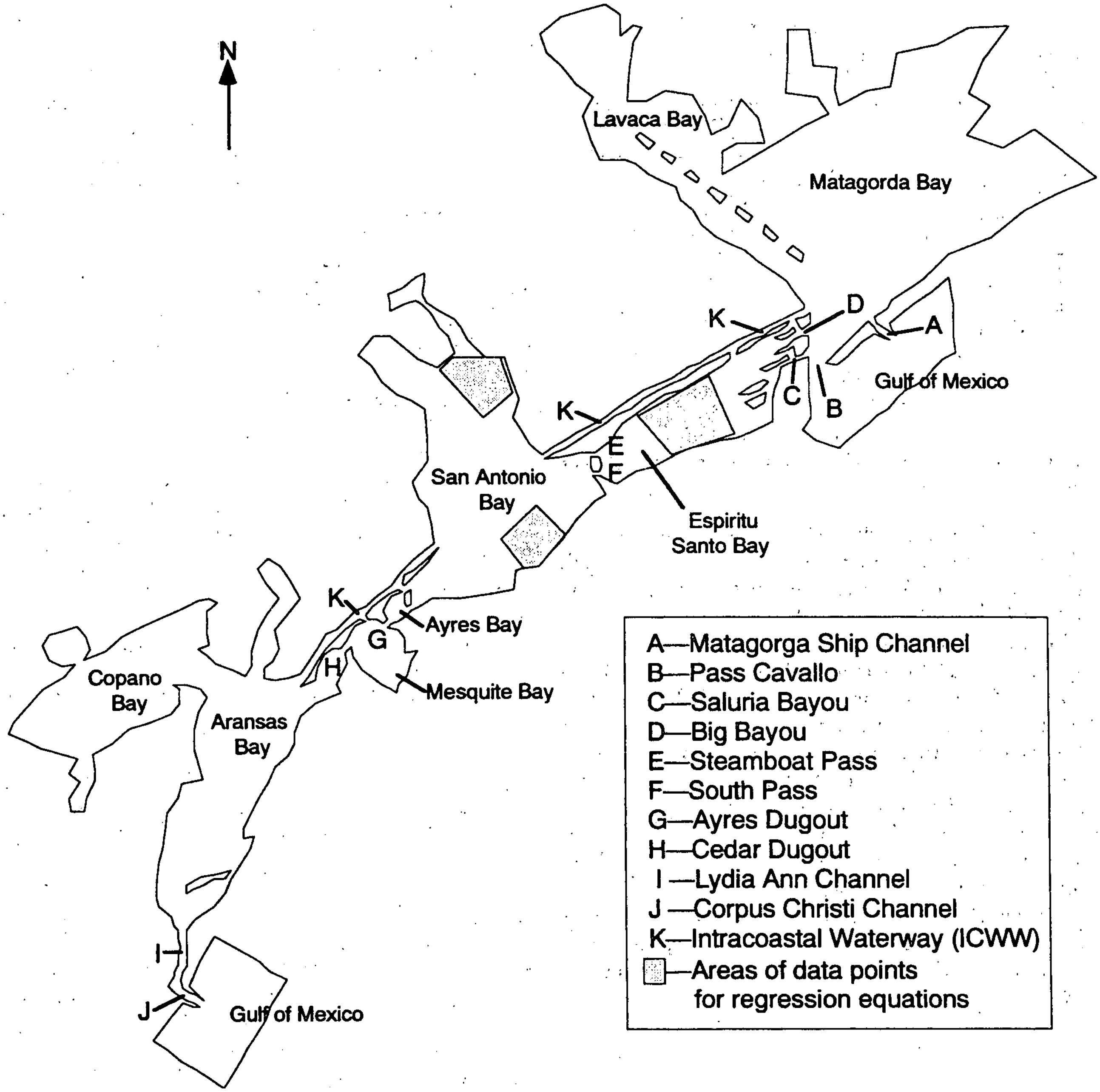

Figure 8.4.1. Boundaries, bays, passes, and channels for the Three-Bay Model of the Matagorda, San Antonio, and Aransas bay systems.

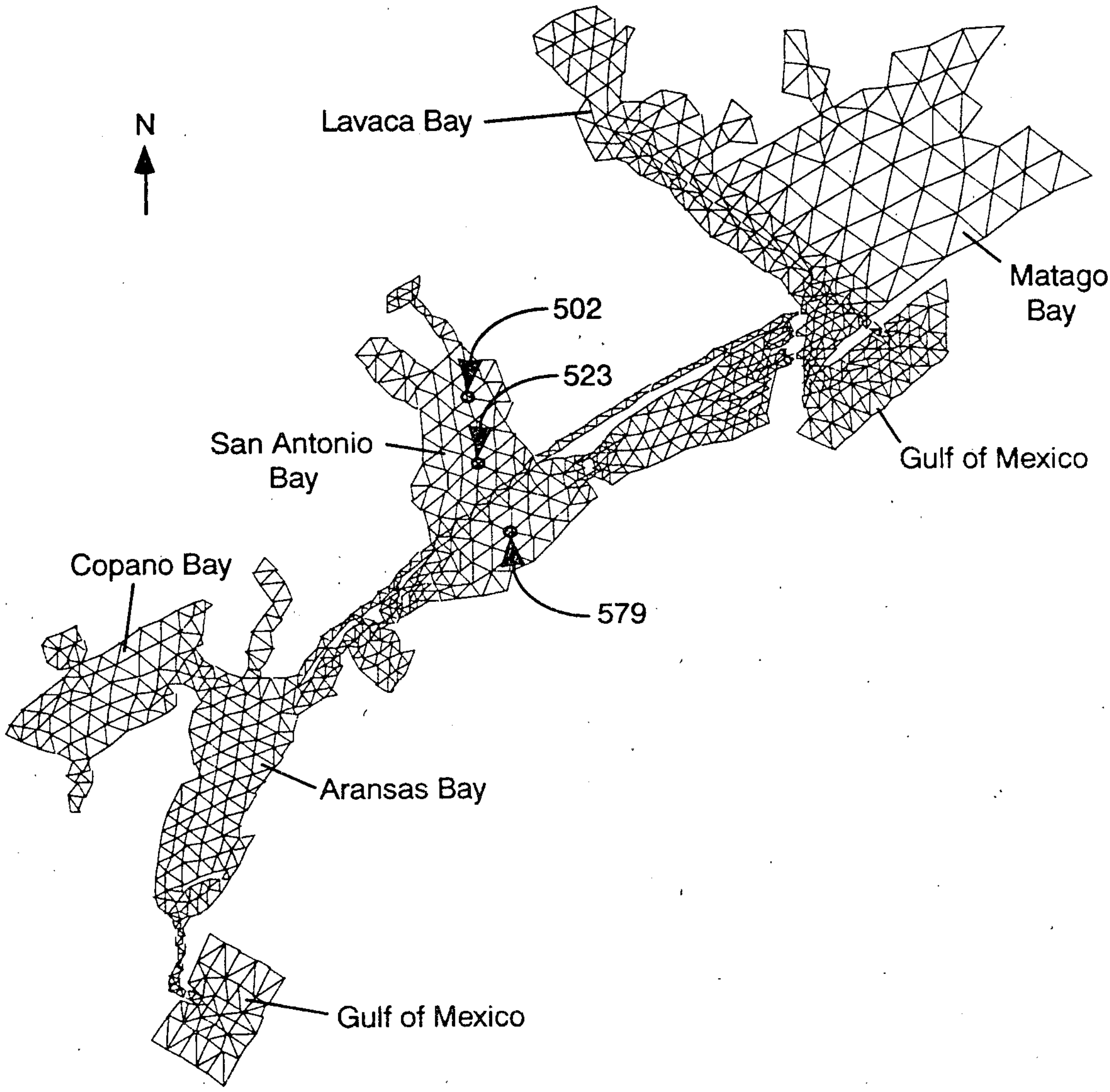

Figure 8.4.2. Computational grid for the Three-Bay Model of the Matagorda, San Antonio, and Aransas bay systems.

Quantitatively, the model estimates 7.3 million acre-ft of water was exchanged (i.e., this volume of water went in and about the same amount went out) annually through the eastern boundary comprising Saluria Bayou, Big Bayou, and the Intracoastal Waterway (ICWW); 4.6 million acre-ft through Steamboat Pass and South Pass; 2.2 million acre-ft through the western boundary comprising Cedar Dugout and the ICWW; and 1.2 million acre-ft through Ayres Dugout and the ICWW, or roughly four times more water through Steamboat Pass and South Pass than through Ayres Dugout and the ICWW.

Salinity. Table 8.4.1 lists the observed salinities and the monthly inflow volumes preceding the observed date. Figures 8.4.11, 8.4.12, and 8.4.13 compare the simulated and observed salinities. The simulation does not necessarily trace all the observed values closely, but the model appears to be performing satisfactorily. Salinity response is difficult to trace closely, in part because it is highly variable. It depends on the timing of the sampling in relation to tidal conditions, the layer from which the sample is taken, and other factors such as meteorological conditions. In spite of some discrepancies, these figures clearly exhibit many of the characteristics of the bay. Lower salinities are found in the upper portion of San Antonio Bay; the salinities increase toward the mid- and lower portions of the bay. Figure 8.4.11 shows the wide and sharp fluctuations in salinity that occur in upper San Antonio Bay. Fluctuations are reduced in Figure 8.4.12 for the mid-portion, and are further reduced in Figure 8.4.13 for the lower portion of the bay.

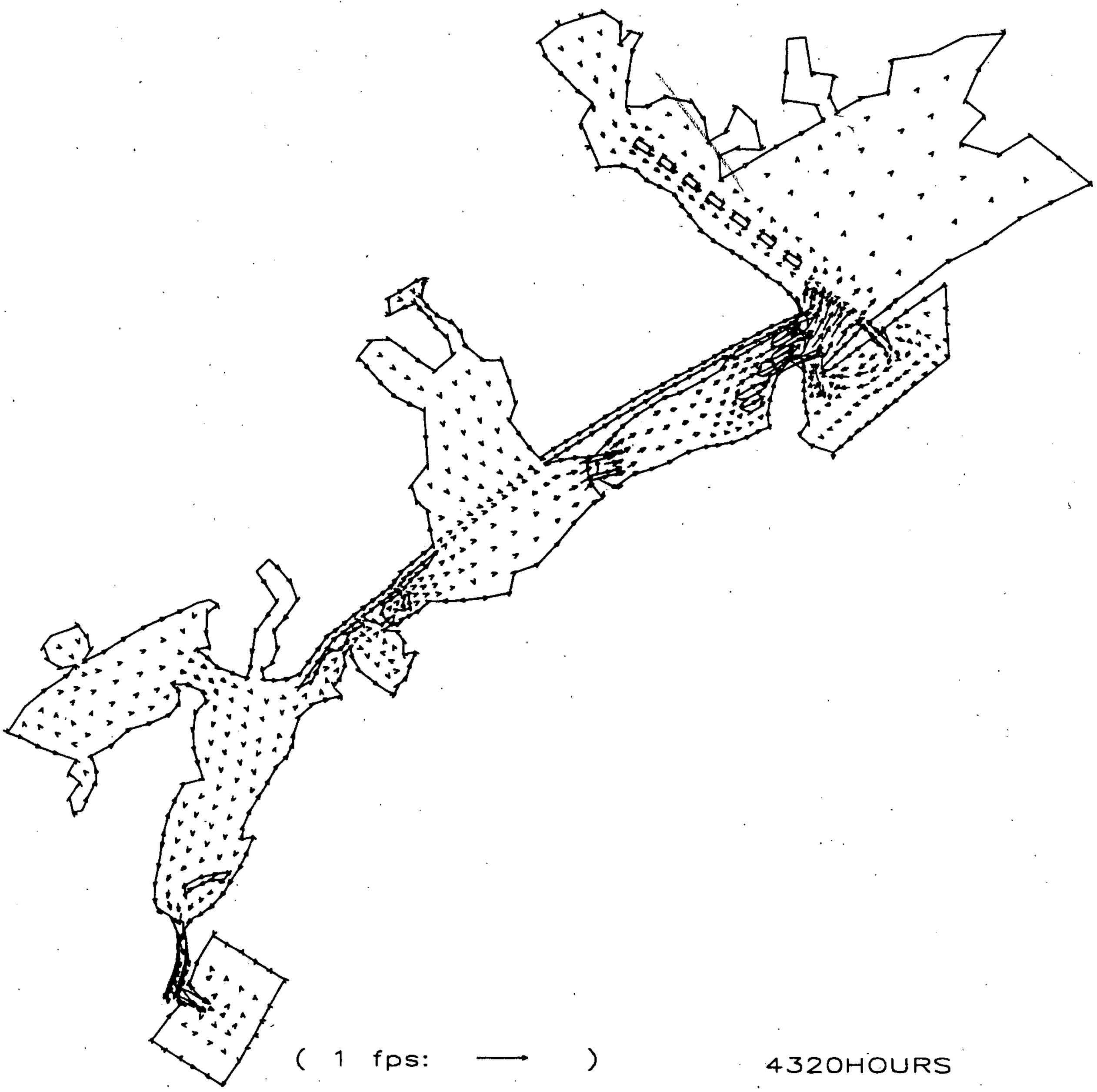

Figure 8.4.3. Simulated velocity vectors for the Three-Bay Model at 0000 hr on June 28, 1984; arrow indicates a velocity of 1 ft/s.

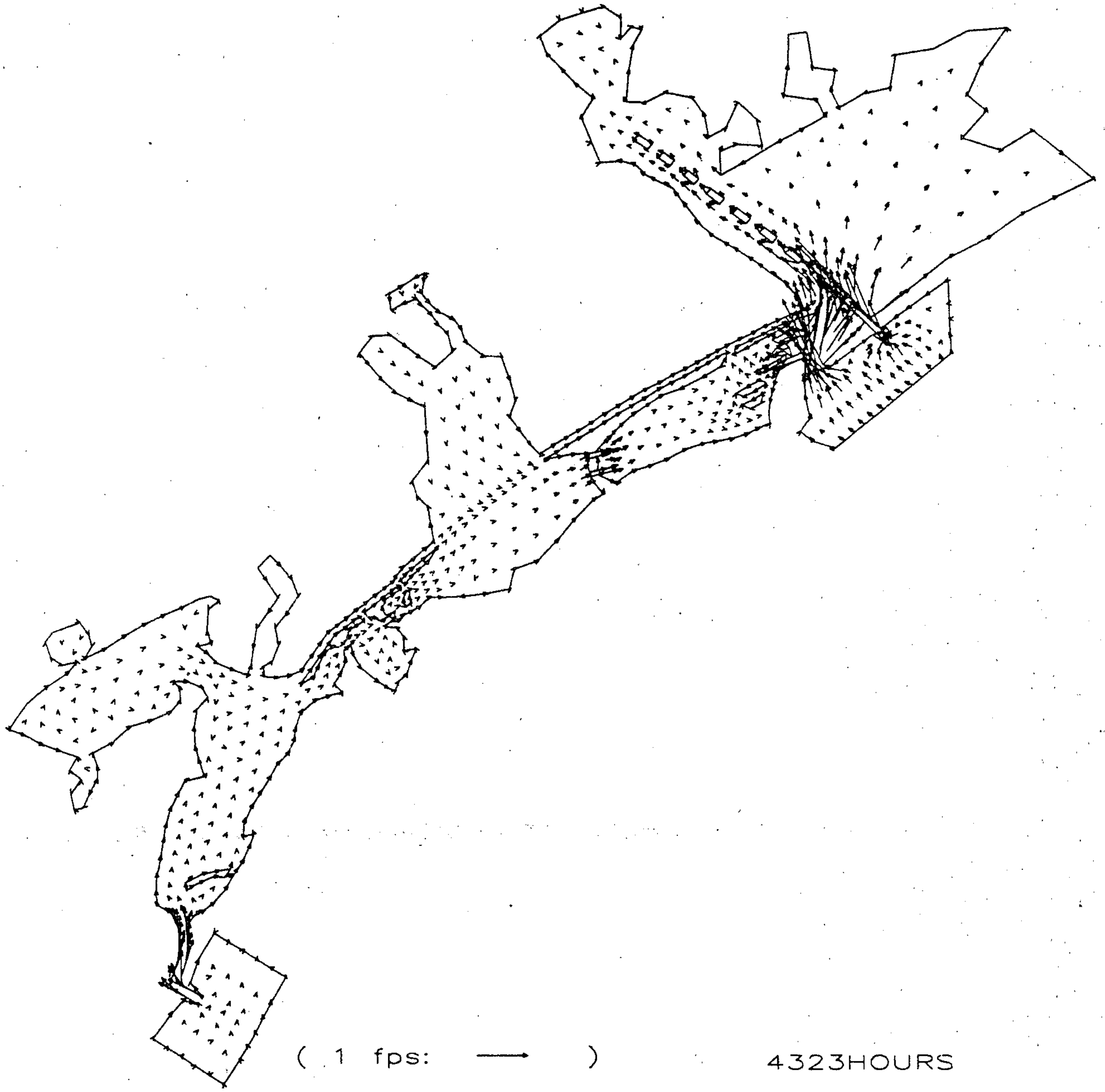

Figure 8.4.4. Simulated velocity vectors for the Three-Bay Model at 0300 hr on June 28, 1984; arrow indicates a velocity of 1 ft/s.

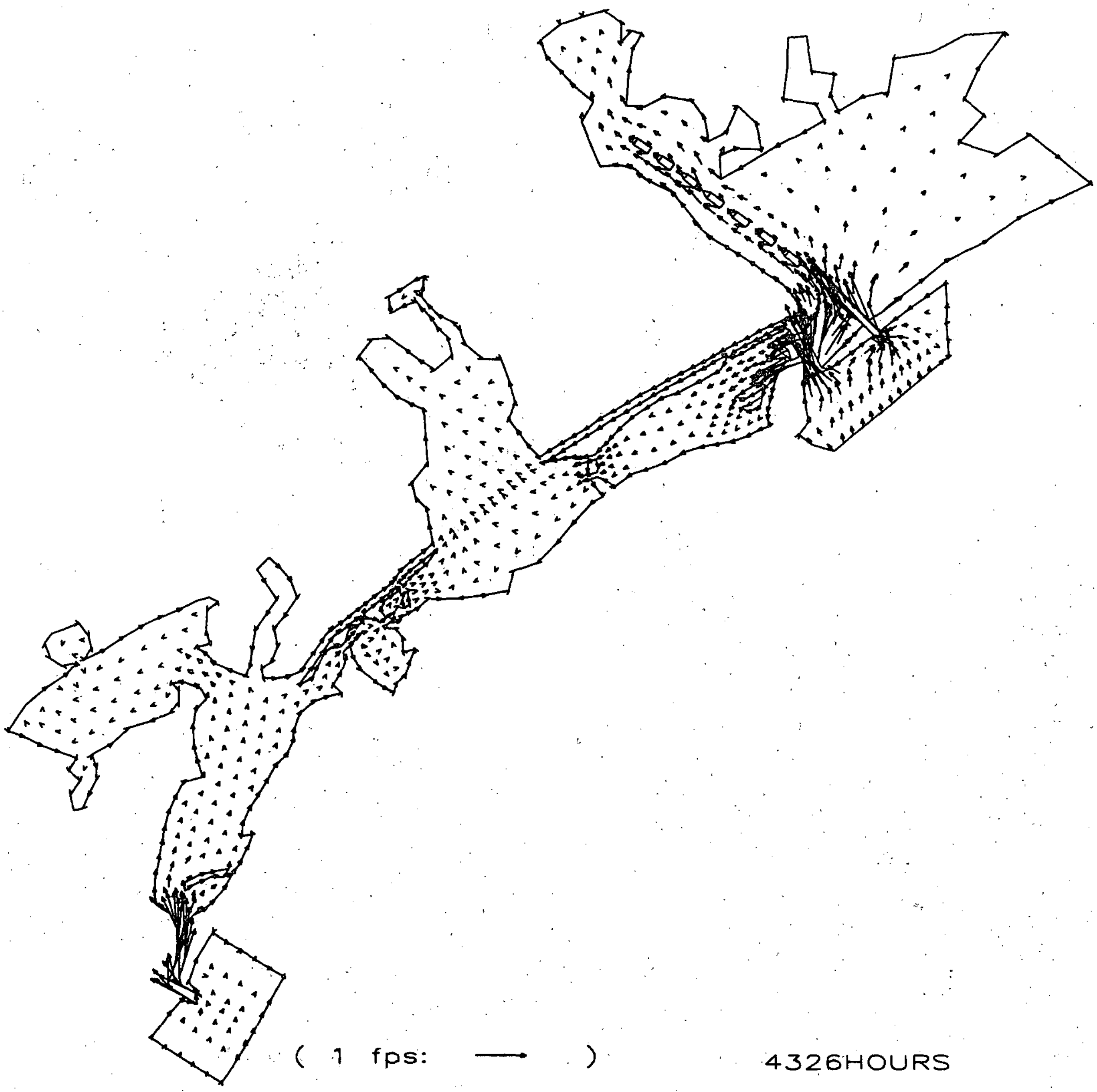

Figure 8.4.5. Simulated velocity vectors for the Three-Bay Model at 0600 hr on June 28, 1984; arrow indicates a velocity of 1 ft/s.

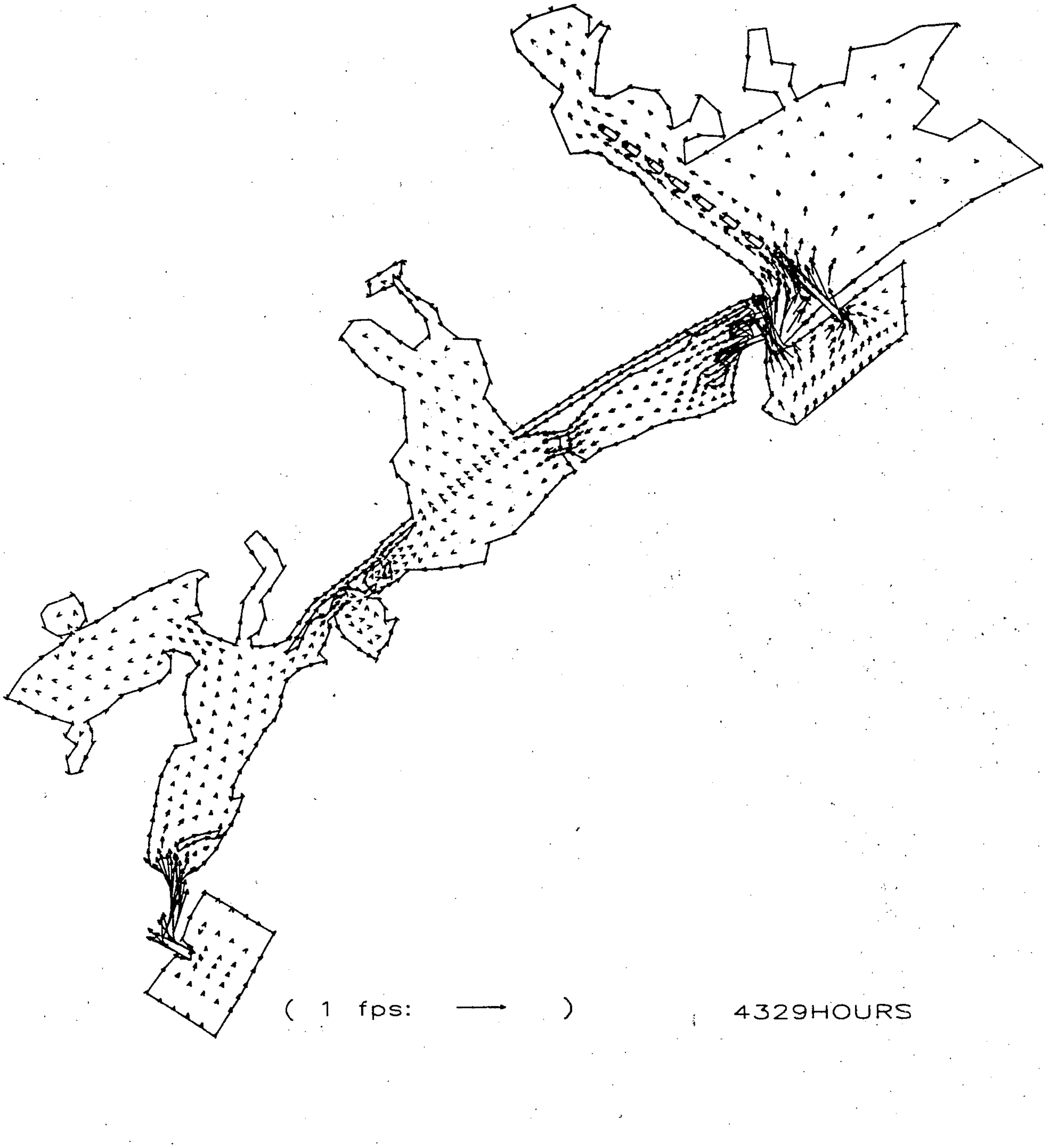

Figure 8.4.6. Simulated velocity vectors for the Three-Bay Model at 0900 hr on June 28, 1984; arrow indicates a velocity of 1 ft/s.

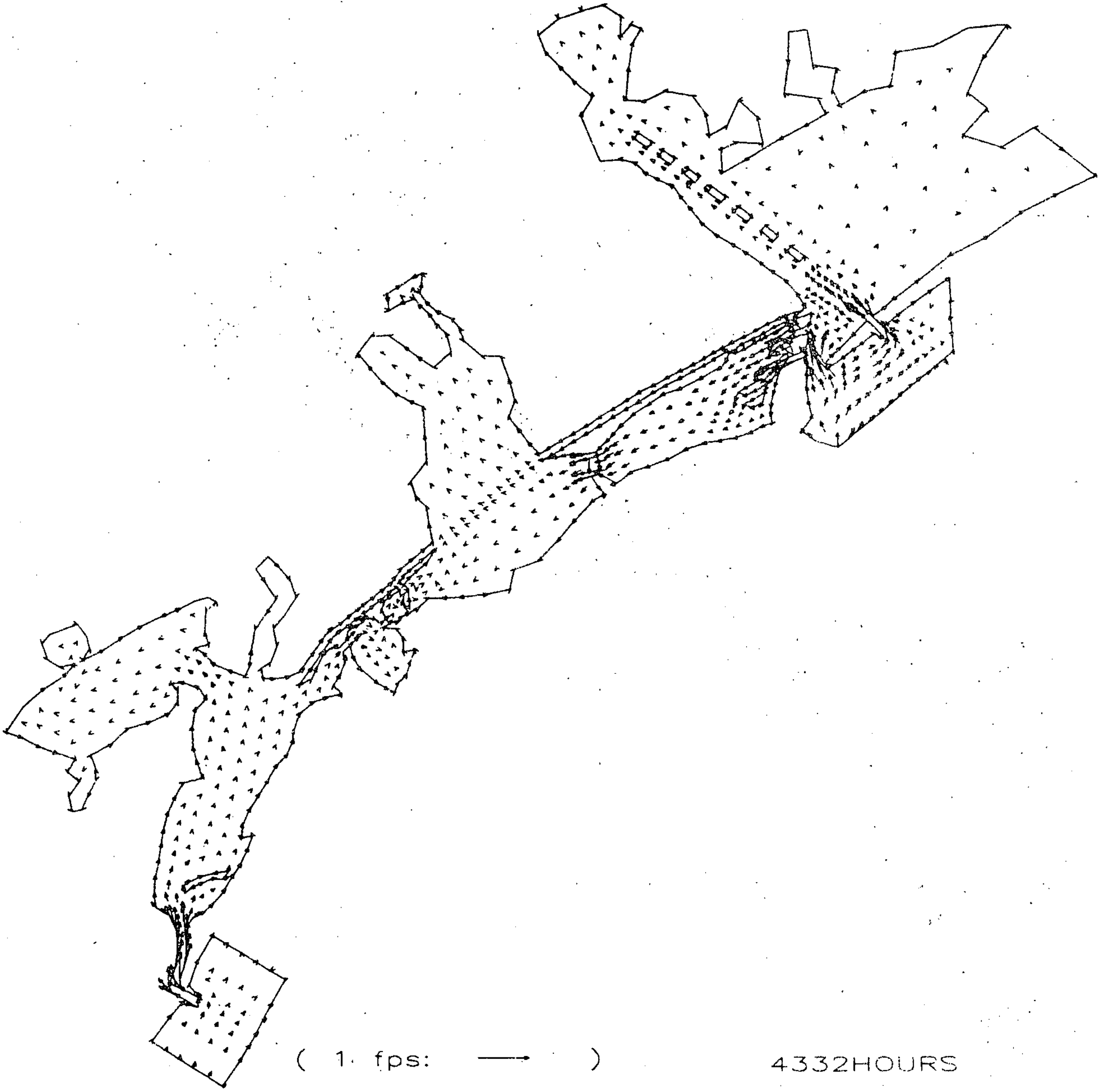

Figure 8.4.7. Simulated velocity vectors for the Three-Bay Model at 1200 hr on June 28, 1984; arrow indicates a velocity of 1 ft/s.

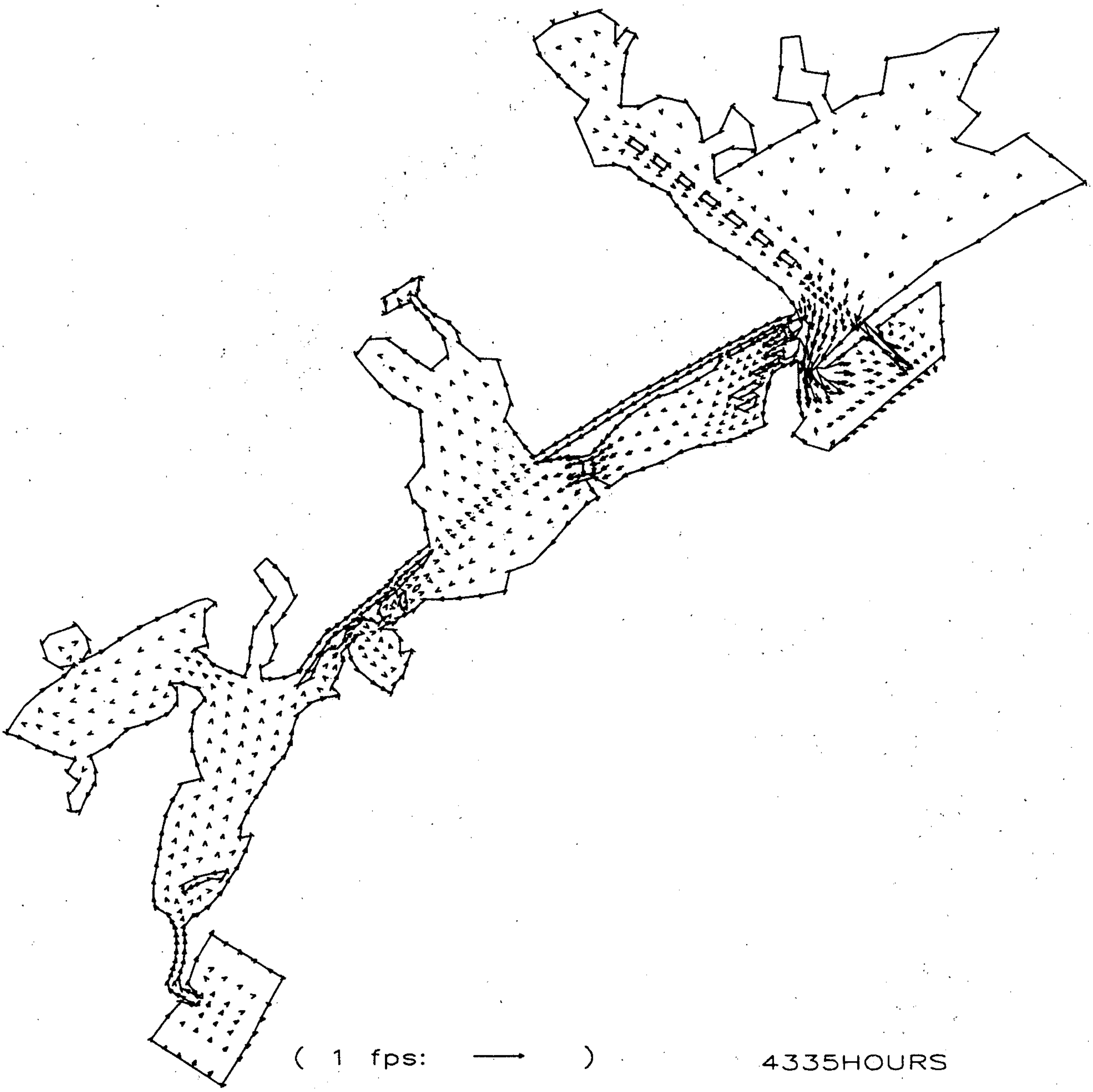

Figure 8.4.8. Simulated velocity vectors for the Three-Bay Model at 1500 hr on June 28, 1984; arrow indicates a velocity of 1 ft/s.

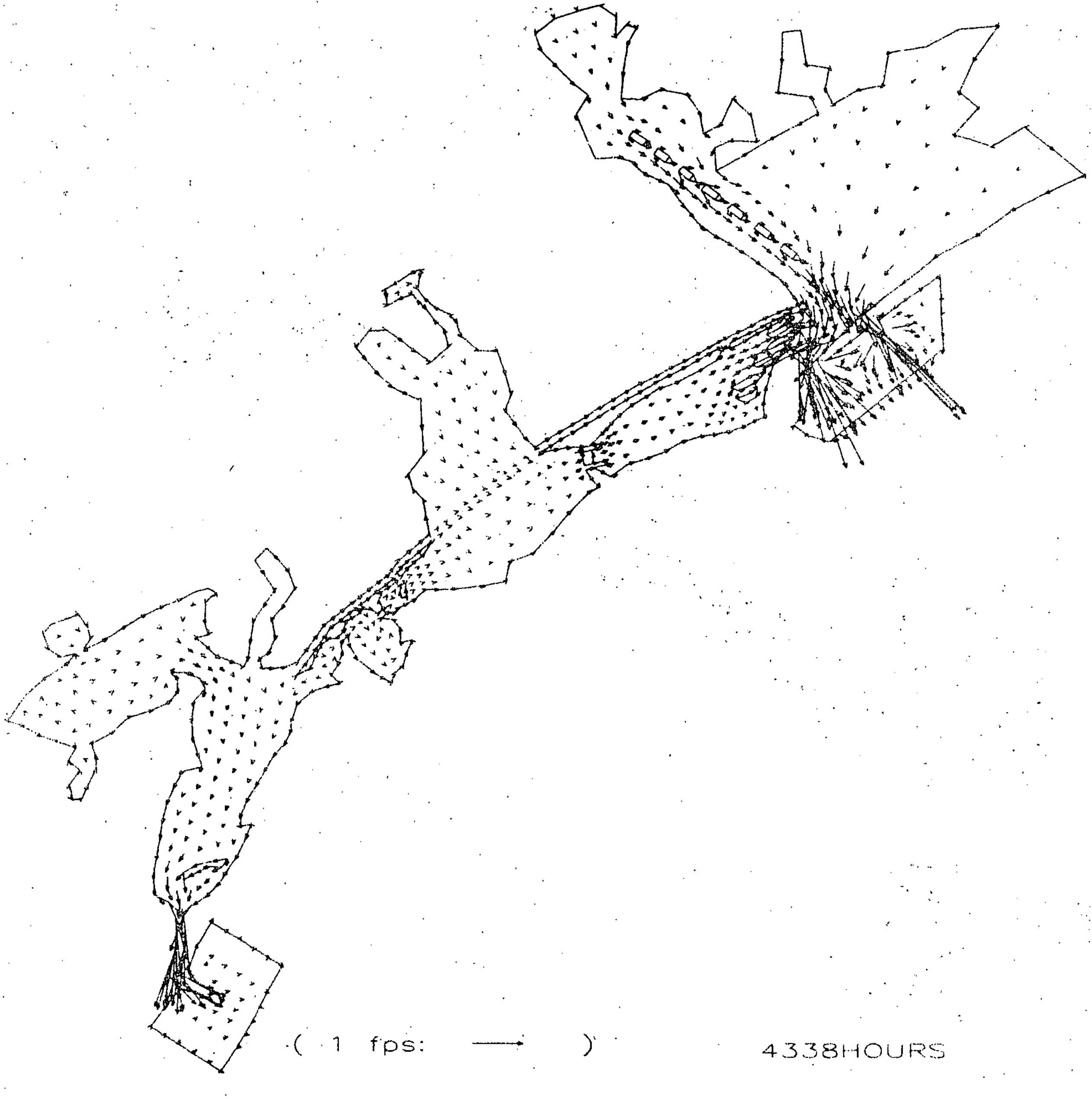

Figure 8.4.9. Simulated velocity vectors for the Three-Bay Model at 1800 hr on June 28, 1984; arrow indicates a velocity of 1 ft/s.

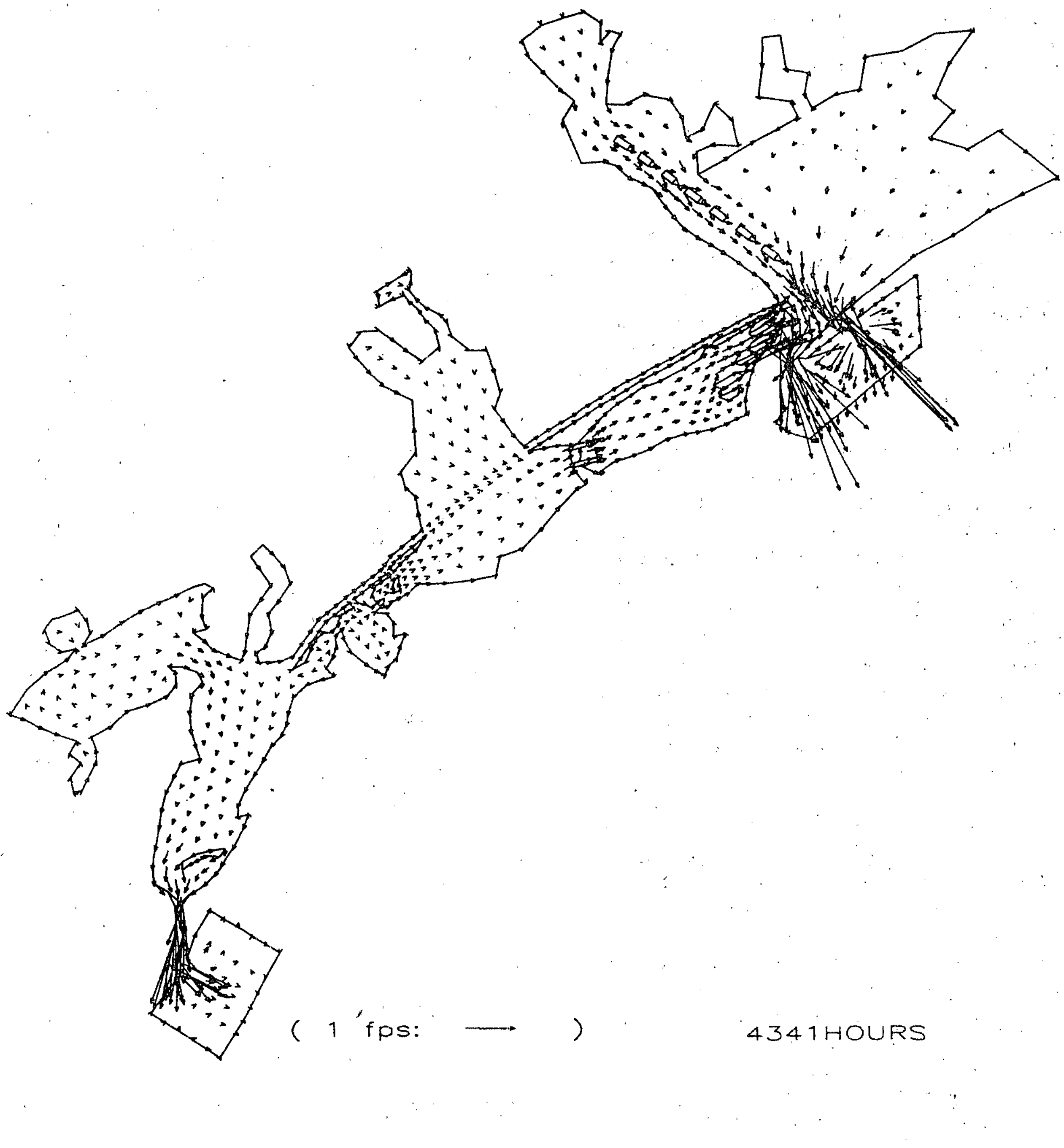

Figure 8.4.10. Simulated velocity vectors for the Three-Bay Model at 2100 hr on June 28, 1984; arrow indicates a velocity of 1 ft/s.

In Figures 8.4.11, 8.4.12, and 8.4.13, the sharp increase in salinity in the first month most likely resulted from cold fronts (see Table 8.4.2 for the meteorological data used as input data from the National Weather Service for Victoria, Texas). Sustained northerly winds probably caused water movement in lower San Antonio Bay from east to west which helped bring in more Gulf water through Saluria Bayou, Big Bayou, Steamboat Pass, and South Pass. As a result, the salinity in all of San Antonio Bay increased rapidly.

For comparison, the 1984 historical monthly inflows, the optimal inflows of the uniform weighting with 50% salinity probability (Case 1), and the MinQ inflows are listed in Table 8.4.3. As can be seen from the table, the summer months of 1984 had very low inflows. This drought condition is well reflected in the simulated salinities in Figures 8.4.11, 8.4.12, and 8.4.13. Significant precipitation came in mid- and late October, causing a sharp drop in salinity. This is also shown in the simulated results in the same figures (see also Table 8.4.2 for October precipitation data).

Salinity pattern. Figures 8.4.14 through 8.4.25 are plots of salinity contours (or isohalines) for the simulated average monthly salinities in 1984. The January average salinity condition shown in Figure 8.4.14 indicates a strong influence by the sea water that came through Steamboat Pass and South Pass into San Antonio Bay. This is evidenced by the fan-shaped isohalines centered around the passes. (Note that water movement is generally perpendicular to the isohalines). In addition, the narrow spacing of isohalines indicates a steep salinity gradient, and the high salinity levels at the passes point out the recent arrival of sea water from the Gulf.

Figures 8.4.15 and 8.4.16 for February and March show milder salinity gradients which may suggest the dispersion of the sea water that arrived in January. Figures 8.4.17 and 8.4.18 for April and May show a rather typical salinity pattern, in sharp contrast with the January condition. The isohalines are lined up in an orderly manner, with the lower salinity contours starting at the upper estuary and gradually increasing toward the lower estuary. Notice that the isohalines change orientation at the "corners" or near Steamboat Pass and South Pass at the eastern side and near Ayres Bay at the western side of San Antonio Bay.

Figures 8.4.19 and 8.4.20 for June and July show the evaporation effect in Mesquite Bay and Ayres Bay. The effect creeps into San Antonio Bay through the southern boundary, probably because the northern boundary has more water movement. The evaporation effect at lower San Antonio Bay appears to be stabilized for August and September as seen in Figures 8.4.21 and 8.4.22, but the salinities are still very high for this portion of the estuary. Figures 8.4.23, 8.4.24, and 8.4.25 for October, November, and

Table 8.4.1. 1984 observed salinities (‰) and monthly inflow volumes (1,000 acre-ft).

Month	Day	Salinity	Inflow
Upper San Antonio Bay:			
Jan	4	13.8	48.8
Feb	15	18.9	66.3
Feb	22	14.1	67.4
Mar	7	16.7	60.0
Apr	4	11.1	86.9
May	23	25.0	38.4
Jul	4	31.1	18.7
Jul	11	28.9	16.1
Jul	18	23.8	11.3
Aug	1	26.6	8.7
Oct	3	32.2	11.5
Oct	10	26.6	17.3
Oct	17	31.1	49.9
Oct	31	19.9	145.0
Nov	14	15.3	163.4
Dec	12	8.0	69.2
Mid-San Antonio Bay:			
Jan	4	11.7	48.8
Feb	8	19.4	90.4
Mar	21	25.5	88.8
May	9	23.9	35.1
May	23	25.0	38.4
Aug	15	23.8	18.9
Sep	12	25.0	17.0
Sep	19	31.1	12.5
Oct	3	32.7	11.5
Oct	31	23.5	145.0
Nov	14	15.5	163.4
Nov	28	22.2	93.4
Dec	12	16.9	69.2
Lower San Antonio Bay:			
Jan	7	16.1	48.6
Jan	21	23.3	78.0
Feb	11	21.6	69.5
Mar	3	20.0	60.3
Mar	10	20.0	59.6
Mar	31	19.4	88.4
Apr	21	24.4	45.6
May	19	27.2	33.0
Jun	2	28.8	38.9
Aug	4	30.5	8.3
Sep	8	30.8	17.7
Sep	29	35.3	11.5
Nov	10	22.0	171.2
Nov	24	25.2	129.9
Dec	8	23.8	71.9

Table 8.4.2. Partial listing of the meteorological input data used for 1984 simulation, precipitation(inches), evaporation (inches), wind speed (mph), and wind direction (degrees). Mon = month, Prec = precipitation, W.S. = wind speed, W.D. = wind direction.

Mon	Day	Prec	Evap	W.S.	W.D.	Mon	Day	Prec	Evap	W.S.	W.D.	Mon	Day	Prec	Evap	W.S.	W.D.
1	1	0.00	0.11	9.9	104	6	1	0.00	0.33	10.8	146	9	1	0.34	0.34	10.3	109
1	2	0.00	0.13	11.2	44	6	2	0.00	0.30	9.8	140	9	2	1.03	0.07	8.7	99
1	3	0.00	0.07	8.2	33	6	3	0.00	0.28	11.1	126	9	3	0.55	0.07	6.9	91
1	4	0.00	0.08	4.5	234	6	4	0.00	0.29	12.6	146	9	4	0.00	0.22	9.5	34
1	5	0.00	0.07	5.9	274	6	5	0.00	0.30	14.4	157	9	5	0.00	0.37	11.8	52
1	6	0.00	0.08	6.0	233	6	6	0.11	0.31	14.2	152	9	6	0.00	0.34	10.7	80
1	7	0.00	0.21	9.9	72	6	7	0.00	0.38	15.7	155	9	7	0.31	0.28	10.8	125
1	8	0.19	0.09	12.9	88	6	8	0.00	0.39	14.8	148	9	8	0.00	0.39	12.3	165
1	9	2.45	0.10	12.8	352	6	9	0.00	0.30	13.1	146	9	9	0.00	0.35	11.3	169
1	10	0.00	0.17	22.9	346	6	10	0.06	0.32	11.4	124	9	10	0.00	0.36	11.6	169
1	11	0.00	0.04	7.2	31	6	11	0.03	0.23	9.8	131	9	11	0.01	0.40	11.4	150
1	12	0.00	0.05	7.3	15	6	12	0.02	0.31	12.1	131	9	12	0.00	0.29	8.8	146
1	13	0.00	0.12	14.3	9	6	13	0.00	0.29	10.3	127	9	13	0.00	0.26	8.3	127
1	14	0.16	0.05	14.2	3	6	14	0.00	0.29	8.9	111	9	14	0.00	0.24	6.9	106
1	15	0.01	0.03	13.0	3	6	15	0.00	0.26	8.3	130	9	15	0.30	0.18	8.7	10
1	16	0.00	0.03	10.4	12	6	16	0.00	0.31	9.8	135	9	16	0.25	0.21	11.8	28
1	17	0.00	0.02	11.8	28	6	17	0.00	0.33	9.9	141	9	17	0.00	0.31	10.6	33
1	18	0.24	0.06	18.9	1	6	18	0.00	0.32	9.6	137	9	18	0.00	0.26	9.4	32
1	19	0.00	0.06	12.2	16	6	19	0.00	0.35	9.7	153	9	19	0.00	0.31	10.4	26
1	20	0.00	0.11	15.2	25	6	20	0.00	0.31	9.4	143	9	20	0.12	0.18	11.0	21
1	21	0.00	0.08	9.8	47	6	21	0.00	0.34	8.9	157	9	21	0.63	0.06	10.4	3
1	22	0.14	0.03	9.8	86	6	22	0.00	0.34	8.9	165	9	22	0.00	0.13	8.2	183
1	23	0.39	0.01	3.9	124	6	23	0.00	0.34	8.7	174	9	23	0.00	0.20	9.5	132
1	24	1.05	0.01	10.4	26	6	24	0.00	0.24	6.6	191	9	24	0.00	0.20	10.8	136
1	25	0.32	0.07	11.1	15	6	25	0.00	0.34	8.6	201	9	25	0.00	0.22	9.2	122
1	26	0.00	0.04	5.2	38	6	26	0.04	0.37	8.8	180	9	26	0.00	0.19	10.9	11
1	27	0.00	0.10	7.3	319	6	27	0.00	0.38	9.8	175	9	27	0.00	0.14	9.8	37
1	28	0.00	0.12	6.0	237	6	28	0.00	0.29	7.3	167	9	28	0.00	0.19	12.4	22
1	29	0.00	0.11	8.1	201	6	29	0.00	0.26	7.7	177	9	29	0.06	0.12	13.5	6
1	30	0.02	0.17	12.1	9	6	30	0.00	0.40	10.4	165	9	30	0.02	0.19	10.1	359
1	31	0.05	0.07	7.9	349	7	1	0.00	0.23	7.7	174	10	1	0.00	0.17	7.9	78
2	1	0.00	0.05	5.9	213	7	2	0.02	0.36	9.8	162	10	2	0.00	0.17	8.1	96
2	2	0.28	0.02	6.3	14	7	3	0.00	0.46	11.5	157	10	3	0.00	0.23	9.6	107
2	3	0.03	0.11	9.1	349	7	4	0.00	0.51	11.8	170	10	4	0.00	0.17	9.9	120
2	4	0.00	0.10	5.4	333	7	5	0.00	0.42	9.2	162	10	5	0.01	0.25	11.6	160
2	5	0.00	0.22	11.1	334	7	6	0.00	0.38	8.6	181	10	6	0.00	0.34	13.0	164
2	6	0.00	0.14	11.0	35	7	7	0.00	0.28	7.8	153	10	7	0.01	0.06	7.3	69
2	7	0.00	0.09	8.1	112	7	8	0.00	0.40	8.7	150	10	8	0.22	0.12	7.4	127
2	8	0.04	0.03	10.3	96	7	9	0.00	0.39	9.6	164	10	9	0.46	0.10	7.9	103
2	9	0.04	0.12	9.4	192	7	10	0.00	0.49	11.4	163	10	10	0.01	0.11	7.4	100
2	10	0.00	0.14	15.1	180	7	11	0.00	0.40	9.7	140	10	11	0.12	0.14	8.7	123
2	11	0.01	0.12	15.8	165	7	12	0.01	0.42	8.9	174	10	12	0.00	0.22	9.4	149
2	12	0.04	0.20	12.5	311	7	13	0.00	0.39	8.1	165	10	13	0.04	0.09	9.4	126
2	13	0.00	0.18	7.4	200	7	14	0.06	0.37	8.5	162	10	14	1.30	0.13	8.0	176
2	14	0.00	0.17	10.6	149	7	15	0.01	0.42	9.1	169	10	15	0.00	0.26	12.4	177
2	15	0.05	0.12	10.1	177	7	16	0.00	0.48	10.1	187	10	16	0.00	0.23	11.9	167
2	16	0.00	0.14	6.8	35	7	17	0.00	0.49	11.3	197	10	17	0.07	0.22	11.9	64
2	17	0.00	0.11	12.9	140	7	18	0.00	0.41	10.6	201	10	18	0.00	0.31	16.1	164
2	18	0.08	0.12	13.5	130	7	19	0.00	0.28	7.8	167	10	19	3.51	0.14	11.9	129
2	19	0.00	0.20	15.8	25	7	20	0.00	0.22	7.6	159	10	20	0.11	0.27	14.2	168
2	20	0.31	0.06	16.6	5	7	21	0.02	0.33	9.4	109	10	21	0.00	0.16	13.7	122
2	21	0.01	0.10	8.1	333	7	22	1.11	0.26	8.6	108	10	22	0.56	0.08	13.9	14
2	22	0.00	0.12	7.1	228	7	23	0.00	0.38	9.2	93	10	23	0.06	0.07	13.7	9
2	23	0.00	0.16	9.2	175	7	24	0.00	0.22	9.0	49	10	24	0.01	0.04	8.1	16
2	24	0.00	0.17	8.7	36	7	25	0.47	0.23	7.3	119	10	25	0.44	0.08	9.0	64
2	25	0.00	0.20	13.5	138	7	26	0.17	0.27	8.4	119	10	26	3.45	0.11	9.1	133
2	26	0.02	0.48	21.8	242	7	27	0.02	0.28	7.5	132	10	27	0.05	0.13	9.0	155
2	27	0.00	0.32	23.1	321	7	28	0.47	0.14	7.6	53	10	28	0.00	0.17	8.5	175
2	28	0.00	0.20	13.4	333	7	29	0.01	0.38	9.1	26	10	29	0.00	0.11	6.3	104
2	29	0.00	0.08	7.3	46	7	30	0.00	0.44	10.7	35	10	30	0.01	0.12	7.6	87
						7	31	0.00	0.37	8.8	58	10	31	0.00	0.18	10.3	134

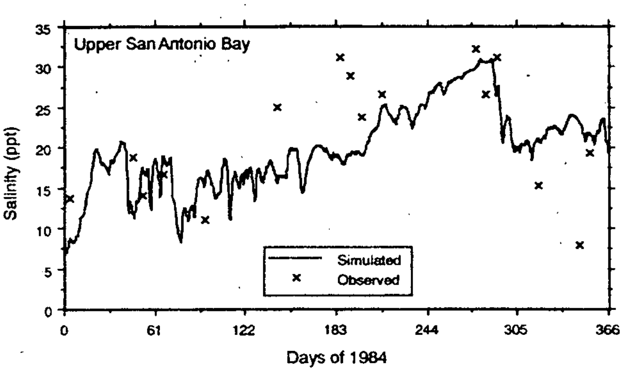

Figure 8.4.11. Simulated and observed salinities in upper San Antonio Bay.

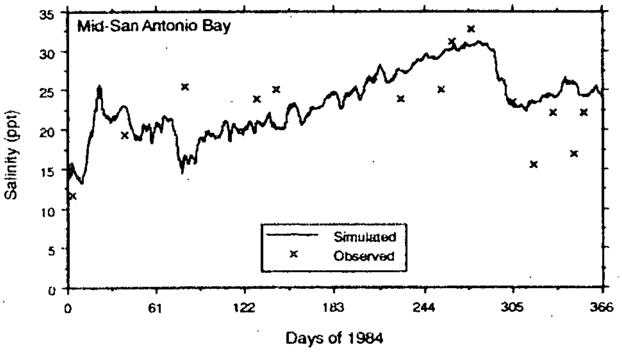

Figure 8.4.12. Simulated and observed salinities in mid-San Antonio Bay.

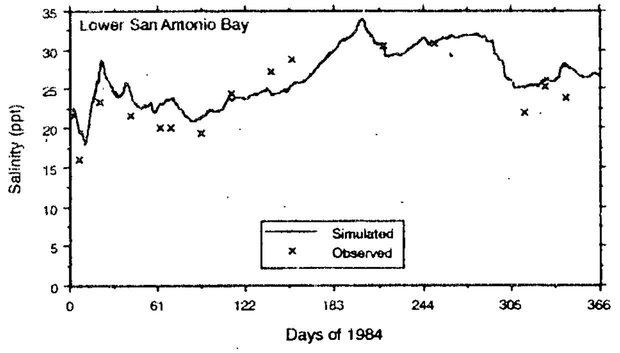

Figure 8.4.13. Simulated and observed salinities in lower San Antonio Bay.

December, respectively, show decreased salinities compared to those of summer months.

Figures 8.4.14 through 8.4.25 also show the location of the areas from which the salinity-inflow regression equations were taken. The isohalines in these areas can be compared with the salinity bounds from the TXEMP to judge how well the computed inflows agree with the simulated salinities from the TXBLEND model. The ranges of salinities for each month and each area are shown in Table 8.4.4. There are two months in the upper San Antonio Bay area where the simulated salinities exceed the salinity upper bound throughout the area (September and October). In lower San Antonio Bay, the salinity upper bounds throughout the area are exceeded for six months, although the largest violations occur from July through October.

There are several reasons why the simulated salinities may exceed the salinity upper bounds. First, the inflow-salinity equations that were used in the TXEMP are very simplified; they represent the middle range of inflows and salinities better than the more extreme values. For example, for the upper San Antonio Bay equation, only about 25 of the 359 data points were taken at salinities equal to or greater than 20‰ or at inflows of less than 50,000 acre-ft/month. Consequently, this equation may understate the salinity levels that occur at low inflows. In actual use, it may be preferable to use another form of the regression equation that better represents the relationship between low inflow and high salinity to provide a more accurate analysis.

A second reason involves the 1984 input conditions used with the TXBLEND simulation. The year 1984 was extremely dry in this basin. Annual evaporation at the bay was the highest recorded between 1977 and 1987, and direct precipitation on the bay was the second lowest recorded over the same time period. These conditions may have biased the simulated salinities upward compared to more typical, average weather conditions. Furthermore, the solution from the TXEMP does not guarantee that the salinity bounds will be observed 100% of the time. The highest probability of satisfying the bounds is 63.6%, as indicated by the MaxSalP. This test case was obtained under a more relaxed 50% SalP, so it is not surprising that simulated salinities exceed the bounds during some months in parts of the estuary. The results do suggest the desirability of reexamining the inflow-salinity equations used in the TXEMP Model and the tidal and weather inputs to the TXBLEND model to refine the relationships used in this example analysis before it becomes operational.

Simulated Salinity Conditions for Optimal Inflows and Minimum Historical Inflows

Figures 8.4.26, 8.4.27, and 8.4.28 show the comparisons of simulated salinities, one for the 1984 historical inflows and the other for the optimal inflows (or MaxH solution) of the uniform weighting case with 50% salinity probability. Interestingly, the optimal inflow case has a very similar pattern to the historical case. The difference is an overall drop in salinity for optimal inflows during the summer months, which is most likely a cumulative effect of the inflows during the spring and summer months (Table 8.4.3). The effect of the increase in freshwater inflows

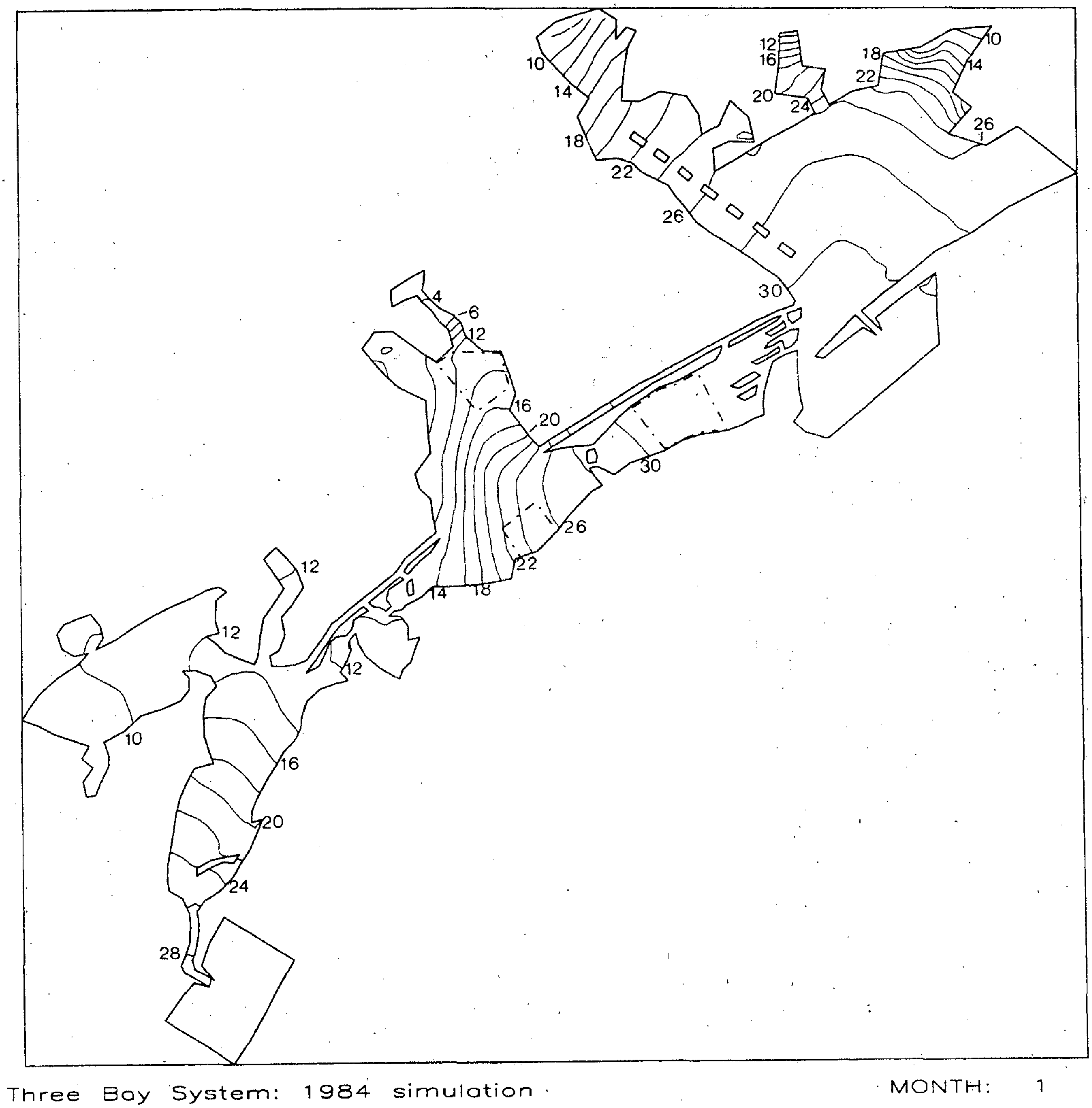

Figure 8.4.14. Isohalines of the simulated salinity from the Three-Bay Model for January 1984.

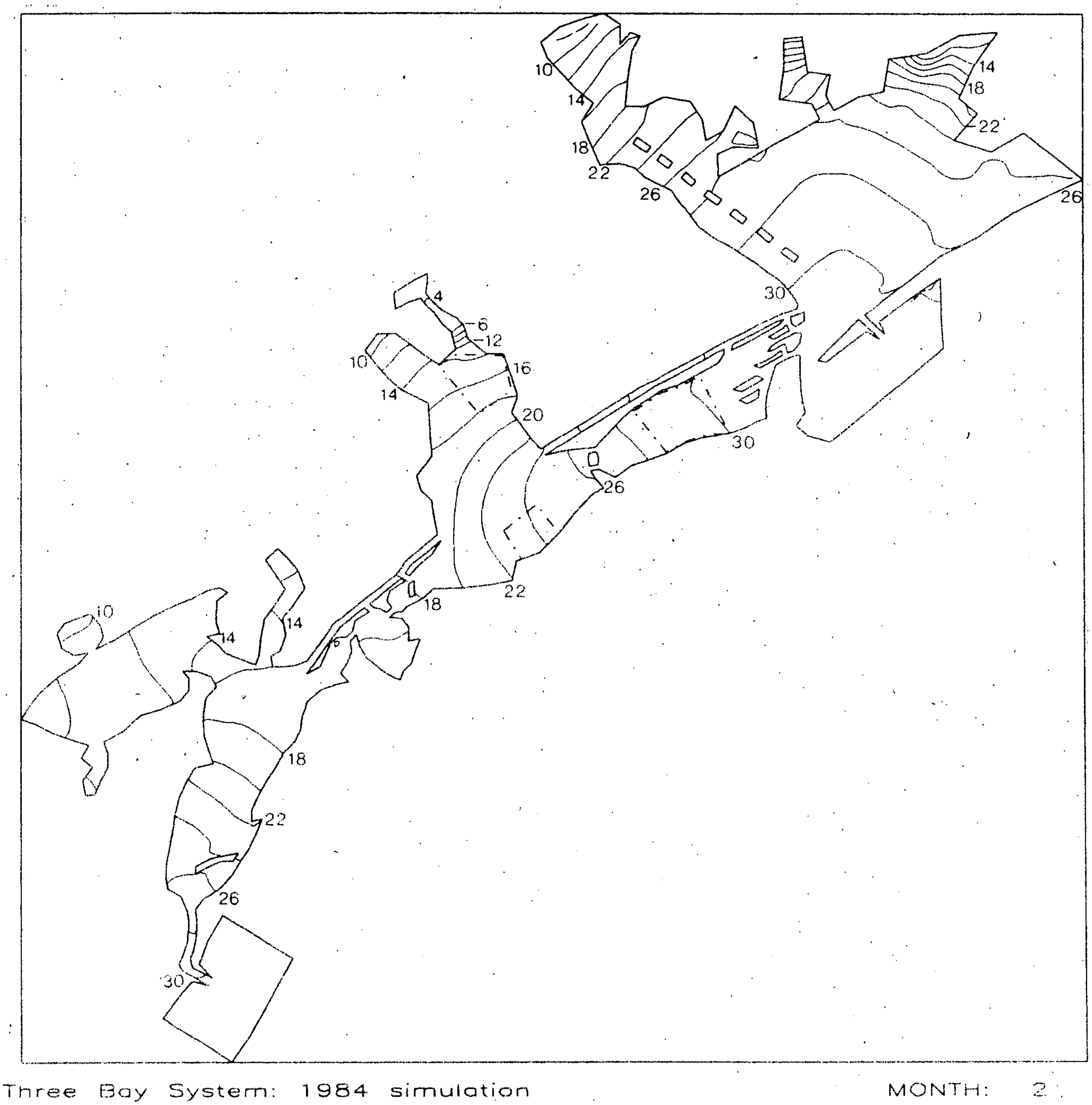

Figure 8.4.15. Isohalines of the simulated salinity from the Three-Bay Model for February 1984.

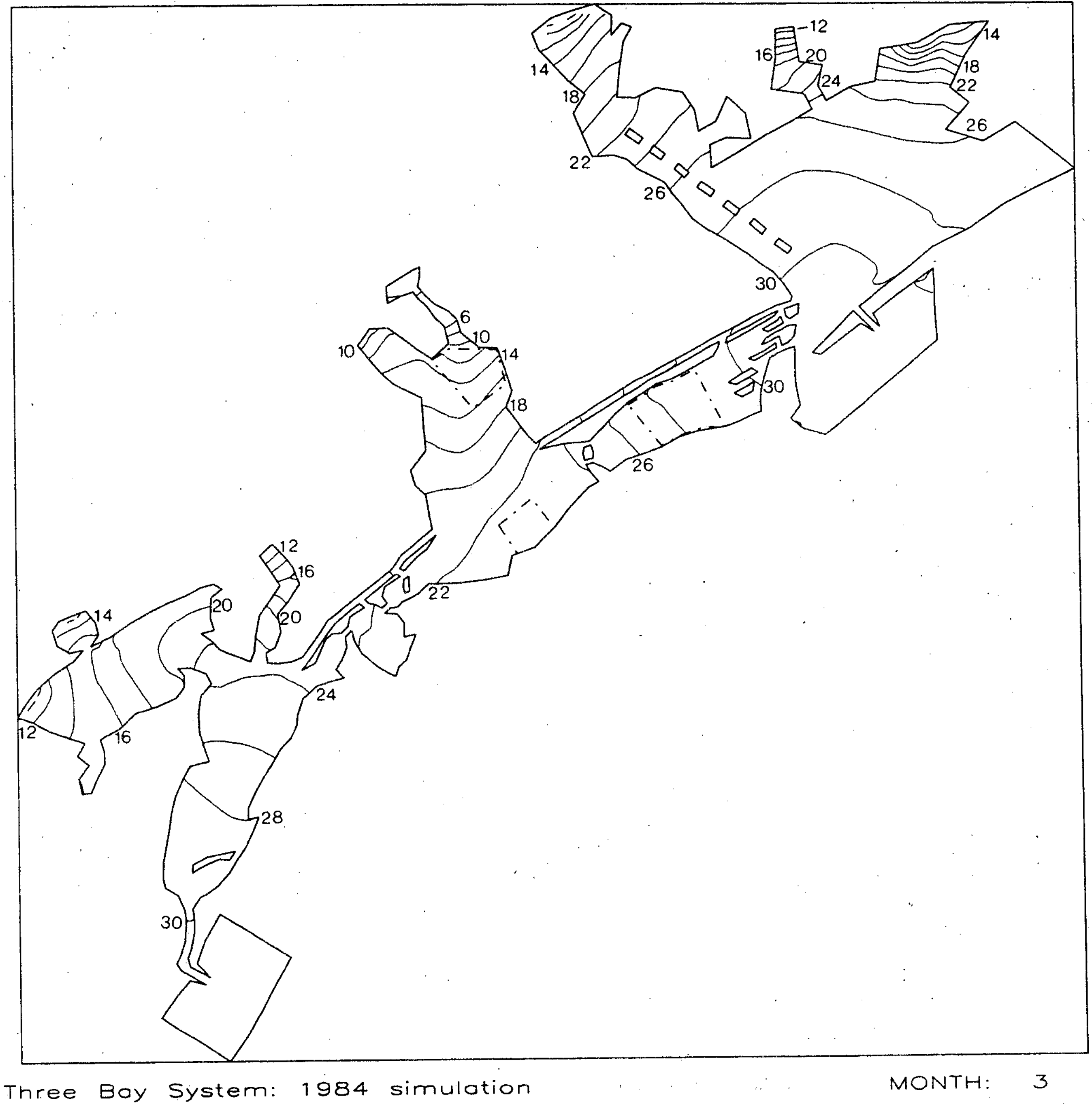

Figure 8.4.16. Isohalines of the simulated salinity from the Three-Bay Model for March 1984.

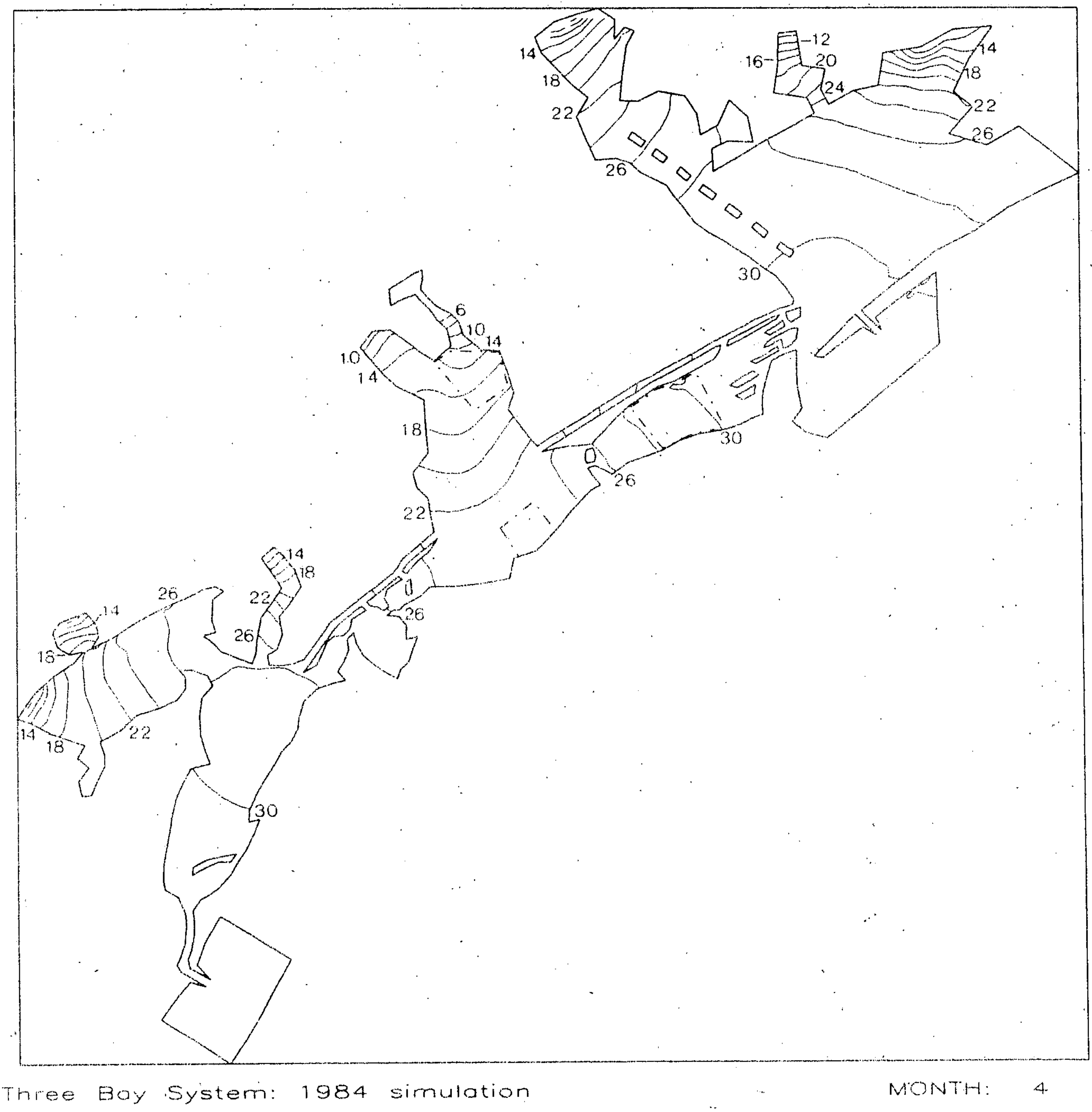

Figure 8.4.17. Isohalines of the simulated salinity from the Three-Bay Model for April 1984.

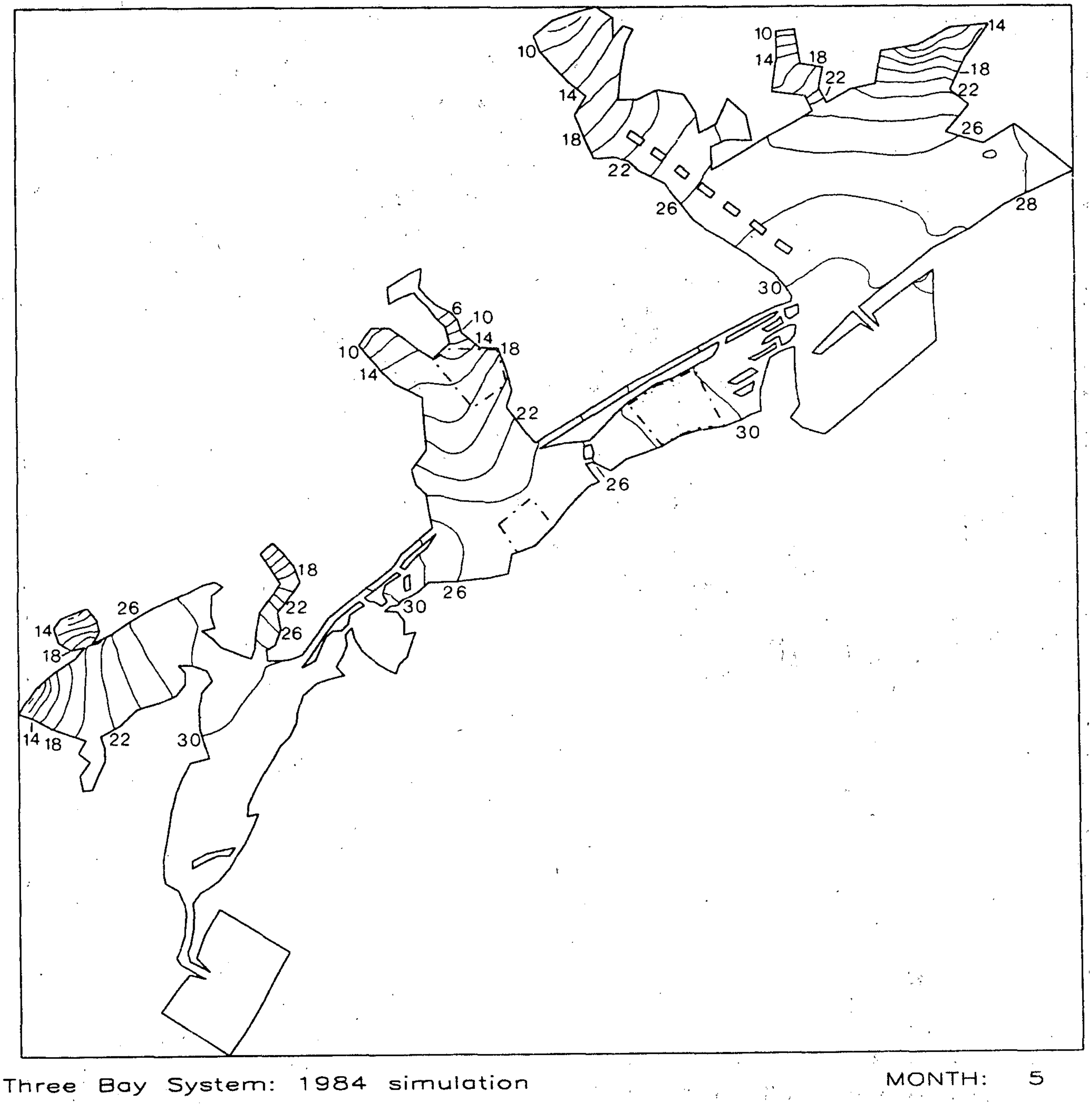

Figure 8.4.18. Isohalines of the simulated salinity from the Three-Bay Model for May 1984.

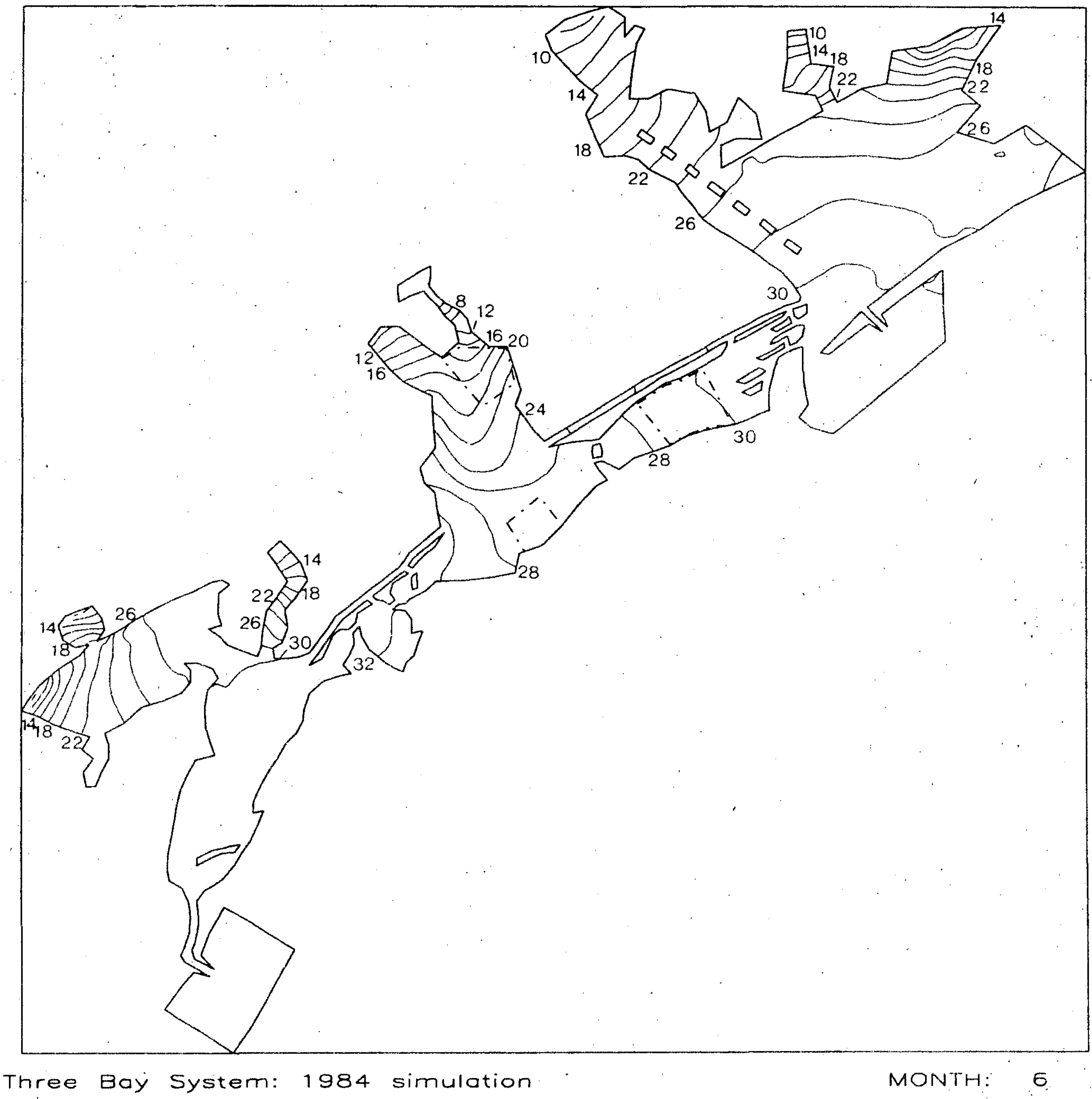

Figure 8.4.19. Isohalines of the simulated salinity from the Three-Bay Model for June 1984.

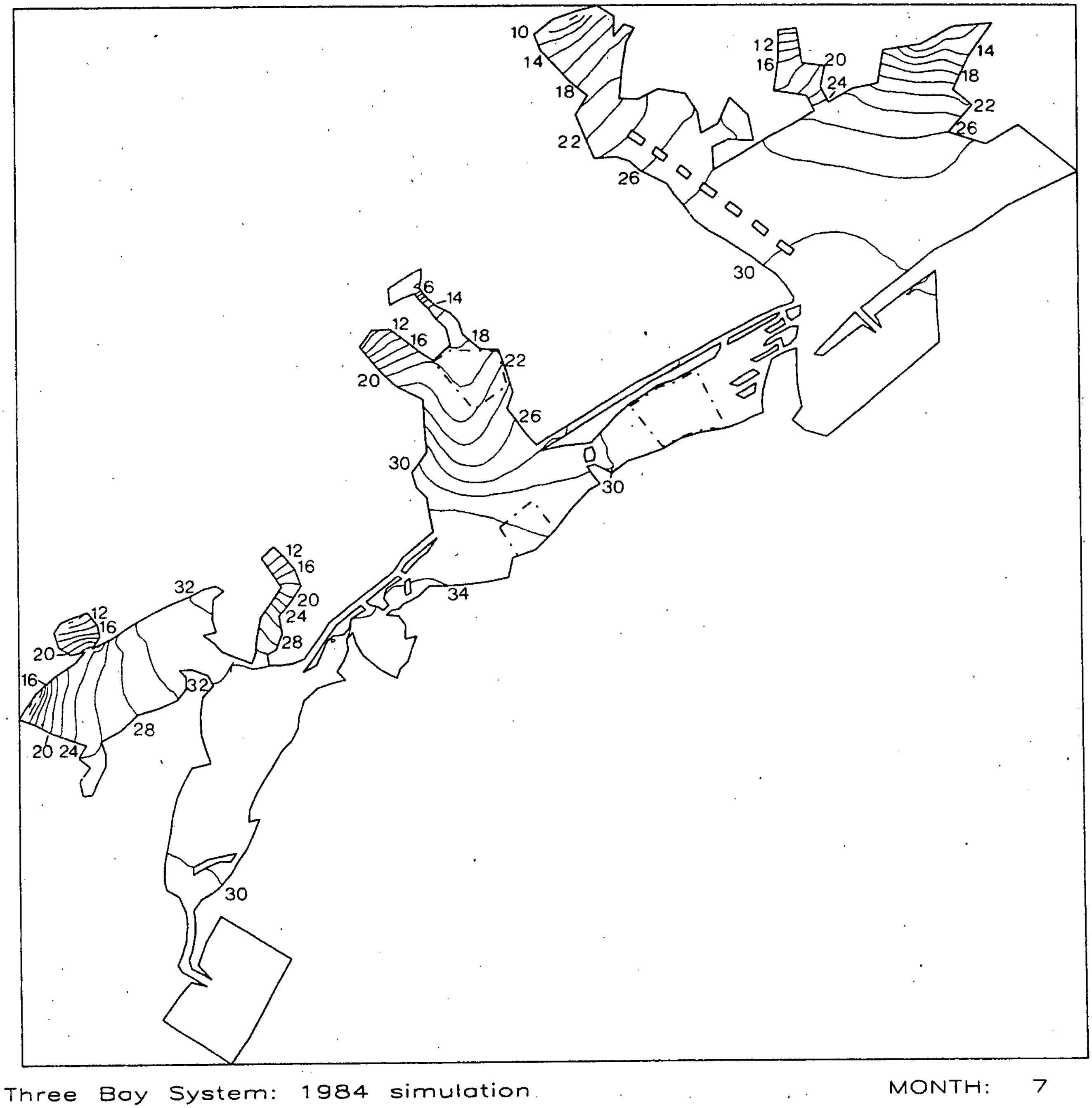

Figure 8.4.20. Isohalines of the simulated salinity from the Three-Bay Model for July 1984.

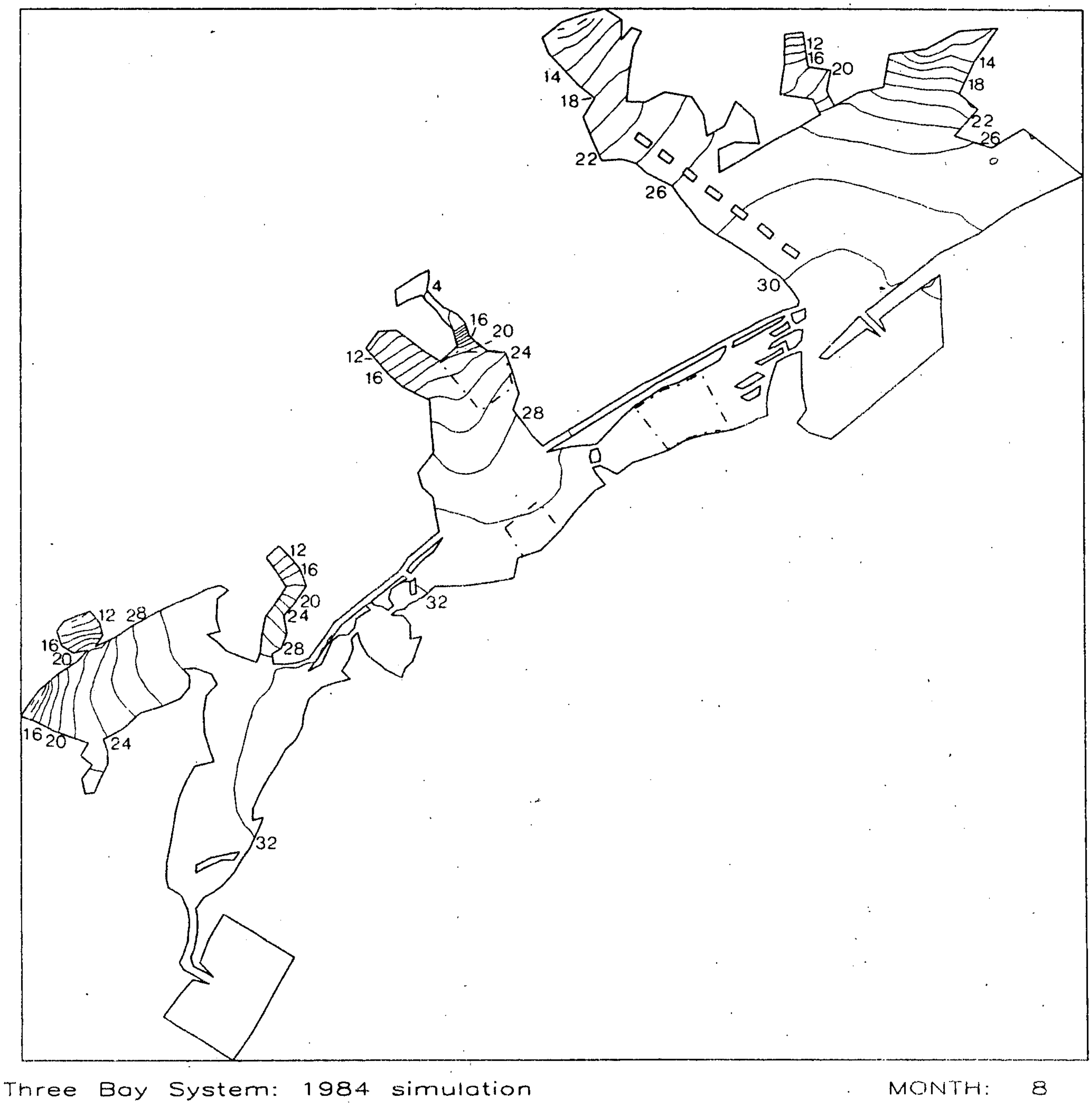

Figure 8.4.21. Isohalines of the simulated salinity from the Three-Bay Model for August 1984.

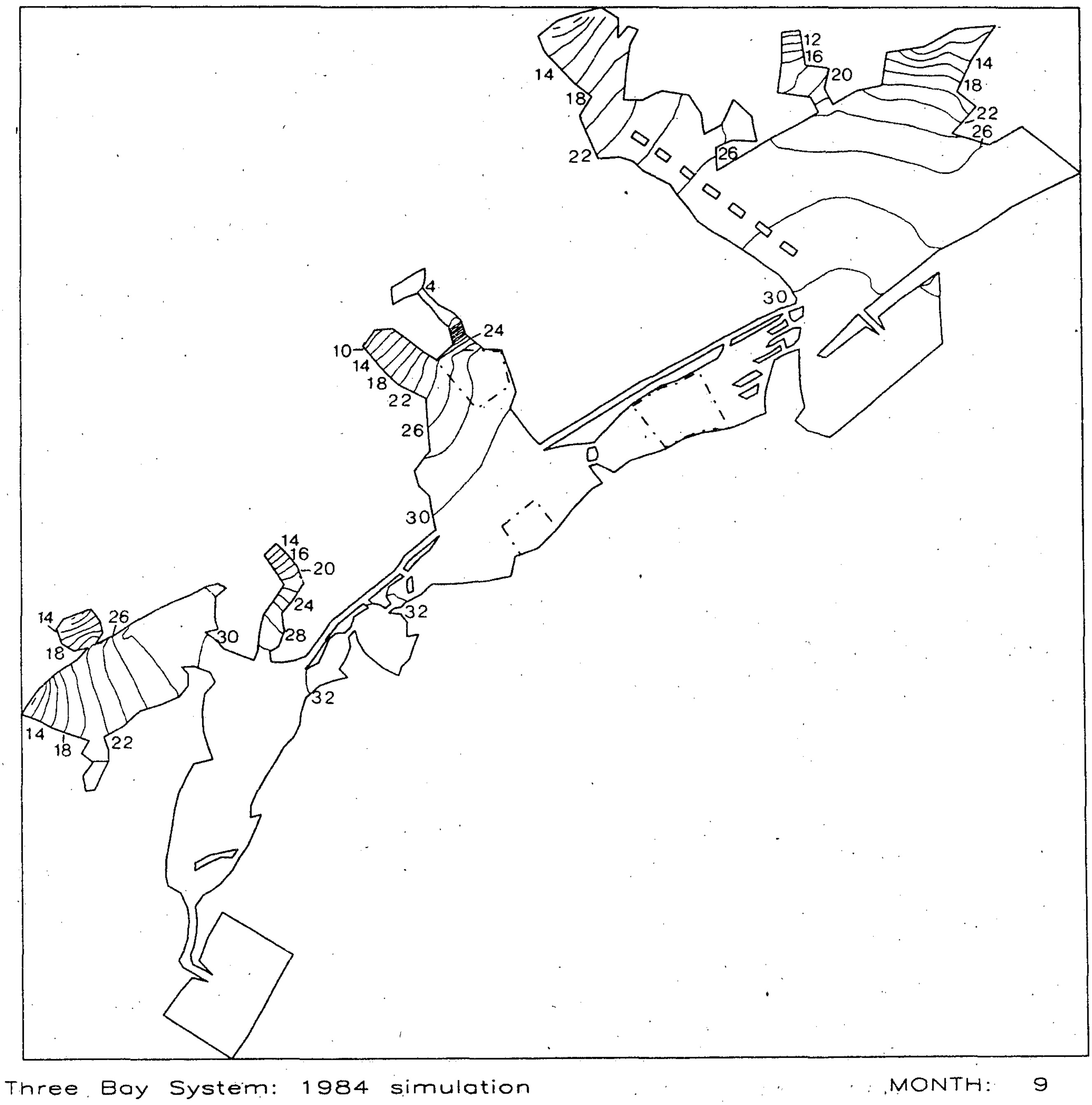

Figure 8.4.22. Isohalines of the simulated salinity from the Three-Bay Model for September 1984.

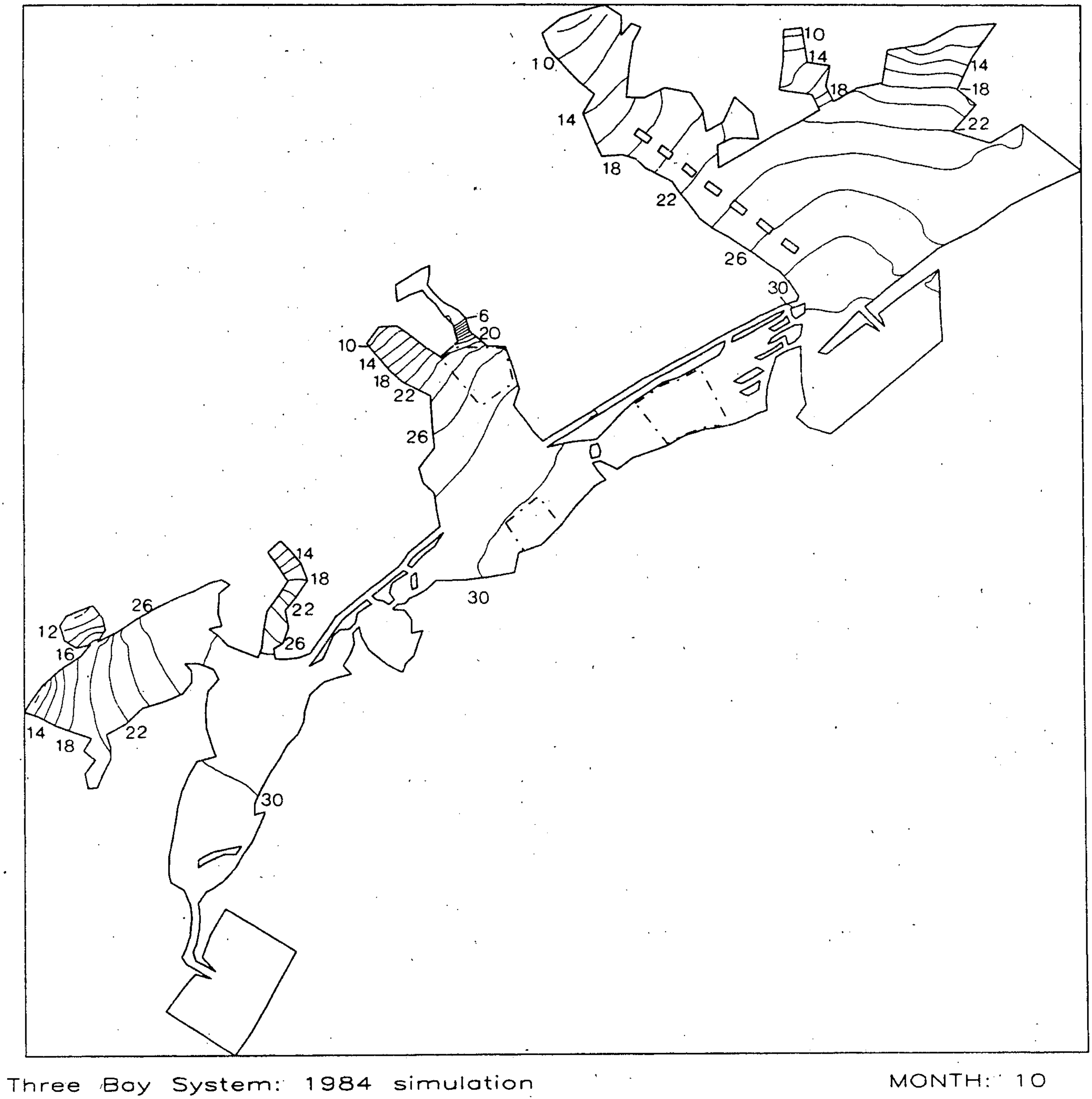

Figure 8.4.23. Isohalines of the simulated salinity from the Three-Bay Model for October 1984.

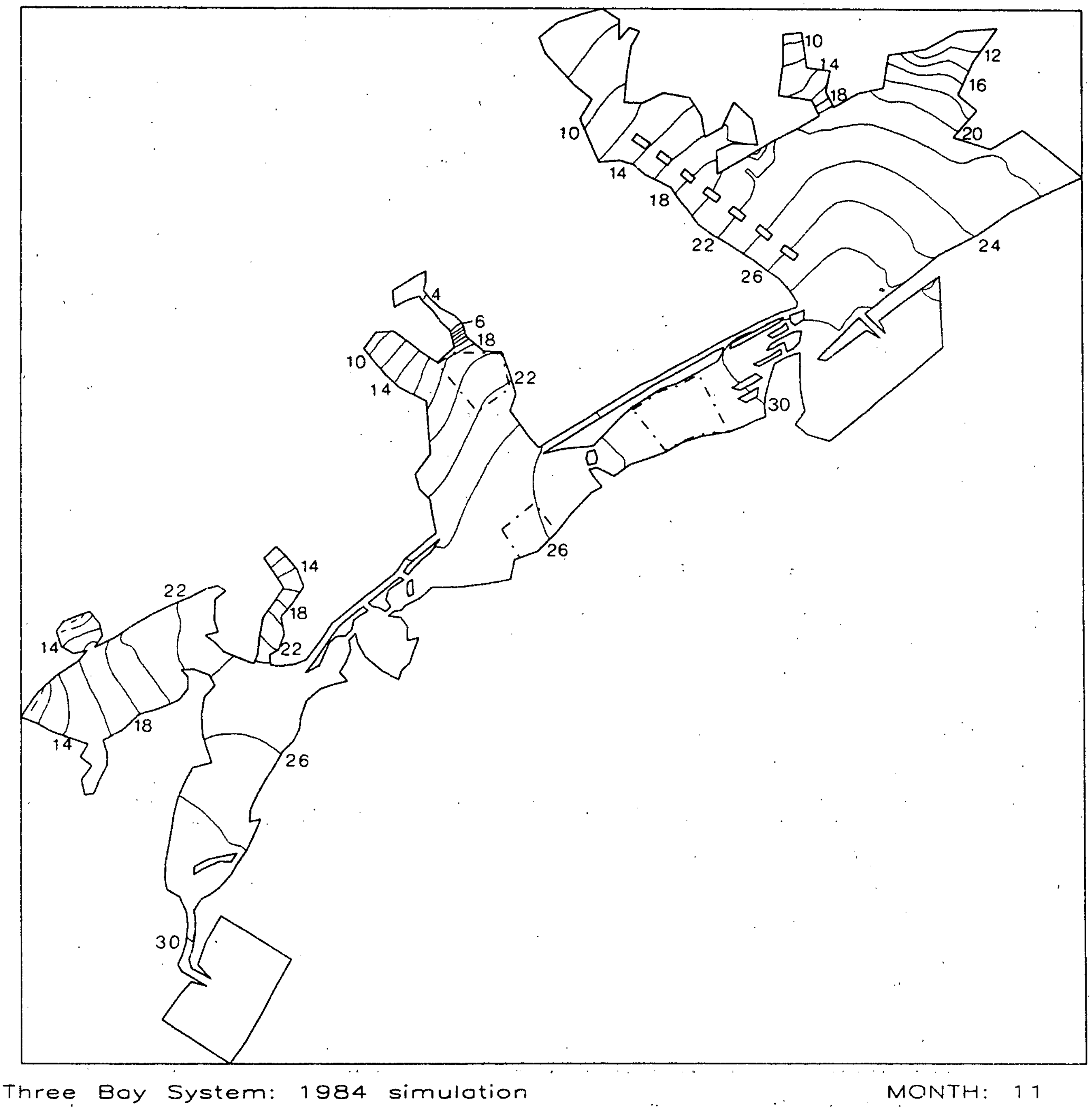

Figure 8.4.24. Isohalines of the simulated salinity from the Three-Bay Model for November 1984.

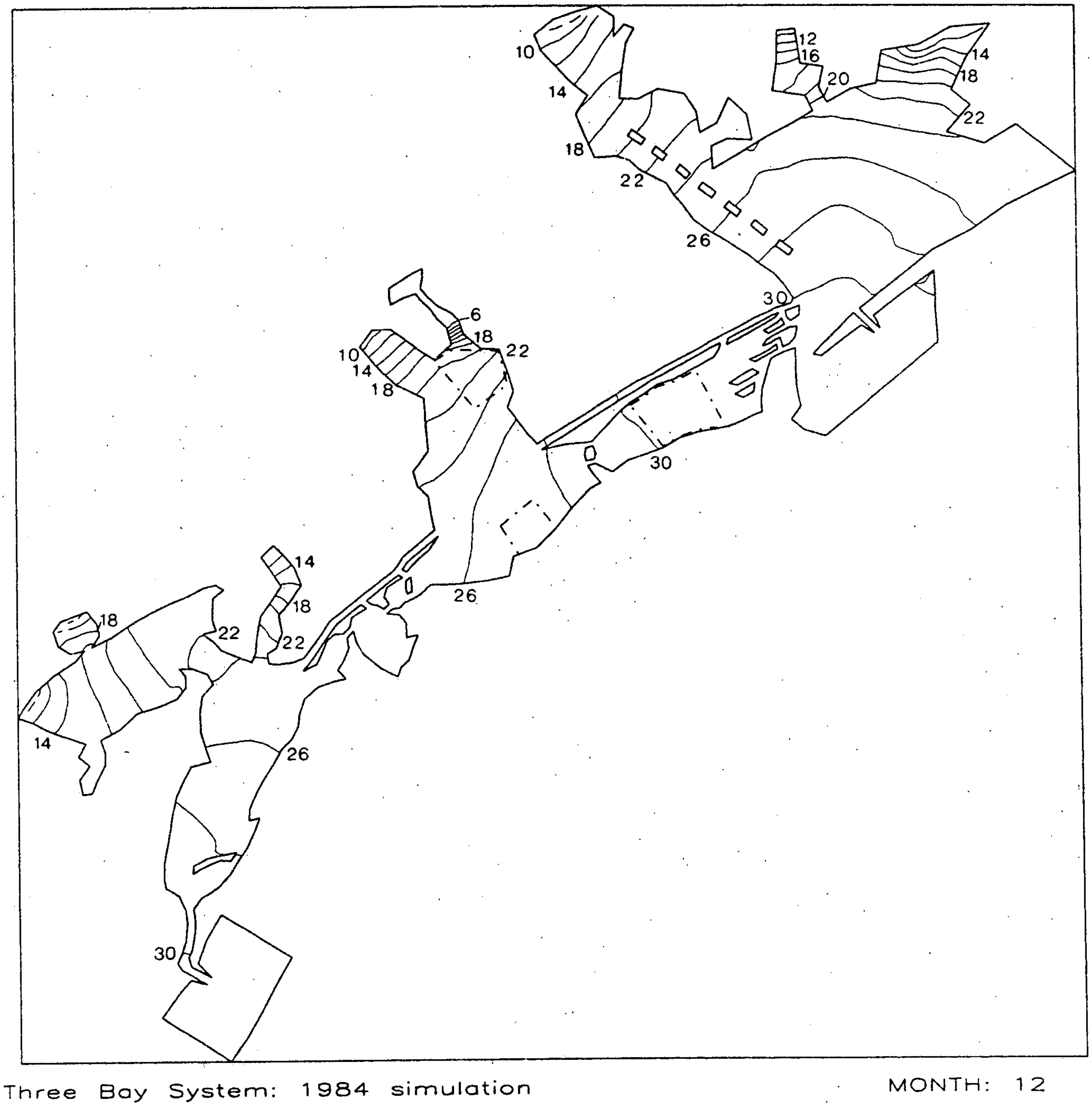

Figure 8.4.25. Isohalines of the simulated salinity from the Three-Bay Model for December 1984.

during the summer months is felt more in the upper part of the bay than in the lower part, with an approximate 6 to 7‰ reduction in salinity at the upper part, 4 to 5‰ at the mid-part, and 2 to 3‰ at the lower part of the bay.

In Figure 8.4.28 for lower San Antonio Bay, the period representing July has a sharp peak near day 200. This peak reflects the strong evaporation effect during July 1984, especially in Mesquite Bay and Ayres Bay.

Figures 8.4.29, 8.4.30, and 8.4.31 show the comparisons of the simulated salinities between the 1984 historical inflows and the minimum inflows (or MinQ solution). They show almost identical salinities in spite of inflow differences during the summer months (Table 8.4.3).

8.5 EVALUATION OF THE INFLOW ESTIMATES

Previous Studies

Several studies have addressed the links between freshwater inflow and productivity in the Guadalupe Estuary (Childress et al. 1975; TDWR 1980b; EH&A 1986). The first two of these studies considered the magnitude of commercial harvest of certain finfish and shellfish species as a function of freshwater inflow.

The Childress et al. study. Upon examination of 1961-1968 and 1971-1974 commercial landing data and historical gaged freshwater inflow data, Childress et al. (1975) concluded that an annual gaged inflow of 1.6 to 2.4 million acre-ft delivered on a seasonal schedule appeared to be associated with high shellfish production in San Antonio Bay. They found that total shellfish landings increased with annual gaged inflows up to 2.3 million acre-ft. Above this

Table 8.4.3. 1984, optimal, and minimum monthly inflows (1,000 acre-ft).

Month	1984	Optimal	Minimum
Jan	90.3	70.3	70.3
Feb	58.5	70.3	70.3
Mar	90.0	85.6	70.3
Apr	37.7	85.6	70.3
May	39.7	90.1	70.3
Jun	20.0	105.6	125.4
Jul	9.1	95.2	70.3
Aug	22.9	94.9	70.3
Sep	11.6	70.3	70.3
Oct	147.9	70.3	70.3
Nov	89.6	70.3	70.3
Dec	82.5	70.3	70.3
Annual	699.8	978.8	898.7

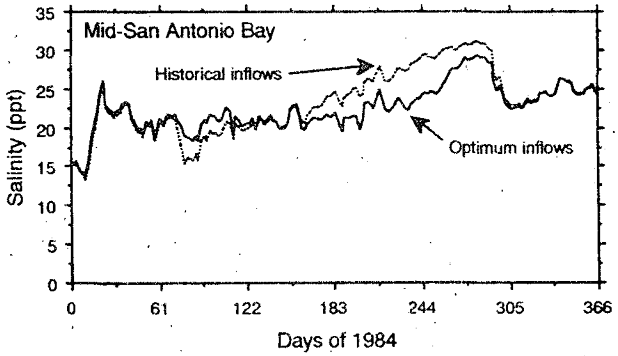

Figure 8.4.27. Simulated salinities for historical and optimal inflows at the 50% SalP for mid-San Antonio Bay.

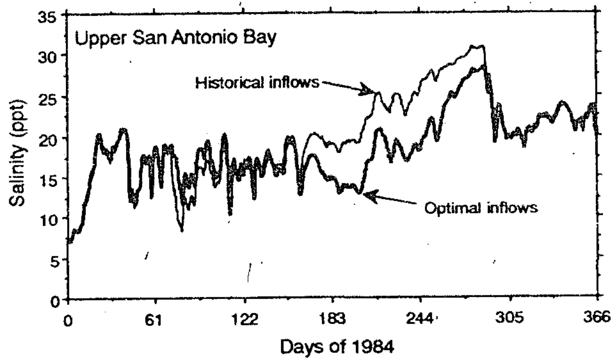

Figure 8.4.26. Simulated salinities for historical and optimal inflows at the 50% SalP for upper San Antonio Bay.

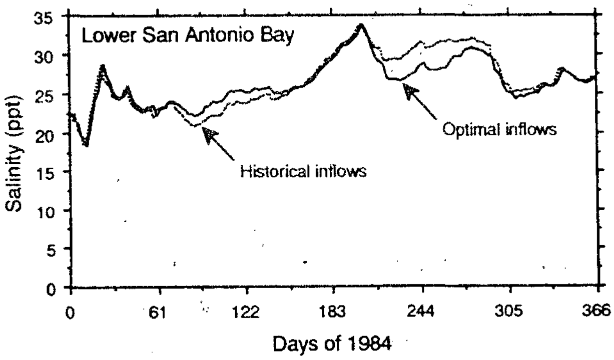

Figure 8.4.28. Simulated salinities for historical and optimal inflows at the 50% SalP for lower San Antonio Bay.

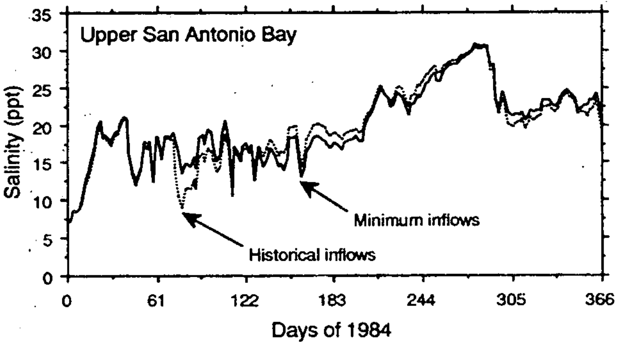

Figure 8.4.29. Simulated salinities for historical and minimum inflows at the 50% SalP for upper San Antonio Bay.

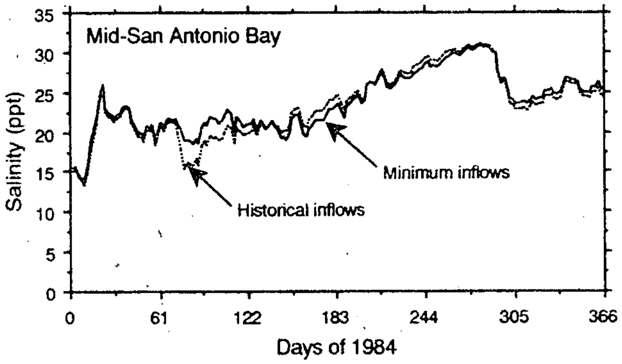

Figure 8.4.30. Simulated salinities for historical and minimum inflows at the 50% SalP for mid-San Antonio Bay.

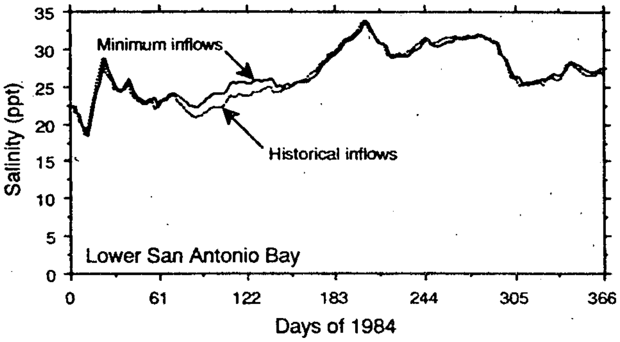

Figure 8.4.31. Simulated salinities for historical and minimum inflows at the 50% SalP for lower San Antonio Bay.

level, a decline in shellfish landings was generally noted. The ideal freshwater inflow seasonal delivery schedule was based on that of the three years of highest shellfish production of the period analyzed—1965, 1969, and 1972. January through April inflow accounted for 32.7% of the annual total, May through June accounted for 41.3%, and 26.0% occurred during July through December.

A similar relationship could not be developed for finfish landings in San Antonio Bay. Childress et al. postulated that one of the reasons for a lack of significant correlation of inflow with finfish landings could be attributed to finfish mobility and less dependence on currents associated with inflow that provide nutrients.

Table 8.4.4. Ranges of simulated salinites from TXBLEND in the three areas from which the inflow-salinity regression equation data was taken. Where isohalines from figures 8.4.14 through 8.4.25 did not cross the regression data area, the salinity was estimated by observing the nearest isohalines. The salinity ranges in parenthesis under the headings are the salinity lower and upper bounds used in the TXEMP Model analysis. Underlined values indicate salinites within the regression data areas that exceed the upper bounds (Section 8.2 and tables 8.2.7 and 8.2.8).

Month	Upper San Antonio Bay (1 to 20‰)	Lower San Antonio Bay (5 to 25‰)	Espiritu Santo Bay (10 to 40‰)
January	10 to 15‰	22 to 25‰	30 to 31‰
February	14 to 16‰	23 to 24‰	29 to 30‰
March	11 to 16‰	~23‰	27 to 29‰
April	13 to 17‰	~23‰	28 to 30‰
May	13 to 20‰	~25‰	28 to 30‰
June	15 to 23‰	~27‰	29 to 30‰
July	19 to 23‰	31 to 33‰	~31‰
August	20 to 26‰	~30‰	~31‰
September	24 to 29‰	~31‰	~31‰
October	22 to 27‰	~30‰	~31‰
November	18 to 22‰	25 to 26‰	~29‰
December	19 to 24‰	~27‰	30 to 31‰

The TDWR inflow study. A more detailed look at the freshwater inflow issue was taken by TDWR (1980b). The TDWR theorized that the inflow-salinity-biological relationships could be quantified and examined via three "key" indicators: (1) inundation frequency of marsh areas, which serve as important nutrient sources and nursery habitat areas, (2) monthly salinity levels, and (3) seasonal total freshwater inflow needs of the 1962-1976 commercial finfish and shellfish harvests. Equations were formulated that related 1962-1976 annual commercial fisheries harvests to seasonal freshwater inflows, monthly salinities to monthly freshwater inflows, and upper and lower monthly salinity bounds to viable salinity gradients for selected estuarine organisms for each Texas estuary. A mathematical programming model was used to determine monthly freshwater inflow requirements addressed by the three "key" indicators.

Three scenarios were analyzed for the Guadalupe Estuary. The subsistence scenario (Alternative I) provided freshwater inflows necessary to meet minimum monthly salinity levels and marsh inundation needs, but resulted in a decrease in finfish harvests. Annual gaged inflows of 1.2 million acre-ft (combined inflows of 1.6 million acre-ft), distributed monthly, were deemed sufficient to satisfy the subsistence scenario requirements. The maintenance scenario (Alternative II) also provided adequate freshwater inflows for salinity and inundation needs, but in contrast to the subsistence scenario, predicted that red drum, spotted seatrout, shrimp, and shellfish harvests would be maintained at or above 1962-1976 average levels. Annual gaged inflows of 1.6 million acre-ft (combined inflows of 2.0 million acre-ft), distributed monthly, fulfilled the maintenance scenario conditions. The enhancement scenario (Alternative III) provided inflows sufficient to maximize predicted shrimp harvests, but at the expense of declines in the harvests of both red drum and spotted seatrout. Annual gaged inflows necessary to meet the enhancement scenario criteria were 1.8 million acre-ft (2.3 million acre-ft of combined inflows) and followed a specific monthly pattern.

The EH&A river basin study. Espey, Huston & Associates, Inc. (EH&A 1986) conducted a study on the potential effects of reservoir development in the Guadalupe and San Antonio river basins on the downstream Guadalupe Estuary. Their approach to analyze the effects comprised three steps. First, the minimum monthly inflows necessary to protect the viability of the bay system were established. Second, operation of the proposed reservoir was simulated by a computer model with an assumed operating rule to determine reservoir yield from 1940 through 1982. Third, the result of the simulation was assessed as to the volume of inflows to the San Antonio Bay and the frequency of the occurrence.

To relate freshwater inflow quantities to the condition of the bays, regression equations between inflow and salinity were developed for nine sampling stations in San Antonio Bay. Biological reasoning was used to select salinity viability limits for each month for upper, middle, and lower portions of the bay. Based on these regression equations and the salinity limits, the minimum monthly combined inflows were determined to be 50,000 acre-ft from January through May; 40,000 acre-ft for June, July, and August; 70,000 acre-ft for September and October; and 50,000 acre-ft for November and December, totaling 610,000 acre-ft annually. The substantial differences in the estimates of freshwater needs among EH&A, TDWR, and Childress et al. can be attributed to the uncertainty associated with the commercial fisheries' data on which the latter two partly based their estimates, as well as differences in the estimated salinity viability limits and inundation requirements.

Comparisons among Recommended Inflows

Maximum harvest comparison. Figure 8.5.1 compares three recommended inflow scenarios: MaxH is from the results reported in Section 8.3 of this study for the uniform species weighting case, with 50% salinity probability; TDWR Alternative II is the Alternative II (maintenance) inflows reported by TDWR (1980b) and Childress Minimum is the minimum inflows recommended by Childress et al. (1975). The Childress et al. inflows have been adjusted upward to represent combined monthly inflows by assuming that the gaged inflow constitutes 80% of the combined inflow.

The Childress et al. and TDWR Alternative II inflows are similar, approximately 2.0 million acre-ft, while that of the MaxH solution from this study is about 1.0 million acre-ft, half of the Childress et al. minimum and TDWR Alternative II inflows. The Childress et al. inflows are greater than the historical median for 6 months of the year, while the TDWR inflows exceed the median for 10 months of the year. The monthly distribution of the Childress et al. recommended minimum provides more inflows during May and June, about twice the historical median during these months. The TDWR's Alternative II provides more inflow in May, June, and September. The MaxH solution has highest flows during March through June; in May and June, the MaxH inflows are one-half to one-third those reported by Childress et al. and TDWR.

Minimum inflow comparison. Figure 8.5.2 compares the MinQ solution for the 50% salinity probability and EH&A's minimum inflows. On an annual basis, the MinQ inflow is 0.9 million acre-ft, while the EH&A inflow is 0.6 million acre-ft. On a monthly basis, the MinQ solution recommends more inflow in May, but EH&A recommends more inflows in September and October.

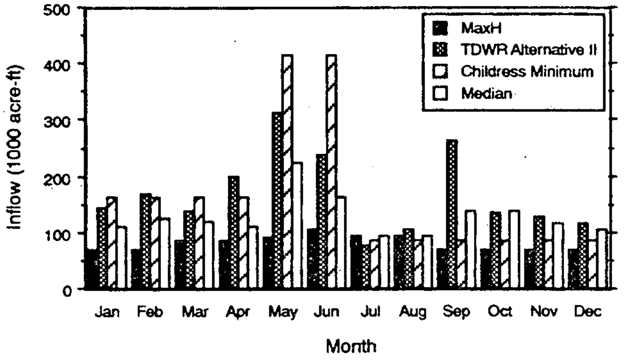

Figure 8.5.1. Comparison of median and recommended inflows for maximizing harvest from Childress et al. (1975), TDWR (1980b), and this study.

8.6 CONCLUSIONS

Estuarine Mathematical Programming Model. This chapter has presented the TXEMP Model for determining the freshwater inflow needs of an estuary. This procedure has been specifically designed to use an estimate of estuarine organism productivity as the performance variable so that the inflows needed to maintain the productivity of various sport and commercial fish and shellfish species can be determined, in accordance with the requirements of the TEXAS WATER CODE 11.147(a). In addition, this assessment model specifically includes elements involving salinity requirements of the animals and plants, and the nutrient and sediment loading regimes of the estuary, also in accordance with the TEXAS WATER CODE 11.147(a).

The TXEMP Model allows the user to include salinity limits in the analysis that are based on the life history requirements of estuarine organisms, and which can change during the year. The method includes a means to specify a salinity gradient and area-specific salinity limits throughout the estuary, and to change that gradient and the limits as desired. The TXEMP Model has provisions to include the historical pattern and quantity of inflow into the estuary through the use of inflow constraints. Users of this model can set the constraints so they are realistic in terms of historical flows, or they can be set to any desired level.

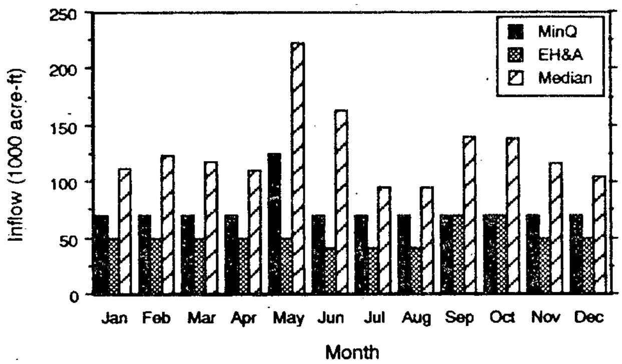

Figure 8.5.2. Comparison of median and recommended inflows to minimize inflow while satisfying viability limits (EH&A 1980) or satisfying all constraints (this study).

The user of theTXEMP Model can set targets for harvest of fishery species to ensure that the legislative intent of maintaining organism productivity is specifically addressed. Moreover, the procedure requires the user to select specific goals for estuarine management by specifying objective functions for optimization. This forces specification of species groups for which the estuary is being managed.

Harvest targets are selected by examining historical harvest statistics and determining appropriate levels for inclusion in the model. The harvest target information, as well as the equations relating inflow and salinity, and inflow and fishery harvest, are based on data collected over the last 20 to 25 years, which is specific to the estuary under study. The procedure takes advantage of the stochastic nature of the inflow-salinity and inflow-harvest equations in the analysis to provide a better definition of the risks involving tradeoffs between harvest and water allocation.

This assessment method is multiobjective, which allows the response of the harvest variables to be thoroughly investigated over a range of managerial options. The method is modular and open-ended so that additional quantitative relationships between inflow and production or constraints can be added as additional information becomes available.

Results of the Guadalupe example analysis. Using the TXEMP Model, an example analysis was done for the Guadalupe Estuary. Ten example cases consisting of different combinations of single species and groups of species were analyzed, although the most interesting and potentially useful evaluations resulted from multiple-species runs. It was clear that decisions concerning the selection of species for management and the relative importance of each species can strongly influence the inflow requirements. Needs of individual species for fresh water compete in the TXEMP Model, depending on the objectives for management. The competition can be extreme, as in the case of black drum whose freshwater inflow needs were in conflict with several other groups.

Attempts to decrease risk in the harvest of estuarine organisms by increasing the probability of satisfying salinity constraints resulted in a narrowing of the range of feasible inflow solutions and a decrease in the predicted harvest. In all cases, blue crab strongly dominated the harvest by weight.

In this example, neither the nutrient nor the sediment constraints ever controlled the inflow amount. Overall, inflow was controlled by salinity requirements (upper bounds) in lower San Antonio Bay. The salinity limits in the upper bay and in Espiritu Santo Bay did not control inflow amounts.

Inflow requirements ranged from 0.9 to 1.5 million acre-ft/yr, depending on the species and weightings among the species. This is half to two-thirds of the inflow requirements that were determined in previous studies that were based on productivity maintenance goals. The minimum inflow requirement (0.9 million acre-ft/yr) from this analysis is about 50% greater than was calculated by another study that was based only on viability limits of estuarine organisms.

Circulation and salinity model. This section has also presented results of a simulation of circulation and salinity conditions in the Guadalupe Estuary from the TXBLEND Model. The salinity values calculated by the model are reasonable and the general patterns of salinity distribution are consistent with measurements taken there. In general, the results from the TXBLEND Model using optimal inflows show that salinities in the zones used for the regressions do not substantially exceed the salinity bounds that were set for the TXEMP Model analysis.

CHAPTER 9: SUMMARY OF RESULTS AND RECOMMENDATIONS

9.0 INTRODUCTION

The purpose of this chapter is to provide a perspective about the effects of freshwater inflows on estuarine communities, and to summarize the findings and assessment methods that were presented in the preceding chapters and are available to determine freshwater inflow requirements. A series of objectives were defined in Chapter 2 for the Bay and Estuary Studies. Section 9.1 will discuss how each objective was satisfied and what conclusions can be drawn about the effects of freshwater inflow on salinity, sediment, and nutrient loading, and on maintenance of productivity of fish, shellfish, and other estuarine life.

Much of the information about inflow, salinity, nutrients, sediment, and organisms needed to use the TXEMP Model to estimate inflow requirements comes from combining and refining other types of data to provide the necessary equations and constraints for the analysis. Quantitative relationships or statistical values used in the method often represent hundreds of data points or the results of computations from other models. Section 9.2 reviews the availability of this type of information for other Texas estuaries, and provides suggestions about the kinds of data that must be gathered or analyzed to allow this technique to be applied to other estuaries. In a few instances, specific studies or research efforts that are required to provide the needed information are described.

Before the TXEMP Model can be used operationally, there are a number of decisions that must be made about the management objectives for each estuary. Some of these decisions are technical in nature and pertain to the manner in which the assessment method is used. Other decisions, however, have to do with basic management goals concerning the types of living marine resources that are desired, and the availability of fresh water, nutrients, and sediment to provide a suitable environment for their production. Section 9.3 provides a discussion of these issues and identifies some of the kinds of managerial decisions that will have to be made by policy-making bodies before the assessment technique can be applied to Texas estuaries.

9.1 SATISFYING STUDY OBJECTIVES

Eight objectives for the Bay and Estuary Studies were defined in Chapter 2. The information resulting from projects that were undertaken to address these objectives has provided the databases and methods proposed in this report for assessing the freshwater inflow needs of Texas bays and estuaries. It has also provided the foundation necessary to realize the five other post-study objectives that were also identified in Chapter 2. The specific means by which the eight study objectives were satisfied are discussed below.

Objective 1: Compile Inflow, Hydrographic, and Biological Data

Inflow data. The freshwater inflow studies completed at the end of the 1970's (TWDR 1980a, 1980b, 1981c, 1981d, 1981e, 1983) presented hydrology data for the Texas coast from 1941 through 1976. This study has extended the base of hydrology information through 1987 for all of the drainage basins from the Nueces Estuary through the Sabine-Neches Estuary. The information in the hydrology data files includes gaged and ungaged flows to the estuary as well as direct precipitation and evaporation estimates.

In situ salinity and water quality measurements. In 1986, in situ recording instruments were acquired and placed in the Nueces, Mission-Aransas, Guadalupe, Lavaca-Colorado, and Trinity-San Jacinto estuaries. In each estuary, one of the instruments was located in the upper, freshwater-influenced region, while a second instrument was located in the lower, marine-influenced region. In the spring of 1990, the instruments were redeployed to provide data from the Sabine-Neches Estuary and a wider area of the Trinity-San Jacinto Estuary. The instruments measured temperature, salinity and conductivity, oxygen, pH, and, in some instances, redox potential at an interval of once every one and one-half hours. The data from these instruments as well as other data from the TWDB, TPWD, TNRCC, and Texas Department of Health have been combined to provide data for inflow-salinity regression equations and to provide a historical record of salinity in Texas estuaries.

Additional water quality and biological data. Flow and water quality data from intensive inflow studies in the Lavaca-Colorado, Mission-Aransas, Nueces, and Galveston estuaries have been entered into computer files for analysis. Five years of water quality samples at more than 90 stations from the Laguna Madre to the Sabine-Neches Estuary were collected to supplement routine monitoring data from the TNRCC. These measurements included macronutrients, chlorophyll, total organic carbon, turbidity, and conductivity. Data from this sampling program has been entered into the Coastal Data System computer files maintained by the TWDB. In addition, computer files containing original data from cooperating universities and agencies that performed studies under contract with the Bays and Estuaries Studies program have also been obtained and stored.

Objective 2: Develop Circulation and Salinity Models for Texas Bays

Limitations of existing models. The circulation and conservative transport (salinity) models prepared for the previous bay and estuary studies (TDWR 1980a, 1980b, 1981c, 1981d, 1981e) provided a good first approximation of water movement and salinity patterns that would be produced within the estuaries under various inflow regimes. These models did have drawbacks, however. Because computational power was limited and the models were run to steady state to calculate net flows over a particular time period of interest, the grid sizes, time steps, and areal extent of the models had to be conservative. Only square or rectangular grids could be used, and individual grid cells were quite large—one nautical mile square. Each cell in the grid was significantly wider than the dredged channels, natural passes, and many other portions of the bays themselves. A number of assumptions about flows between adjacent bay systems had to be made since it was not possible to model several interconnected bay systems. Furthermore, there was no possibility of running the models to simulate periods as long as a year.

Improvements from finite element models. In the past decade, there have been significant advances in the mathematical approaches to hydrodynamic modeling. Major changes have involved the refinement of finite difference models and the development of finite-element models. TXBLEND is a finite element model which provides some distinct advantages over the earlier simulation models. TXBLEND can use triangular elements (grid cells) which are particularly adaptable to the bay's geometry, providing the fine detail of flows in channels, passes, and around islands. Not all of the elements must be the same size. Smaller elements can be used to model flow in channels, and larger elements can be used for open bay areas where fine detail is not as important. Moreover, the element size can be reduced so that one or more elements can fit within the width of a channel. This provides greater accuracy in simulating the details of water movement in these critical areas. Overall, the finite element method substantially improves the representation of system geometry and will certainly improve the accuracy of the circulation and transport simulation.

Recent advances in computational speed of small computers and workstations, and decreases in prices of hardware allow problems involving much larger grids to be run in significantly less time than was possible a decade ago. It has become possible to run the models to simulate an entire annual cycle so that salinity changes can be computed and evaluated for the entire year. The initial application of TXBLEND required the simulation of three estuarine systems along the Texas coast that are interconnected by the ICWW (Guadalupe, Lavaca-Colorado, and Mission-Aransas estuaries). The Guadalupe Estuary has only a small direct connection to the Gulf through Cedar Bayou; tidal fluctuation depends on interconnection with Aransas Bay, Matagorda Bay, and Pass Cavallo. Thus, simulation of all three estuaries as a group was necessary to determine the circulation and salinity patterns in the Guadalupe Estuary. TXBLEND allowed summation of the interflows back and forth among the three systems, which was necessary for calculating the nutrient budget of the Guadalupe Estuary.

Model limitations. Like every model, TXBLEND has its own set of drawbacks. Although it provides extreme flexibility in preparing grids, much greater attention must be paid to the details of the grid. In particular, mass conservation with the TXBLEND model near inflow points is very sensitive to the grid design. It is necessary to take great care with the grid so that water mass is neither created nor lost in the computations. Nevertheless, the improvements in resolution and accuracy of circulation patterns provided by the finite element technique should improve the quality of decisions made using the model results. A new version of the Three-Bay model is available and will be used in the future.

Objective 3: Evaluate Effects of Salinity and Salinity Change on Estuarine Plants and Animals

Freshwater inflow strongly influences the salinity of estuarine water; the magnitude and timing of inflow events largely determines how rapidly and to what extent the salinity decreases from freshwater dilution of tidal waters in the estuaries. Estuarine plants and animals have different degrees of tolerance to salinity change. The discussion below focuses on the effects of inflow and salinity on zooplankton, benthic organisms, and finfish and shellfish abundance. It is difficult to divorce the effects of salinity on plants from consideration of primary productivity, so the effects of inflow and salinity on phytoplankton and macrophytes are reviewed in the discussion of Objective 6, effects of inflow on primary (plant) production.

The effects of salinity and inflow on zooplankton. Estuarine microzooplankton and macrozooplankton are quite euryhaline and can survive large changes in salinity levels associated with freshwater inflows. Production of tintinnids, which are part of the microzooplankton, appears to be particularly stimulated by freshwater inflows. Measurements of *Acartia tonsa*, the dominant macrozooplankton in Texas estuaries, also indicate that secondary production of this species is stimulated by high inflows. Very large inflows into the upper portions of estuaries change both the salinity of the water and the community of macrozooplankton; freshwater forms are swept into the estuary, and estuarine and marine forms are washed out.

Several patterns emerge when zooplankton populations are analyzed on an annual basis. Except for very dry periods with extremely low inflows, the abundance of macrozooplankton appears to be higher in the more saline portions of estuaries closer to the Gulf than in the upper, fresher areas. Abundance of macrozooplankton is higher in the winter and spring, and lower in the summer and fall.

In the Nueces Estuary, a year with relatively high inflow (about one bay volume per year) with respect to the annual median (about one-half bay volume per year) resulted in twice the abundance of macrozooplankton compared to a year with very low inflow (about 0.1 bay volume per year). The opposite pattern was seen in Lavaca Bay and the Guadalupe Estuary which had higher macrozooplankton abundances in the years of lower inflow than in the years of higher inflow. The low inflows to these two bay systems, however, were substantially higher on a bay volume basis (3.5 and 5.5 bay volumes/yr, respectively) than the low inflow to the Nueces Estuary. The high inflows to Lavaca Bay and the Guadalupe Estuary were much higher on a relative basis (9 and 14.5 bay volumes/yr, respectively) than the high flow volume seen in the Nueces Estuary. The Lavaca and Guadalupe high flows were probably large enough that macrozooplankton were displaced out of the estuary into the Gulf by the water movement, thus reducing the abundance of macrozooplankton. The results suggest that macrozooplankton abundance is low at very low inflows, but increases with inflow up to some maximum level probably associated with the influence of the "washout" rate. Inflow increases beyond this point, however, carry macrozooplankton out of the estuary faster than they can be replaced through increased production. The inflow volume producing the highest abundance appears to be between one and six bay volumes per year. Just how reproduction and grazing rates interact with inflow events is not understood, although it is clear that they do influence zooplankton density.

Effects of salinity and inflow on benthic organisms and processes. There is limited information for Texas estuaries about microbenthic animals, which include bacteria and protozoans. The little information available suggests that high densities of microbenthos are associated with periods of high inflow, but not enough data is available to materially substantiate this suggestion. We do not have enough information to clearly characterize the abundance, biomass, or productivity of this benthic group.

There seems to be a strong association between meiobenthic animals and salinity. Under normal inflow conditions, total meiobenthic abundance is several times greater in the more marine areas than in the fresher, river-influenced areas. Meiofaunal community structure is different in the upper and lower bay regions. Nematodes are most abundant at lower bay sites, while they represent a smaller fraction at upper bay sites. Nematodes require high-salinity environments. When the salinity declines as the result of large inflows, a surge of juvenile mollusk recruitment occurs at upper bay stations. This offsets losses in the nematode populations so total meiofaunal abundance remains about the same. In lower bay areas, low salinities cause the nematode populations to decline, but there is no simultaneous increase in other groups as there is in the upper bay area. Consequently, the total meiobenthic abundance declines.

From several discontinuous studies, it appears that conditions eventually become favorable for meiofaunal population increase as the salinity begins to rise. Meiofaunal abundance remains high for weeks to months, possibly the result of organic and inorganic material inputs accompanying the large freshwater inflow events, which could provide food resources for meiofaunal populations. Eventually meiofaunal abundance decreases to lower levels. A definitive study covering several-years that have different inflow frequencies is needed to substantiate this general pattern.

The macrobenthic invertebrates of Texas estuaries include many species which tolerate a broad range of salinities. Typically, benthic species diversity increases with increasing salinity within an estuary. But after comparing results of several studies on various Texas bays, macrobenthic abundance did not show a consistent trend with salinity. High abundances have been found at locations with salinities near marine levels and at low salinities. Total abundance is influenced by a number of biotic and abiotic factors that are not thoroughly understood. A conceptual model has been proposed to explain the cycles and patterns of benthos abundance in Texas estuaries based on responses to flood and drought cycles, which seem to characterize the Texas climate. Adherence to the model's response pattern has not yet been confirmed.

Recycling inorganic nutrients is an important process in providing sufficient materials for high levels of phytoplankton production. Since all three benthic groups are involved in the recycling of nutrients from plant and animal tissues and other pieces of organic detritus that settle to the bottom, it would seem that regeneration of nutrients from particulate organic materials would parallel the patterns of abundance or biomass of benthic organisms. Unfortunately, there is no good correlation for macrobenthic organisms, and abundance and biomass information on micro- and meiobenthic populations is inadequate to test the hypothesis. While there is evidence that the magnitude and duration of inflows may influence the level of regeneration, it is unclear how it occurs. Multi-year studies capturing several sequences of high and low inflow, abundance and biomass measurements of all three benthic components, and benthic and water column regeneration are needed to fully establish the functional relationship between inflow and nutrient regeneration.

Finfish and shellfish abundance with respect to salinity regime. Texas bays and estuaries vary greatly in salinity, freshwater inflow, evaporation rate, and habitat type. Much of the difference in fish abundance among estuaries may be attributed to differences in inflow quantities, salinity regimes, habitat characteristics, or some combination of these factors. In sections 6.2 and 6.7 of this report, we examined the relationship between the relative abundance of several fish and shellfish species and the salinity regimes of Texas estuaries. Total catch for each species by trawl, gill net, or bag seine was converted to catch per unit effort (CPUE) and averaged monthly. The CPUE's were then tested for differences among estuarine systems in one model, and regressed against mean estuarine salinity or salinity regimes within an estuary in another model. No relationship between estuarine salinity and gill net catch was detected for any large adult or subadult fishes. Several smaller fishes and shrimp, caught in otter trawls, were selective for salinity regime. Of the animals captured in the trawl, blue crab (*Callinectes sapidus*), brown shrimp (*Penaeus aztecus*), and southern flounder (*Paralichthys lethostigma*) preferred coastal estuaries with mean salinities of 17 to 21‰. White shrimp (*Penaeus setiferus*) and Gulf menhaden (*Brevoortia patronus*) favored less saline estuaries such as Galveston Bay (15‰ salinity); pinfish (*Lagodon rhomboides*) selected higher salinity regimes such as Corpus Christi Bay (31‰ salinity); and Atlantic croaker (*Micropogonias undulatus*) reached highest abundance when mean salinities were between 20 and 30‰. Bag seine data from the Guadalupe Estuary agreed with this conclusion and provided evidence that alteration of freshwater inflow may significantly change relative fish abundance among estuaries and the local fish distribution within an estuary.

Objective 4: Assess Water Quality Trends

This objective concerned water quality in the estuaries in the form of trends over the past two decades, and the relationship of dissolved nutrients in estuarine waters to inflow conditions. In this report, the focus of water quality analyses was on the loading and concentrations of nitrogen, phosphorus, and organic carbon, which are major nutrients fueling the productivity of the estuaries.

Effect of inflow on estuarine nutrient concentrations. The relationship between nutrient concentration and inflow was examined statistically for the Guadalupe Estuary and graphically for five mid-coast estuaries. Under all inflow conditions, higher nutrient concentrations were found in the upper reaches of the estuaries, near river mouths. During periods of inflows higher than median levels, areas of high nutrient concentrations appear in middle regions of the estuaries. There is also some evidence of "outwelling" of nutrient-rich waters into the Gulf from the passes at Bolivar Roads and Aransas Pass during these high inflow periods. Lower bay areas near the barrier islands, however, may not show a marked increase in nutrient concentrations as inflows increase. Concentrations of nitrogen and phosphorus in Texas bays are usually above the threshold levels at which primary production would be limited.

Nutrient trends in the Guadalupe Estuary. Trends in concentrations of nitrogen, phosphorus, and total organic carbon during the 1968 to 1987 period were tested for the Guadalupe Estuary. The patterns were not consistent over the estuary. While there has been a trend of increased concentrations in the San Antonio River contributing to the estuary, dissolved inorganic nitrogen concentrations in mid-San Antonio Bay and parts of the lower estuary have decreased. Conversely, total phosphorus concentrations have increased in the estuary, while not showing an increase in the Guadalupe or San Antonio rivers. Concentrations of total organic carbon have also decreased in the estuary over this period.

Adequacy of estuarine nutrient supply. Texas estuaries are moderately to well-supplied with nitrogen, phosphorus, and total organic carbon from drainage basin sources. The Trinity-San Jacinto and Guadalupe estuaries receive as much of these nutrients on an annual average as estuaries on the East Coast, which have developed symptoms of eutrophication. River sources provide the largest share of nitrogen to the Guadalupe Estuary and to upper coast estuaries, while direct rain, rainfall runoff from local watersheds, and wastewater discharges contribute the largest portion of nitrogen for estuaries of the lower Texas coast (Mission-Aransas and Nueces). The ratio of phosphorus to nitrogen of nutrient loads entering Texas estuaries suggests that nitrogen is more likely to be important in controlling primary production than phosphorus.

Fate of nitrogen in the Guadalupe Estuary. Nutrient budget calculations for the Guadalupe Estuary showed that the estuary exported 30 to 33% of incoming nitrogen to neighboring bays or the Gulf of Mexico. Losses of nitrogen to fisheries harvests, denitrification, and burial, however, could approach 50% of incoming nitrogen in a dry year, versus less than 20% of incoming nitrogen in a year of high inflows. During a year of high inflow, therefore, the Guadalupe Estuary may store a significant proportion of incoming nitrogen, which could be recycled in later years to support primary productivity.

Objective 5: Determine Effects of Inflows on River Deltas and Bay Sedimentation

Availability of information. There is a substantial amount of information on the suspended sediment load for the Sabine, Neches, Trinity, Colorado, Lavaca, Guadalupe, San Antonio, Mission, and Nueces rivers from data collected by the TWDB and the USGS. While the data sources for suspended sediment load are ample, there is basically no information about bedload transport in the same rivers. This is a significant information gap, since the coarser fractions of sediment play an especially important role in the development of river banks, levees, and bars in river deltas.

Discharge-suspended sediment load relationships. Due to the availability of suspended sediment data and good stream discharge information, quantitative relationships between monthly streamflow and monthly suspended sediment load posed no problem to prepare. The relationships can be used to estimate suspended sediment load under a wide range of inflow conditions.

Graphic analysis through cumulative mass curves of suspended sediment load and inflow for several river systems showed times in the recent past when the trend of suspended sediment load to the estuaries changed dramatically. Reduction in sediment load occurred immediately after completion of Lake Livingston on the Trinity River (1968). Reductions in sediment load on some other rivers were not closely linked in time to the construction of reservoirs over the period of record, although a number of major reservoir projects predate the period over which suspended sediment information is available. Sediment load reductions for the Lavaca, Guadalupe, and San Antonio rivers occurred in 1958, a year after the end of a three-year drought. Reduction in sediment load occurred on the Nueces River in 1972, 14 years after the construction of Lake Corpus Christi and a decade before the completion of the Choke Canyon Reservoir.

From sparse data, it appears that the Mission River had a reduction in sediment load in the 1980's, while the Colorado River may have had an increase during the same time period. The Neches River had an increasing trend in sediment load from 1960 to about 1980, followed by a slight decrease. The Sabine River had no trend from 1974 through the early 1980's, although a slight decrease appears to have occurred in the last few years of data. Changes in sediment load that do not correspond with any natural or water resource development event may have resulted from shifts in land use activities such as agricultural practices that cannot be attributed to a particular date.

Effects of inflow on river delta and bay sedimentation. Through the use of aerial photographs spanning several decades, it was possible to document decreases in vegetated areas and increases in open water areas in some river deltas of Texas bays. While changes in delta areas can be easily identified, it is not yet possible to quantitatively relate these changes to decreases in suspended sediment load. No general analytical model exists yet that relates the aerial extent of deltas or vegetated areas with the sediment load. Until such a model is available, and until more extensive and accurate measurements of bathymetry, sediment thickness, relative sea-level rise, sediment transport, compaction, progradation, and aggradation are made, it will be difficult to produce more than a qualitative judgment of the effects that changes in sediment load will have on river deltas and bay sedimentation.

Sediment loads in the Nueces, Lavaca, and Trinity rivers appear to be strongly influenced by upstream land uses and water resource developments. While sediment loads to the river deltas have been reduced for each of these river systems, the effects of the reductions are complicated by other factors including relative sea-level rise, local geologic changes, levee and canal construction, and upstream agricultural practices. The areas of the Nueces and Trinity rivers' deltas have declined as the result of reduced sediment load.

Lavaca Delta area decreases predating Lake Texana may have been exacerbated by other factors, but the recent completion of the reservoir may contribute to more extensive and accelerated delta submergence. Changes in the Guadalupe Delta are probably due to the natural delta growth and decay cycle, and not to the effects of any upstream water resource developments. This delta is still actively developing, although the focus of deposition is now on Mission Lake rather than on the old lower delta. The sediment load in the Colorado River has been greatly reduced by the construction of the highland lakes, but sediment continues to be delivered to the Colorado Delta and will continue to accumulate as the diversion project at the mouth of the Colorado directs river flow into the eastern arm of Matagorda Bay.

Objective 6: Evaluate the Effects of Freshwater Inflows on Estuarine Primary (Plant) Production

The assessment of the effect of salinity on estuarine plants, which was stated in Objective 3, is included in the discussion below. In addition, analyses of other freshwater inflow effects on plant production besides salinity, such as light limitation (turbidity), inundation, nutrient loading, and biogeochemical cycling of essential nutrients, are presented.

Effects on phytoplankton communities. In Section 5.1, the conclusion was reached that no clear relationship between phytoplankton and salinity could be established from existing studies. Based on regression analysis of data for several different bay systems, inflow rates versus chlorophyll-*a* did not show a simple relationship. For the Nueces Bay system, salinity levels explained only minor amounts of variation in chlorophyll levels, while light availability in the water was the best predictor of phytoplankton biomass. In studies of the Guadalupe and Nueces estuaries, nutrient input, especially of nitrogen, was considered a limiting factor on phytoplankton growth. The relationship between freshwater inflows and phytoplankton production in the Trinity-San Jacinto and Laguna Madre estuaries can best be termed paradoxical. In both cases, comparably high rates of phytoplankton primary production occur, although the Trinity-San Jacinto has high inflow rates and turbid waters, while the Laguna Madre has low inflows but much higher water clarity. Much research is still required to fully understand the effects of freshwater inflows on estuarine phytoplankton community dynamics.

Effects on input of organic material to bays. Review of existing data on input dynamics in Section 5.6 led to the conclusion that delta marshes of Texas bays were not unusual by demonstrating inconsistent patterns of material movement into bays. Most transport was hypothesized to occur during major storm or flood events rather than on normal tidal cycles, and such episodic transport has been difficult to measure. Carbon tracer studies using stable carbon isotope ratios provided evidence that the Guadalupe River carries organic matter from terrestrial sources into San Antonio Bay. The corresponding process of transport of delta marsh carbon into bays could not be adequately determined due to the overlap in carbon isotope values of upland and marsh plants. The terrestrial carbon distributed throughout the San Antonio Bay system by riverine inflows was traceable in food webs to economically important fish and shellfish species. This was in contrast to the Nueces Estuary with its low freshwater inflow, where phytoplankton and seagrasses were more important carbon sources. Although riverine transport of terrestrial carbon was demonstrated by some studies, the quantitative significance of this carbon to bay carbon budgets still remains undetermined.

Effects on submerged and emergent vegetation. Requirements of estuarine-rooted vegetation for specialized inundation cycles and salinity, nutrient, and sediment loading regimes controlled by freshwater inflows or local bay processes were reviewed in Section 5.2. Emergent halophytic vascular plant communities in Texas bays were considered more affected by water level variations (inundation cycles) than by salinity changes of the tidal waters. This reflects their fundamental requirements as aquatic plants for moisture, and secondly, their wide tolerance as halophytes to salinity variations which occur in the dynamic estuarine environment. The species most susceptible to freshwater inflow alterations were identified as sensitive marsh species occurring in the freshwater- to low-brackish delta areas of estuaries such as the Guadalupe Estuary. Plants such as bulrushes (*Scirpus* spp.), cattails (*Typha* spp.), water hyacinth (*Eichhornia crassipes*), and arrowhead (*Sagittaria* spp.) are restricted to this low salinity zone near the river mouth and depend on proper flooding from freshwater inflows to maintain the unique combinations of hydroperiods, low salinity, and nutrient-laden water. Inundation in the lower portion of bays is accomplished by tidal cycles with higher salinity waters, but this water must still supply the marsh with sufficient sediment and nutrient loads from freshwater inflows.

Submerged vegetation, represented by water nymph (*Najas guadalupensis*) and widgeongrass (*Ruppia maritima*)from low-salinity regions (0 to 10‰) or seagrasses in higher-salinity regions (15 to 35‰), presents a unique case for estuarine vascular plant management. The submerged growth habit of these plants makes them susceptible to ecological limitations by water clarity and light availability. While water turbidity is a major factor contributing to this light limitation, the sources of turbidity and other light-attenuating material in bay waters are complicated. Local conditions (wind, sediment types, local runoff) are difficult to differentiate from freshwater inflow-induced effects. Nutrients supplied in freshwater inflows or other sources also pose a potentially deleterious factor due to excess nutrient-loading which can stimulate phytoplankton blooms or overgrowth of epiphytes attached to seagrass leaves. This mechanism, proposed as a serious problem for submerged vegetation and maintenance of seagrass beds, requires more documentation in Texas bays.

Objective 7: Develop Statistical Harvest-inflow Regression Equations for Commercial Fisheries

Data and method of analysis. Harvest data for several commercial fishery species [white shrimp, brown shrimp, blue crab, eastern oyster (*Crassostrea virginica*), and black drum (*Pogonias cromis*) were available through 1987. Commercial harvests of spotted seatrout (*Cynoscion nebulosus*) and red drum (*Sciaenops ocellatus*) ceased in 1981 when these species were declared to be state gamefishes. For shrimp, there were 29 years of commercial harvest data, and for most of the other species, 26 years of information. Regression analyses were performed using harvest as the dependent variable and freshwater inflow, air temperature, and fishing effort as the independent variables; separate analyses were done for gaged streamflows and combined freshwater inflows. The inflow volumes were represented in the equations as seasonal (two-month) inflows which were antecedent to the harvests. The inclusion of seasonal inflows provides a measure of biological realism by representing a potential interaction between inflows and the physiological state of the animals at different times in their life cycles. Particular attention was paid to the statistical procedures and the selection of the "best" regression equation.

Equations. The fishery analysis included all seven species for the Trinity-San Jacinto, Lavaca-Colorado, Guadalupe, Mission-Aransas, and Nueces estuaries, although only a subset of these equations has been presented in Chapter 6 of this report. Equations for all species from the Guadalupe Estuary were presented, and comparisons among white shrimp, blue crab, and spotted seatrout were made with equations from all five of the estuaries. The 26 regression equations presented explained an average 69% of the harvest variance (range 44 to 90%). While the statistical equations alone cannot establish biological cause and effect connections, they do represent the quantitative character of the relationships that exist between fishery harvest and the environmental variables tested.

Objective 8: Develop Methodology to Define Objective Functions and Constraints for Use with Optimization Procedures

Development of the Estuarine Mathematical Programming Model. A new optimization model (TXEMP) for determining inflow requirements was created with a number of advanced features. The model is a nonlinear, multiobjective mathematical programming (optimization) model which can incorporate the statistical uncertainty of the relationships between inflow and salinity or inflow and fisheries harvest through the use of chance constraints. Harvest is used as the measure of system productivity that indicates the response of the biological community to the hydrologic regime. Other measures of productivity could be used as well where a relationship between the indicator variable and inflow or salinity is found. The model has six objective functions that can be used for optimization: minimize or maximize inflow, minimize or maximize harvest, maximize probability of achieving salinity constraints, and maximize probability of achieving harvest targets.

The establishment of a salinity gradient throughout the estuary is taken into account in this model by including area-specific inflow-salinity regression equations and salinity constraints for different regions of the estuary. The salinity-inflow regression equations contain the essence of the spatial relationship of the gradient, and their use assumes the historical distribution from the specific inflow source locations.

The model is hydrologically and biologically realistic since users must include constraints on inflows that are based on historical inflow patterns and constraints on harvest levels that reflect historical yields. The model also includes consideration of nutrient and sediment loading as minimum inflow constraints. This is consistent with legislative requirements for state water rights permits.

Presentation of an example analysis. An example analysis using this model was presented for the Guadalupe Estuary. The prototype analysis uses real data but presumes a number of management objectives and policy decisions for the purpose of illustrating the methodology. If the model were being run for an actual water rights permit analysis, the objectives and policy decisions would have to be set by governmental permitting and resource management authorities. The example shows results for each species by itself and for several different weightings of species. From these results, it is possible to see the conflicting requirements for inflow of individual species and to note the effect of the various weighting schemes. The analysis examines one case in depth, showing how different probability levels for meeting salinity constraints affect the harvest-inflow performance curves, and examines the sensitivity of the results to various bounds in the model.

Conclusions about the Effects of Freshwater Inflow

Freshwater inflow has a significant influence on many components of estuarine ecosystems. It is a strong determinant of salinity and materially influences the input and distribution of nutrients throughout the bay. Years with high inflow probably have the important effect of providing nutritive materials that are stored in bay bottoms, to be processed and used later to maintain estuarine productivity during periods of lower inflow.

Freshwater inflow transports suspended sediment and is responsible for building deltas and maintaining habitat structure. Terrestrial organic material carried by river flow is distributed throughout estuaries that have moderate to large inflows. It is not clear whether carbon produced in river delta wetlands is transported into the estuary by freshwater inflow at any time except during flood conditions.

No obvious relationship has been shown between freshwater inflow or salinity and phytoplankton production or abundance. Light availability for photosynthesis appears to be a better simple predictor than inflow. Marsh plants are more affected by water level and soil moisture than by inflow or salinity, which shows the wide osmotic tolerance of halophytic marsh plants. Some fresh and brackish emergent species and a few submergent species are susceptible to elevated salinities, but most of the seagrasses prefer higher salinities. Submergent vegetation is very sensitive to light availability in turbid bay waters. Thus, high levels of turbidity carried by inflows, or "shading" by phytoplankton blooms or heavy epiphyte overgrowth resulting from too high levels of nutrients in inflows, may limit the growth of submerged plant species.

The abundance of zooplankton appears to be related to freshwater inflows. In general, zooplankton abundance is greater in the more marine areas than in river-influenced areas. There seems to be a stimulatory effect of inflow on zooplankton production, while high inflows transport zooplankton out of the estuaries to the Gulf. Since these are opposing processes, there appears to be a balance point of maximum zooplankton abundance that is a function of inflow.

Meiobenthic species are typically more abundant in the higher salinity regions of estuaries. High inflows decrease this abundance and alter the community structure in both high- and low-salinity areas. Macrobenthic species abundance is affected by inflows, but simple plots of benthic abundance versus salinity do not reveal a clear relationship. A proposed mechanism of control of macrobenthic abundance by inflow still awaits testing. Benthic organisms are instrumental in the regeneration of nutrients from particulate organic materials on the bay bottom; while there is some indication that elevated levels of regeneration are linked to inflow events, this also awaits definitive testing.

From trawl and bag seine studies, the abundance of smaller finfish, shrimp, oyster, and crab is selective with respect to salinity. Thus, alterations in freshwater inflow would directly affect their distribution and abundance in Texas estuaries. The abundance of many adult finfish collected by gill nets did not indicate a relationship between abundance and salinity. Therefore, changes in inflow would not have direct effects on the abundance of adults of these species, although it could indirectly influence their distribution through changes in their habitats. There does not appear to be a relationship between inflow and the transport of larval organisms from the Gulf to the bays. However, there is a very clear relationship between salinity and both metabolism and reproduction. Salinity extremes from very high or very low inflows require additional metabolic effort for osmoregulation and body maintenance. In addition, there appear to be optimal salinity levels for adult ovarian growth, egg fertilization and embryological development, hatching, and larval survival. For species that reproduce within the estuary, altered freshwater inflows may strongly influence reproductive success.

9.2 INFORMATION NEEDED TO APPLY THE METHODOLOGY TO OTHER ESTUARIES

Introduction

In the development of the TXEMP Model and the application to the Guadalupe Estuary, a minimum set of information and results from other models was needed to provide the equations and constraints for the example analysis. Information types were selected to be responsive to the legislative requirement that beneficial inflows, computed by the assessment methodology, must provide a salinity, nutrient, and sediment loading regime that is adequate to support an environment in which the productivity of several different fishery species is maintained. TXEMP directly addresses these legislative requirements by incorporating information about salinity bounds for communities of organisms in specific estuarine areas throughout the year, as well as annual minimum nitrogen and sediment loads for the estuary.

Assuming the same minimum set of information is needed to apply the assessment technique to the other estuaries, this section discusses the status of this information for each major estuarine system. Table 9.2.1 shows the overall status of data for the seven major Texas estuaries that contain large bays or lagoons as of June, 1994. Three additional estuarine systems exist on the Texas coast: the Brazos River, the San Bernard River, and the Rio Grande. All are river estuaries that open directly into the Gulf of Mexico. While estuarine species are found in the tidal regions of each river estuary, the habitat area for the produc-

Table 9.2.1. Status of data files and information other than policy decisions needed to apply method to other Texas estuaries.

Information topic	Estuary Laguna Madre	Nueces	Mission-Aransas	Guadalupe	Lavaca-Colorado	Trinity-San Jacinto	Sabine-Neches
Hydrology update from 1977-1987		√	√	√	√ Update with Colorado River diversion	√ Update with Christmas Bay	√
Sediment loading				√	In progress	In progress	
Hydrodynamic and conservative transport model			3-bay	3-bay	3-bay, detailed model in progress	√	
Inflow-salinity regressions		√	√	√	√	√	Data compiled
Nutrient balance		Loading data compiled	Loading data compiled	√	In progress	√	
Fishery equations		√[a]	√[a]	√[a]	√[a]	√[a]	Inadequate fishery data
Areal distribution of wetland types	√[b]	√[b]	√[b]	√	√[b]	√	√[b]
Inventory of consumers by area	√	√	√	√	√	√	√
Abundance of major consumers		Partially complete		√		√	

[a] Equations are based on fishery-dependent data instead of fishery-independent sampling methods.

[b] Generalized floral assemblage or wetland maps, not detailed with respect to species distribution or abundance.

tion of estuarine species is only a small fraction of the bay-bottom and wetland areas associated with the seven major estuaries. Consequently, the development of an assessment technique has focused on the seven large estuarine systems rather than the smaller river estuary environments. Comparable data for the three river estuaries is very sparse, so several years of intense data collection would be required to provide the same degree of understanding of salinity, nutrients, sediment deposition, and productivity.

The second part of this section addresses information beyond the minimum required to operate the assessment technique. While these other types of information are not absolutely necessary to use the methodology, they would expand it by including additional expressions of productivity and other topic areas that are appropriate for estuarine management.

Information Necessary to Operate the TXEMP Model

Hydrology. The hydrological data needed to apply the TXEMP Model to the remaining major estuaries in Texas consists of historical freshwater inflow estimates for each estuary. These data are used to compute basic inflow statistics such as mean, median, and percentile inflows that are used as either targets or bounds for the optimization solution. Furthermore, daily freshwater inflows are used to establish statistical relationships between salinity and freshwater inflow, and monthly freshwater inflows are used to relate fish harvest to inflow.

Monthly freshwater inflows have been compiled for the Nueces, Mission-Aransas, Guadalupe, Lavaca-Colorado, Trinity-San Jacinto, and Sabine-Neches estuaries for the period 1941 to 1987. Daily freshwater inflows, based on gaged, diversion, and return flow measurements and ungaged model simulations, are available for the same estuaries for the period 1977 through 1987. Daily freshwater inflows used in inflow-salinity regression equations, based on gaged measurements, ungaged model simulations, and estimated diversion and return flows, are available for the period 1960 through 1976 for the Nueces, Mission-Aransas, Guadalupe, and Lavaca-Colorado estuaries.

Both monthly and daily data are now available for use with the estuarine programming model. Since this freshwater inflow data were compiled, the U.S. Army Corps of Engineers has constructed the Mouth of the Colorado River Project which diverts river discharges into Matagorda Bay. This project will alter inflow statistics as well as the salinity structure in Matagorda Bay, and may render current relationships inadequate for determining inflow needs. In light of this new construction, freshwater inflow estimates, salinity-inflow relationships, and harvest-inflow relationships will need to be re-calculated for the Lavaca-Colorado Estuary. It may not be possible to establish these relationships until sufficient data have been gathered over a period long enough to determine the full effect of the diversion on inflows, salinity, and harvest. In a cooperative effort with the TWDB, TPWD, and TNRCC, the Lower Colorado River Authority (LCRA) has undertaken the development and analysis of much of the information needed to run the TXEMP Model for the Lavaca-Colorado Estuary. They are currently extending the hydrology database from 1988 through 1992.

For the Laguna Madre Estuary, only monthly freshwater inflows are currently available for the period 1941 to 1976. Extension of the Laguna Madre freshwater inflow database through 1994 will probably not begin until late 1996 pending publication of gaged inflows by the USGS. Other future work includes the development of hydrology data for Christmas Bay, a recently designated coastal preserve on the southwest end of the Trinity-San Jacinto Estuary.

Sediment loading requirement. Several possible management objectives could be used to quantify a sediment loading regime adequate to maintain an ecologically sound environment in Texas bays and estuaries. The most encompassing objective would be to satisfy the sediment requirement of an entire estuarine system, so adequate material could be delivered and the estuary would retain its current physical form. Unfortunately, knowledge about relationships between sediment loading and the many aspects of sediment transport, accretion, and erosion in estuaries are too incomplete to allow quantification of these relationships. Preliminary measurements of delta aggradation have been made in the Colorado and Trinity deltas; areal extent comparisons based on photographs have been made for the Nueces, Guadalupe, Lavaca, Colorado, San Jacinto, and Trinity deltas (White and Calnan 1990a, b). While these studies provide a good start for determining the extent of change of some of the delta areas, additional efforts involving direct measurement of aggradation through time, and areal change from photographs, are required to provide an empirical relationship between inflow and delta area, volume, or elevation. For future estuary studies, we can assume that some measure of delta change—aggradation or areal extent—could be related to sediment load carried by the rivers.

Currently, the major deficiency is a lack of a generally accepted simple relationship defining the influence of inflow or sediment load of rivers with the maintenance of bay-bottom bathymetry or wetland habitat. Costanza et al. (1990) prepared a spatial modeling system to relate habitats (termed ecological landscapes) with water flow, sedimentation, primary production, organic matter, turbulent resuspension, and subsidence. This model may be applicable to the sediment loading problem, although it might require considerable modification of some of the underlying relationships to account for differences in habitat types. The model would require extensive data gathering, calibration, and testing. While it might provide a long-term solution for sediment loading analysis, the extensive effort needed for testing and calibration may be premature given the paucity of information about sediment transport, deposition, and resuspension in Texas estuaries. A more empirical method will have to serve until adequate information is available for a modeling approach.

In the example analysis in Section 7.5, we determined the sediment needs for various portions of the Guadalupe Estuary delta. The final estimate was made on the basis of a small, well-defined area (Mission Lake) where it was clear that river flow provided the sediment for maintenance of bathymetry. The peculiar circumstances of the Guadalupe Delta and Mission Lake allowed a simple analysis for determination of the requirement. The decision to limit the area under consideration was made as a practical matter, so the analysis could go forth; the requirements for sediment throughout the rest of the estuary were discounted because of the need for a bounded area in which depth change could be estimated. In effect, this was a policy decision based on knowledge of the limitations of available information. Perhaps the responsible policy-making bodies (Section 9.3) will provide additional guidance in the future about the areas on which to focus sediment loading analyses, since estimates for the whole estuary are not yet possible. Based on knowledge from available information on the value of habitats that may be affected by sediment loading, we believe that focusing attention on maintenance of the delta wetlands is appropriate. However, the decision on how to define the area of an estuary in which the inflow-sediment relationship is based will strongly influence the methods used to estimate the loading requirements.

Assuming that maintenance of or increase in existing delta areas is a suitable objective, a conceptual framework for a simple analysis involves two steps: selecting the appropriate delta areas on which to focus the analysis, and determining the details of how to relate river flow or sediment load to the extent, elevation, or volume of the selected area(s). This type of analysis will require several years to complete since there are very few direct measurements of aggradation currently available, and new measurements will have to be collected to represent an adequate time span so that change can be detected, and a range of inflow conditions experienced. It may be possible to collect additional areal extent information from historical aerial photographs to provide more data relating delta area and inflow.

There are many complications to successfully completing the second step of this conceptual plan, most of which involve inadequacies of the available information. For example, historical data for sediment load in Texas includes only suspended sediment (Section 4.4 and Holley 1991), so it may be difficult to relate sediment load carried by rivers to aggrading or accreting areas in which sand, largely carried as bedload, is an important constituent material. Some experts suspect that most sediment transport in the deltas is episodic, coinciding with storms such as hurricanes, or with major floods (Gross et al. 1978; Milliman and Meade 1983; White and Calnan 1990b). The effect of episodic events compared to more normal flows will have to be evaluated. There are no measurements of the contribution of ungaged areas to sediment load; consequently, current estimates of sediment load to the estuary may be low. Compaction of newly deposited material along with general subsidence in the area and sea-level rise (White and Calnan 1990b) may complicate the analysis significantly. And, human activities in the area, such as dredging or construction, may overwhelm or disguise steady geologic processes that work over long periods of time.

TWDB staff in conjunction with researchers from the Bureau of Economic Geology of The University of Texas at Austin, and the U.S. Geological Survey in Denver, Colorado, are currently working on a method to relate river flow with delta deposition. Cores are used to date short increments in the top meter of deposits from the Trinity and Lavaca delta, using naturally occurring radioisotopes of lead, cesium, and beryllium. This provides estimates of rates of deposition over short time spans (months to years). An analytical procedure is being developed to relate these deposition rates with the historical flow patterns of the river. Isotope measurements are complete for the Trinity Delta, and analysis is under way to relate the sediment accumulation with river flow. Cores have been collected from the Lavaca Delta and are currently being analyzed.

A comprehensive program is needed to look at the quantity and quality of sediment carried by the rivers to the deltas, the transport and distribution of sediment in the deltas, the effect of episodic events on the deltas' sediment movement, and other sediment sources, sinks, and forces (other than inflow) that may carry sediment to or from the deltas. Since formation and maintenance of deltas is part of a larger picture of sediment processes within an estuary, including shoreline erosion, it would be reasonable to make delta maintenance studies part of a larger effort on understanding the movement of sediment in the estuary as a whole.

Hydrodynamic and conservative transport model. A Three-Bay simulation model encompassing the Mission-Aransas, Guadalupe, and Lavaca-Colorado estuaries using the TXBLEND hydrodynamic and conservative transport model has been presented in this report. Application of a hydrodynamic model is an important component in the freshwater inflow assessment methodology because it supplies essential information for calculating the nutrient budget (needed for setting the nutrient constraint) and it provides a necessary check on salinity levels computed by the TXEMP Model. The Three-Bay Model was calibrated using 1988 field survey data; a technical report describing the calibration and application of the model will be prepared.

After calibration, a question was raised regarding TXBLEND's treatment of the conservation of mass. Under some circumstances a portion of inflow, especially if it is very small or very large, can be lost at the inflow point if the finite element grid is too coarse. The problem is usually resolved by further refinement of the grid. In addition, the TXBLEND model has been revised to minimize this problem and testing of the revised version is complete. The TWDB intends to continue to improve existing hydrodynamic models and to investigate others, including three-dimensional models, to provide the best available models for calculation of estuarine circulation and salinity.

With regard to the modeling status of other estuaries, field studies have been undertaken in all estuaries to allow calibration of hydrodynamic and transport models, although additional data may need to be collected for refinement of the models and for verification. Finite element grids have been prepared for all seven estuaries, and preliminary model runs have been made for the Nueces and Lavaca-Colorado estuaries. Calibration of the Trinity-San Jacinto Estuary model is nearly complete. The LCRA is currently calibrating the model for the Lavaca-Colorado Estuary. Calibration has not been started for the Sabine-Neches, Mission-Aransas, and Laguna Madre estuaries.

Inflow-salinity regression equations. Regression equations predicting salinity as a function of freshwater inflow have been completed for the Nueces, Mission-Aransas, Guadalupe, Lavaca-Colorado, and Trinity-San Jacinto estuaries. Data compilations needed to calculate salinity-inflow relationships are complete for the Sabine-Neches Estuary, but not for the Laguna Madre. Experience in using the equations in the optimization model has indicated that additional attention should be given to the selection of particular bay areas to represent the estuary's salinity gradient (refer to figures in Section 4.1). In addition, the possibility of using other multivariate inflow-salinity equations should be evaluated. Other regression forms could include the various sources of inflow or other factors as independent variables that might better describe the relationship between inflow and salinity. This would allow better assessment of effects on bay salinities resulting from changes to major inflow sources, while preserving the idea that all inflows have influence to some degree throughout the estuary. The development of multiple regression equations would be a refinement to the assessment method that could be incorporated into the TXEMP Model as they became available. Consideration has been given to incorporating evaporation into the salinity-inflow regressions, although results of tests show adding evaporation reduces variance only about half the time.

Nutrient loading. Calculation of nutrient budgets depends chiefly on estimates of nutrient loading to the estuaries, estimates of exchange rates between estuaries and adjacent waters, especially the Gulf, and on measurements of other rates, such as biological activities, which affect nutrient concentrations. Nutrient loading data is nearly complete, with information compiled for the major estuaries except for Sabine Lake and Laguna Madre. Estimates of tidal exchange and advective transport of materials between adjoining estuaries was shown in Section 7.3 to be very important to the budget. These estimates were derived from the application of the hydrodynamic model. Therefore, nutrient budget calculations will lag behind hydrodynamic model development for each system.

Parameters for rates of nutrient loss in some important biological and geochemical processes have been measured for several estuaries in recent studies. These measurements may be applicable to estuaries other than those from which they were made. However, given the range of salinities and environments encompassed by the bays, from low-salinity mud flats to the high-salinity seagrass beds of Laguna Madre, it would be surprising if the values of these parameters were constant. Denitrification in particular should be measured in the bays of the upper coast. Measurements should be made to establish the relationship of denitrification rate with temperature, water column nitrogen concentration, and salinity.

The rate of burial of nutrients below the level of biological activity in the sediments probably accounts for a relatively low proportion of nutrient loss, but there is very little data to support an estimate of the rate. Nutrient loss to burial might be greatest near active deltas, but estimates are needed for all bays.

Exchange of materials with the Gulf of Mexico is obviously important in establishing many characteristics of Texas estuaries. With respect to nutrient budgets, calculations showed that tidal inflows made surprisingly large contributions of nutrients to the bays, even when Gulf nutrient concentrations were low. More information on Gulf nutrient concentrations on the coastal shelf would help to more accurately determine Gulf-bay exchange. In addition, the rate of mixing between Gulf and bay waters during flood tide determines how much new Gulf water is actually incorporated in the bay with each tidal cycle. This mixing rate is likely to be dependent on the morphology of each major pass. The nutrient budget presented for the Guadalupe Estuary relied on an estimate of mixing rate proposed for Aransas Pass. Mixing rate estimates are needed for all other major passes.

The nutrient budget is complete for the Guadalupe Estuary and nearly complete for the Trinity-San Jacinto Estuary. The LCRA, with the assistance of the TWDB, is currently preparing a nutrient budget for the Lavaca-Colorado Estuary.

Fishery equations. Fishery harvest data for brown shrimp, white shrimp, blue crab, eastern oyster, spotted seatrout, red drum, and black drum have been analyzed for the Sabine-Neches, Trinity-San Jacinto, Lavaca-Colorado, Guadalupe, Mission-Aransas, and Nueces estuaries. Statistically significant ($P < 0.05$) equations for these estuaries are available for all species, except those indicated in Table 9.2.2.

Fishery harvest data for the Laguna Madre are available for 1962 through 1987, but the complementary freshwater inflow data are not available beyond 1976. Consequently, no new harvest equations have been developed for the Laguna Madre, although completion of the hydrological analyses will allow the regression analyses to be performed. Harvest data for Sabine Lake are problematic, biased by unfavorable changes in estuarine conditions (TDWR 1981e), and contain broken (discontinuous) time series records as a result of shifts in the local fishing industry. Although fishery analyses were completed for Sabine Lake, problems with the data caused a lack of confidence in most of the statistical equations that were produced.

Table 9.2.2. Summary of analyses of coastal fisheries harvests. The meaning of the symbols is as follows: X = statistically significant harvest equation developed; O = no statistically significant harvest equation; Z = harvest equation developed from discontinuous data; NF = no fishery data available; NH = no hydrology data available.

Estuary	Flow source	White shrimp	Brown shrimp	Blue crab	Bay oyster	Red drum	Spotted seatrout	Black drum
Sabine-Neches	Gaged	Z	Z	X	NF	Z	Z	Z
	Combined	Z	O	X	NF	Z	Z	Z
Trinity-San Jacinto	Gaged	X	X	X	X	X	X	O
	Combined	X	X	X	X	X	X	X
Lavaca-Colorado	Gaged	X	O	X	O	X	X	X
	Combined	X	O	X	O	X	X	X
Guadalupe	Gaged	X	X	X	X	X	X	X
	Combined	X	X	X	X	X	X	X
Mission-Aransas	Gaged	X	X	X	X	X	X	X
	Combined	X	X	X	X	X	X	X
Nueces	Gaged	X	X	Z	NF	X	X	X
	Combined	X	X	Z	NF	X	X	X
Upper Laguna Madre	Gaged	NH	NH	NH	NF	NH	NH	NH
	Combined	NH	NH	NH	NF	NH	NH	NH
Lower Laguna Madre	Gaged	NH	NH	NH	NH	NH	NH	NH
	Combined	NH	NH	NH	NH	NH	NH	NH

All 74 of the fishery equations from the regression analyses are based on commercial harvest data over a 26- to 29-year period. There are many potential sources of error in this fishery-dependent data including problems with incomplete data reporting, inaccuracies due to different levels of precision in reporting, concerns about the measures of effort, and questions of whether landings from one estuary include some catch from adjacent estuaries. In addition, the harvest information does not include recreational catch. While these are serious problems, the fact remains that the commercial harvest information is a long-term data set, spanning almost three decades for some species. No other database of estuarine animal abundance in Texas provides as long a period of record. During this time, the coastal fisheries experienced a wide range of freshwater inflows, which allows statistical testing to discern any relationships that might exist between fishery performance and inflows.

The fishery-independent monitoring data from TPWD is a data set that shows promise of providing improved information on changes in fish and shellfish abundance or distribution as a function of inflows. The TPWD staff have collected samples with gill nets since 1977 and with trawls since 1983, using a randomized stratified sampling strategy. Rigorous statistical considerations have guided this sampling program so that statistically unbiased estimates of abundance can be determined. At some future time, there will be enough of this data to perform regression analyses to estimate annual fishery abundance or distribution that avoids problems inherent with the fishery-dependent harvest data. If such an analysis is successfully demonstrated, the fishery-independent sampling data could be used to develop fishery equations for all the estuaries, without the biases of harvest data. This database contains information on prey species as well as most economically important fishery species.

Areal distribution of wetland types. Two complete series of printed maps of Texas coastal wetlands are available, both based on 1979 NASA color-infrared photography. The environmental maps prepared by The University of Texas Bureau of Economic Geology for the submerged lands of Texas publications series (White et al. 1983, 1985, 1986, 1987, 1988, 1989) accurately delineated areal distribution of wetlands for all seven Texas estuaries at this time. Similarly, the USFWS National Wetland Inventory (NWI)

Program has produced coastal wetlands maps for the same time period. However, quantitative inventories of the mapped wetland habitats have not been completed since these maps have not yet been digitized for all parts of the Texas coast. The Galveston Bay area does have total habitat acreage figures for the 1979 photography that were compiled in conjunction with the Galveston Bay National Estuary Program. The USFWS is currently updating the wetland inventory maps based on photography taken in 1992 and 1993, and will produce statistical summaries and areal change estimates of wetland areas for various regions of the coast in addition to updated maps. TPWD has recently completed a special mapping study of 1989 wetlands distribution for Nueces Bay as part of the impact assessment of effects of Choke Canyon Dam on the Nueces River Delta. Similar current habitat maps are needed for the other Texas estuaries, to perform historical trend analyses on Texas coastal wetlands and to evaluate effects of freshwater inflows.

As discussed in Section 5.2, wetland habitat changes caused by freshwater inflow fluctuations will be inadequately assessed if only net changes in "habitat," that is, vegetative ground cover, are measured. Significant wetlands change often occurs which does not involve loss of vegetative ground cover, but rather a shift in species composition of a plant community. These types of habitat alterations require intensive field surveys to document community composition, which is especially difficult to detect by photogrammetric analysis of aerial photography. To make habitat mapping and change-detection results more effective and applicable, target sites should be chosen for regular field monitoring of freshwater inflow effects on habitat. These would be representative areas of vegetative communities in each estuary where established transects would be checked, and samples periodically collected.

The river delta areas of estuaries are most sensitive to effects of salinity intrusion, and flooding of the deltas with fresh water maintains the low-brackish communities there. Because of the critical deficiency in our knowledge of the quantitative inundation requirements of these delta areas, special studies are needed to relate plant community dynamics to delta flooding regimes. To perform this analysis, continuous water level data should be obtained in the estuarine deltas on an ongoing basis as part of regular monitoring programs.

Abundance of major consumers and physiological adaptation studies. To maintain an ecologically sound estuarine system, an understanding of all key components (nekton, benthos, and plankton) and their association with the environment is necessary. Ecologists have developed various methods to study community structure, but in general, each explores only parts of the system and seldom includes all biotic and abiotic components as a whole. The most obvious and simplest way to start is the building of a species inventory, species abundance by habitat type, and information on their ability to cope with the dynamic environment.

The Texas shoreline extends about 400 miles, and each of the ten major and minor estuarine systems is more or less uniquely characterized by different environmental conditions. As a result, species abundance varies from estuary to estuary. In this report, species collected by TPWD gill net, bag seine, and fish trawl have been compiled for the Trinity-San Jacinto, Guadalupe, and Nueces estuarine systems. While benthic communities have been surveyed for the Trinity-San Jacinto and Guadalupe estuaries, species information on plankton in Texas estuaries is scattered and remains to be synthesized. In addition, most of these studies have been done for short time periods in different estuaries. Information spanning several years is needed to determine interannual variability of populations, which is probably linked to interannual climatic variability.

Although there are more than 200 fish and shellfish species in each of the Trinity-San Jacinto, Guadalupe, and Nueces estuaries, the number of species that occurs frequently and abundantly is far fewer. Table 9.2.3 lists the 35 common species that deserve comprehensive studies of their roles and functions in the estuaries. These fish and shellfish are mainly from a few taxonomic groups including families of Ariidae (catfish), Atherinidae (silverside), Clupeidae (menhaden), Cyprinodontidae (killifish), Engraulidae (anchovy), Mugilidae (mullet), Pleuronectidae (flounder), Sciaenidae (drums and croakers), Sparidae (pinfish and sheepshead), Ostreidae (oyster), Portunidae (crab), and Penaeidae (shrimp). The last three groups contain the most valuable members of the invertebrates in Texas.

Bays and estuaries are important nursery grounds for many fish that are estuary-dependent during part of their life cycle. Consequently, estuarine fish communities are often dominated by juveniles which spawn in the passes to the Gulf or in the coastal waters, and then move into estuaries as eggs, larvae, or juveniles. Primary nursery areas in estuaries are emergent vegetated wetlands, open water near river mouth, seagrass beds, and reefs. Depending on species, these habitats may be areas where physiochemical parameters are suitable, foods are abundant, competition among species is less keen, or some degree of protection from predators exists.

The spatial distribution of juvenile fish and invertebrates within an estuary are the results of interactions among physiochemical factors, other species, and themselves. While abiotic factors govern broad spatial distribution, species interactions fine-tune local distributions. Several studies

Table 9.2.3. Species with mean catch per unit effort ≥ 1.0 and % of sample ≥ 25%. Gear type: G=gill net, T=trawl, B=bag seine, D=oyster dredge. Bay system: G=Galveston Bay, S=San Antonio Bay, N=Nueces Bay.

Species	Common name	Gear type				Bay system		
Anchoa mitchilli	bay anchovy		T	B		G	S	N
Archosargus probatocephalus	sheepshead	G					S	N
Arius felis	hardhead catfish	G	T			G	S	N
Bagre marinus	gafftopsail catfish	G				G	S	N
Bairdiella chrysura	silver perch		T					N
Brevoortia patronus	gulf menhaden	G	T	B		G	S	N
Callinectes sapidus	blue crab	G	T	B		G	S	N
Callinectes similis	lesser blue crab		T					N
Crassostrea virginica	eastern oyster				D	G	S	N
Chloroscombrus chrysrus	Atlantic bumper		T					N
Cynoscion arenarius	sand seatrout		T			G		N
Cynoscion nebulosus	spotted seatrout	G				G	S	N
Cyprinodon variegatus	sheepshead minnow			B			S	N
Dorosoma cepedianum	gizzard shad	G				G	S	N
Elops saurus	ladyfish	G						N
Fundulus grandis	gulf killifish			B			S	N
Fundulus similis	longnose killifish			B			S	N
Ictalurus furcatus	blue catfish	G	T				S	
Lagodon rhomboides	pinfish		T	B		G	S	N
Leiostomus xanthurus	spot	G	T	B		G	S	N
Lepisosteus spatula	alligator gar	G					S	
Lolliguncula brevis	brief squid		T				S	N
Menidia spp.	silverside			B		G	S	N
Micropogonias undulatus	Atlantic croaker	G	T	B		G	S	N
Mugil cephalus	striped mullet	G		B		G	S	N
Neopanope texana	mud crab		T					N
Orthopristis chrysoptera	pigfish		T					N
Palaemonetes spp.	grass shrimp			B		G	S	N
Paralichthys lethostigma	southern flounder	G				G	S	N
Penaeus aztecus	brown shrimp		T	B		G	S	N
Penaeus setiferus	white shrimp		T	B		G	S	N
Pogonias cromis	black drum	G				G	S	N
Polydactylus octonemus	Atlantic threadfin		T					N
Sciaenops ocellatus	red drum	G				G	S	N
Sphoeroides parvus	least puffer		T					N

along the Gulf of Mexico indicated that fish abundance in estuaries and coastal waters were highly related to the amount of wetlands, food availability, and climatic factors including river discharge, precipitation, and wind directions during spawning seasons. In Texas, historical data suggested that estuaries with abundant river inflow had higher fish yields per unit area and that fishery harvest was higher in "wet" years than "dry" years.

Relative fish abundances by latitude-longitude and in some occasions, by wet-dry year are available for the Trinity-San Jacinto river system and partially complete for Nueces and Guadalupe estuaries. Since these abundance plots show some degree of relationship among organisms, salinity-river discharge, and wetland area, similar plots should be made available for the same species in other Texas estuaries.

Organisms must be tolerant of changes in salinity and temperature to survive and grow in a varying estuarine environment. Whenever unfavorable conditions are encountered, there are several possible outcomes. For nekton, they either move up and down the estuary, searching for a tolerable location, or physiologically adapt to the conditions if stresses are within tolerable limits. For sessile organisms, they either adjust dynamically to the rapidly changing environments or die if conditions are too rigorous. Usually, there is a wide range within which organisms are able to survive for a short period of time, but there is usually a narrower range within which organisms can achieve maximum growth and reproduction. Different life stages also show different sensitivities to the same conditions. Perhaps, through this differential adaptation, organisms maneuver to optimize growth, reduce competition, and reduce predation rates in a dynamic system.

As a result, the study of physiological aspects in relation to the life cycle of the organisms becomes necessary to understand how organisms are distributed in estuaries or why one habitat is preferred to the other. Without the knowledge of physiological requirements by organisms, the association between species distribution and salinity/habitat can only be assumed or approximated. Aquaculture studies have demonstrated that animal growth can be maintained at salinities outside of the preferred zone if food is supplemented, suggesting that lack of fresh water is not only a physiological stress but also a trophic stress. Freshwater inflow also brings nutrients and allochthonous material that fuels the food web. More research is needed to explore the synergistic connections between salinity stress and trophic relationships.

Undoubtedly, the most difficult problem faced by estuarine organisms is osmoregulation and dealing with salinity variations. Only when this problem is overcome can organisms benefit from the food-rich estuarine environment. There is a lot of literature dealing with salinity and temperature tolerance of organisms in estuaries, but only a few experiments have been designed to cover the entire range encountered in the system. Temperature and salinity often operate synergistically; therefore, carefully planned experiments over the full range of salinity (0 to 50‰) and temperature (5 to 35 °C) are needed for juveniles, subadults, and adults in Texas estuaries. These studies should examine those parameters such as the effects on survival, growth, and reproduction. A suggested list of species in descending ecological and economic priority includes the zooplankton *Acartia tonsa*, grass shrimp (*Palaemonetes pugio*), eastern oyster, Gulf menhaden, bay anchovy (*Anchoa mitchelli*), white shrimp, brown shrimp, blue crab, spotted seatrout, Atlantic croaker, and red drum.

Improvements in the Analytical Procedure and Enhancements to the TXEMP Model

Phytoplankton production with respect to inflow or salinity. Separating direct inflow-related effects (salinity) on phytoplankton and epiphytic algae from effects of local estuarine factors (turbidity, dredging, pollutant discharges, local meteorology) has proven problematical based on data analysis performed "after-the-fact." Since these primary producers depend on nutrient inputs and water clarity for growth, productivity relationships to inflows should be amenable to modeling if sampling and intensive monitoring of these factors are performed according to a properly designed scheme. One of the clearest ways to distinguish inflow effects would be to compare results from distinct "wet" years and "dry" years for each specific estuary. Using total gaged inflow to the estuary, biological data could be lumped into these two "year" categories for comparison. This basic research approach should help to interpret studies of inflow effects on estuarine productivity, especially for phytoplankton, macrophytes, and epiphyte communities.

"Blooms" and excess fouling algae represent deviations from the normal baseline of phytoplankton biomass and benthic primary producers in coastal water bodies. Estuarine environments characteristically develop nuisance blooms of either dinoflagellates or cyanobacteria (blue-green algae), depending on the salinity (Paerl 1988). Oligohaline regions (0 to 5‰) tend to be dominated by cyanobacteria such as *Anabaena* or *Microcystis*, while dinoflagellates such as *Gonyaulax* or *Ptychodiscus* (*Gymnodinium*) which cause red tides, dominate at salinities greater than 5‰. In some waters, other algae such as Chrysophyta may reach bloom proportions causing brown tides. Often, these blooms of nuisance phytoplankton are accompanied by releases of toxic chemicals, reduction of dissolved oxygen (hypoxia and anoxia) in the water column and bottom sediments, and reduction in "desirable" phytoplankton, macroalgae, and submerged vascular plants due to decreased water transparency. Dormant cysts or seed cells are thought to remain viable in coastal sediments for years, only to be triggered into a bloom by appropriate growth conditions.

Such blooms usually are attributed to hydrobiological disturbances including freshwater runoff, water column turnover, thermal stratification, and abnormally high light and temperature conditions (Paerl 1988). Certainly water resource management practices should aim at controlling or moderating blooms through controlling these causative ecological factors once they are understood. Practical, direct methods that decrease undesirable algal populations involve decreasing specific nutrient loadings, reduction in allochthonous (external) organic matter loadings, increasing vertical mixing and water circulation, and controlling N/P ratios to the systems. The relationship of silicon to nitrogen and phosphorus in the nutrient loadings may be a selective factor especially critical to the success of diatoms over cyanobacteria and dinoflagellates in estuarine waters (Hecky and Kilham 1989). Nutrient loading of coastal waters with sewage discharges that are poor in silicon, but relatively high in nitrogen and phosphorus, could stimulate growth of bloom-forming algae, while leading to the elimination of desirable diatoms by competition.

Concerted monitoring of phytoplankton populations and processes, as well as water quality factors, is necessary to detect bloom formation prior to an outbreak. As described above, characteristic species of phytoplankton or overgrowths of epiphytic algae are good indicators of perturbed inflows and poor existing water quality regimes. A variety of parameters can be used to distinguish actual bloom conditions, including chlorophyll levels, primary productivity, oxygen depletion, water color, and turbidity. Bloom conditions have been correlated with excessive inputs of nutrients or reduced circulation in various estuarine systems (Reynolds and Walsby 1975; Kemp et al. 1983). This is significant since, in Sections 4.2 and 4.3, nutrient loadings (of N and P) to many Texas bays currently were reported to be above levels considered limiting to growth of phytoplankton populations. This suggests that a major prerequisite for development of both blooms and eutrophication already exists. Because of the potential for combinations of physical, chemical, and biotic factors to regulate blooms, criteria for estuarine resource management based on the role of freshwater inflows in controlling bloom events should be explored and developed.

Benthic community productivity as an objective function. Productivity of estuarine species in the programming model is represented entirely by the relationship between harvest (a surrogate measure of production) and inflow. Some of the difficulties of using harvest instead of a more direct productivity measurement have been noted earlier in this section. Even if these problems were addressed, the fact remains that only a limited assortment of fishery species—dominated by those of recreational and commercial importance—represents the productivity of the entire estuary. The methodology could be broadened if productivity of other estuarine life were included. This would be consistent with the language in TEXAS WATER CODE 11.147(a), that beneficial inflows provide a sound environment to maintain the productivity of fish, shellfish, and the estuarine life on which they depend.

Various living components of the estuary were examined in chapters 5 and 6 to determine whether there was a demonstrable relationship between their abundance or production and freshwater inflow. The lack of continuous long-term data records hindered the definition of good relationships for several groups, but benthic species seemed to be the most likely to provide an additional production measurement if additional information were collected. Benthic species are generally immobile, so they cannot escape the changes in estuarine waters that occur with fluctuating inflows. Stenohaline (narrowly salt tolerant) members of the benthic community die when salinity conditions deviate too much, and increases in the supply of materials from inflow and the return of moderate salinities usually produce a boom in benthic production and reproduction.

Studies are being continued in two estuaries that may allow definition of an inflow-benthic productivity relationship. The first requirement in defining an objective function for benthic productivity is to demonstrate the mechanism relating inflow and production. Work is currently under way on this topic and the results look promising (Dr. Paul Montagna, University of Texas at Austin, Marine Science Institute, Port Aransas Marine Laboratory; pers. com.).

The second task will be to use the relationship defined above as the basis for analysis of historical benthic data to see whether bay-specific equations can be created. Once the inflow-benthic productivity mechanism is identified, an important step in the second task will be to determine whether additional benthic data is needed for analysis. For some estuaries, there may be inadequate benthic data records with which to prepare a quantitative relationship. If so, a decision will have to be made whether to collect additional data over a several-year period.

If adequate equations can be created, the benthic relationship can be included in the programming model. Since this will constitute an additional objective function, additional policy-level information such as a benthic production constraint will be required.

A final step needs to be added to the overall inflow determination procedure. A comprehensive analysis of predicted salinities from the hydrodynamic and salinity model should be prepared, based on different inflow levels. One of the sets of inflow levels should be the inflow values calculated by the Estuarine Mathematical Programming Model, TXEMP. The salinity patterns calculated by the hydrodynamic and salinity model should be compared with historical salinity data. This will provide a basis for judging the degree of change in the salinity pattern that will result from the inflows calculated by TXEMP.

Information regarding aquatic and terrestrial life forms (plant communities, shellfish, birds, mammals, and other groups), their dependence on specific estuarine conditions required to maintain their health and productivity needs, and their historical distributions should be evaluated and compared to the conditions that will result from the inflows calculated by TXEMP. When appropriate, adjustments to the calculated inflows to mitigate known adverse impacts can be made through consultation with knowledgeable fishery and wildlife scientists.

Include consideration of water quality in programming model. An anticipated extension of the simulation of estuarine salinity dynamics is the simulation of the behavior of other water quality constituents in the estuary. There is a ready application of the existing salinity simulation model in predicting movements of materials which do not undergo rapid transformations, such as some pollutants. Appropriate formulations do exist, however, to enable simulation of dissolved oxygen concentrations, nutrient concentrations, and levels of other materials which are rapidly processed by the biota of the estuary. To realize the useful application of this model to these dynamic constituents, however, data is needed on the rates of biological and chemical processes which affect these materials.

There may be a role for the optimization approach developed here to address future water quality concerns. There are trade-offs to the estuary in a future scenario of higher rates of nutrient loading which may come with increased urbanization of estuarine shores. Increased nutrient loading may bring positive increases in productivity to some estuaries. However, increased nutrient loading also may increase risks of the development of anoxic areas, red tide blooms, or other problems. How these risks weigh against the possible increased productivity depends on many factors, including rates of water exchange, seasonality, and factors which limit the biological community. The framework of the TXEMP Model is uniquely suited to incorporate in a quantitative way our knowledge of the interactions of these various factors. Water quality standards and productivity measures could also be included as targets or controlling parameters. Relationships between loading rates and predicted dissolved oxygen concentrations or other parameters could be used as constraints. It is possible to envision the application of this model to water quality concerns in this way. However, to make it work, more detailed knowledge is required of the best way to express relationships between nutrient loading, pollutant concentrations, and the behavior of the estuarine ecosystem.

Conclusion

The models and methods needed to use the analytical procedure to determine freshwater inflow requirements have been developed. Most of the information about the hydrology of inflowing waters and fishery equations is also available. The models of circulation and conservative transport for several estuaries need to be calibrated, and the nutrient budgets using cumulative flows from these models must still be prepared. Analyses of sediment requirements for the bay systems other than the Guadalupe Estuary will have to be done on a case-by-case basis, probably aimed at determining sediment requirements for maintaining delta wetlands.

Several enhancements to the method were discussed including improved primary productivity relationships and the addition of benthic productivity and water quality components. Because the analytical procedure is somewhat modular, incremental improvements to the analytical procedure as well as new features can be added easily at any time. Some of the techniques and analyses can be applied to other important problems such as the responses of ecosystems to unusual occurrences or deleterious changes from major pollutant spills, eutrophication, or toxic algae blooms. There may be concern over the length of time required for a bay to flush out a pollutant, or the question might be whether currents will sweep a red tide bloom into a bay. The morphometry of passes, the orientation of ship channels, and the volume of freshwater inflows all influence the exchange between major secondary and tertiary bays and the circulation of fresh and salty water within the bays. The models presented here provide a way of combining information on many aspects of estuary hydrodynamics, movements of materials, and ecological processes.

9.3 POLICY DECISIONS THAT MUST BE MADE TO APPLY THE METHODOLOGY

Introduction

In response to statute directives for studies on the effects of freshwater inflows, state scientists and engineers have developed a comprehensive database and methodology for estimating the freshwater inflow needs of Texas bays and estuaries. Since freshwater inflows affect our estuarine (tidal) systems at all basic levels of interaction—physical, chemical, and biological effects—the new method was designed to include at least the minimum needs for each functional level. It also incorporates a technique for optimizing the freshwater inflow needs across all levels of interaction to maintain the ecological integrity of these valuable coastal environments.

The TXEMP Model. The TXEMP Model was cooperatively developed and tested with the Center for Research in Water Resources at The University of Texas at Austin. It allows use of a multiobjective approach to solving the inflow problem and incorporates the statistical uncertainty of correlated relationships between freshwater inflows and resulting bay salinities and fisheries harvests. This is a real advancement in this type of solution technique. Model results are displayed as "performance curves" like the illustrative examples shown in Figure 9.3.1. From these performance curves, decision-makers can select the point that best balances the needs of man and the environment for the benefit of all Texans. As a final check, the freshwater inflow needs calculated by the TXEMP Model are incorporated into the TXBLEND hydrodynamic model to evaluate the overall effects on bay circulation and salinity patterns.

Policy decisions and management objectives. While the logic and equations of the optimization model are built on scientific and engineering analyses, application of the model requires the mathematical expression of all operative constraints, limits, and state resource management objectives. Decisions about these objectives are in the realm of public policy, more than science and engineering. They are

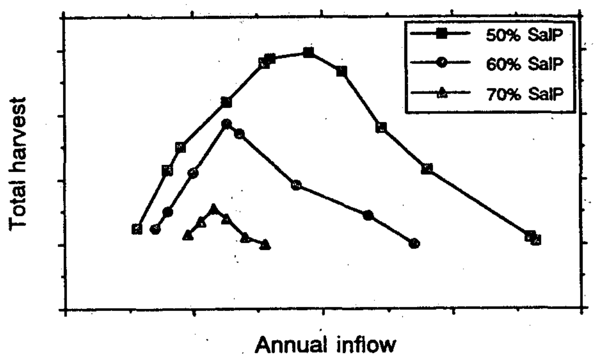

Figure 9.3.1. Example performance curves of harvest versus inflow for different probabilities of meeting the salinity constraints

most appropriately made by the state's policy makers, especially the boards and commissions which are responsible under state statutes for regulating the use and management of bays and estuaries, and their contributing river and coastal drainage basins.

The prototype analysis, prepared jointly by the TWDB and the TPWD and presented in Chapter 8, uses a set of more or less reasonable constraints, limits, and objectives. While these input parameters were sufficient for demonstrating the example, they may not be completely satisfactory for regulatory or judicial purposes. A number of related policy issues must be examined and decisions must be made for this method to be effectively used in the future.

Policy-level Decisions for the Analysis

Species to be included. The indicator organisms that will be used in the analysis must be chosen. At the present time, the choices for this decision are limited. As noted in Section 9.2 and Table 9.2.2, equations relating freshwater inflow to commercial seafood harvest are available for up to seven fishery species in six estuaries, although equations for all species are not available for all estuaries. These species represent a wide spectrum of animals that includes mollusks, crustaceans, and vertebrates. The fishery data were discontinuous in one estuary, and confidence in the resulting equations was low. In the other estuaries, there were only a few instances where no statistically significant equations could be constructed.

Some of the commercial harvest equations may be improved in the future with additional harvest and inflow data. If abundance relationships can be developed from TPWD's fishery-independent monitoring data, it may be possible to provide response equations for more species than just those that are commercially harvested. TPWD's monitoring data can already be used to describe the distributional abundance of many common species, thereby providing another way to biologically interpret results from the hydrodynamic (circulation and salinity) models. Every effort should be made to include species that cover as much of the ecosystem's trophic spectrum as possible.

Relative weighting of species. Also to be decided is the issue of the relative importance of species in the analysis. Should all species be treated equally, or should some be given more preference than others? There may be appropriate policy reasons to justify weighting one species more heavily than another in the analysis. If so, the problem remains of determining what the weighting should be. The prototype analysis includes examples showing results from the use of several different species weightings, including one based solely on the economic value of harvested species. Experience with the TXEMP method has shown that extreme weighting factors are likely to distort the analysis.

Selection of inflow-response equations. Sources of freshwater inflow to the estuary must be considered and will determine which group of inflow-response equations are used, those based on gaged flow alone or the total combined inflow from all contributing drainage basins. Although it is the gaged river flows that are most affected by impoundment and diversion activities, bay salinities and fisheries are affected by inflows to the estuary from all sources. While the use of gaged inflow data alone simplifies the interpretation of the analysis, it may allow additional unexplained error into the optimization procedure. This is particularly true for estuaries with large ungaged drainage areas or where there is a low correlation between gaged and ungaged inflows. For this reason, it may be better to use combined inflows in the model. Another option would be to perform the analyses with both types of inflow records to better evaluate the problem and its solution.

Inflow constraints. Historical inflow information provides a general picture of the volume and patterns of inflow that are characteristic of the river basin. A measure of central tendency is appropriate for use in the model since it provides a realistic estimate of how much water the basin can provide. In the demonstration analysis, the upper bounds on freshwater inflow was set at the median (50th percentile) monthly historical flow, a significantly smaller value than the mean (average) monthly flow, which is skewed upwards by infrequent but large flood events in the historical record. The argument for using the median recognizes that it is more representative of the normal hydrologic conditions since the median inflow is exceeded half the time while the mean inflow is usually encountered much less frequently. For some analyses, however, it may be appropriate to set the upper bound at the mean flow or some other desired level.

After numerous test runs, TPWD and TWDB staff agreed, for demonstration purposes, to use the 10th percentile of the historical monthly flow rate as the lower bounds on inflow. However, existing state or federal minimum streamflow requirements or other considerations could be used to set the required lower bound.

In addition to monthly inflow bounds, seasonal inflow bounds were also set to prevent extrapolating harvests beyond the inflow levels for which the equations are valid (the fishery harvest regressions were derived using two-month seasons in the analysis). Should new types of harvest or abundance equations be prepared, the seasonal period used for the inflow bound may have to be changed to reflect the inflow time span used in the analysis.

Area-specific salinity limits by month. The model provides for the use of monthly upper and lower salinity bounds that are specific to particular areas of the estuary. These bounds represent salinity viability limits within which the economically important and ecologically characteristic fishery species can survive, grow, and reproduce. The number of areas for which salinity bounds are selected will depend on the availability of inflow-salinity regression equations for the estuary; the model can test only the salinity bounds at sites for which there are enough data to develop the equations. Setting the bounds will require interaction between biological scientists, who can interpret the effects of salinity on different life stages at various times of the year, and policy makers or regulators, who have the authority to make the salinity boundary decisions.

Nutrient loading constraint. The nutrient loading constraint (Section 7.4) presented in the example analysis was based on consideration of nitrogen loading to the estuary and the assumption that the minimum acceptable loading should at least balance nutrient losses from the estuary. Indications are that primary production in Texas estuaries is more likely to be limited by nitrogen than phosphorus or carbon. Policy makers may want to consider loading of these other materials, and may wish to refine the minimum acceptable nutrient loading requirement based on information from the nutrient balance analysis.

Sediment loading constraint. Sediment loading and transport are among the least understood of the processes that occur in estuaries. The example analysis included a sediment constraint that was determined in part through a process of elimination. A whole estuary approach, such as used with the nutrient constraint, was not possible. Data availability and other considerations eliminated other specific areas for analysis; maintenance of depth in Mission Lake was one of the few feasible topics left.

Other estuaries have similar problems involving the lack of data and the lack of a clear analytical method. The other estuarine systems will undoubtedly also require an ad hoc approach to sediment load analysis. It may be useful for the policy makers and regulators to evaluate, at the onset, whether such a limited approach is sufficient, and to endorse a general approach such as focusing attention on maintenance of the delta wetlands, as suggested in Section 9.2.

The methods for determining sediment loading requirements for other estuaries are also likely to involve the determination of a minimum sediment load to maintain an elevation, depth, or area. This minimum annual load will represent a balance point at which there will be no net loss or gain. Policy makers may wish to refine this minimum loading requirement based on other considerations such as restoring past losses of certain habitat areas.

Chance constraints for salinity and harvest. This model allows optimal solutions to be calculated that take into account the statistical error (uncertainty) of the salinity and fisheries equations. Using the fishery harvest equation error, the TXEMP model calculations of inflow provide that a given probability level of achieving a particular harvest target will be equaled or surpassed. Using the salinity equation error, the TXEMP model calculations of inflow ensure that a given probability of not violating salinity limits at either the upper or lower bounds will be equaled or exceeded. The probability is usually expressed as a percentage of reliability (usually a value between 50% and about 80%). Policy makers and regulators will have to decide whether to use the stochastic form of the model or to do the analysis in an entirely deterministic manner (setting the probabilities to 50%). If the statistical nature of the salinity or harvest equations were important considerations for the decision at hand, the policy makers or regulators will have to determine the levels of probability that are appropriate for the analysis.

Harvest targets. Harvest targets are the minimum levels of fishery harvests that must be maintained throughout the analysis. Since fishery harvests are used as a surrogate for biological productivity, the harvest targets are important because they define levels of biological productivity that must be maintained by beneficial inflows, which is part of the legislative direction in TEXAS WATER CODE 11.147(a). In the prototype analysis, harvest targets fixed at the mean (average) harvest level for each species proved too restrictive for model operation, so they were reduced to no less than 80% of the means from the data used in the fishery regression analyses. Although this constraint is an important one for policy makers and regulators to set, depending on the management objectives of the estuary, there will have to be some flexibility in selecting the levels to allow the model maneuvering room to find a feasible solution.

Objective function for optimization. Six objective functions for optimization are possible with this model: minimize inflow, maximize harvest, maximize inflow, minimize harvest, maximize probability of achieving harvest targets, and maximize probability of satisfying salinity constraints. The first two objective functions are the ones of most interest in decision-making. Using the 50% salinity probability performance curve in Figure 9.3.1 as an example, the point farthest to the left on the graph represents minimizing freshwater inflows while maintaining fisheries harvests near their mean historic levels. The highest point along the vertical axis of the graph represents the second objective function, maximizing fisheries harvests while maintaining freshwater inflows at levels not to exceed the median historic flows. Solutions between and including these two points will be of greatest interest to the policy and decision makers.

As long as the harvest remains above historic mean levels, the minimum inflow objective function appears to best express the requirements of the statute: determine bay conditions (i.e., sediments, nutrients, and salinity-gradients) adequate to maintain a sound ecological environment necessary for maintenance of the productivity of fish and shellfish (seafood) resources. However, there could be cases where the minimum inflow objective function was not appropriate. These would have to be determined by the policy and decision makers, and another objective or region along the performance curve would have to be selected for inflow decisions.

Conclusion

The policy-level decisions that must be made by state policy makers and regulators to apply this assessment method involve choices about state management objectives, species analyzed, freshwater inflow records, salinity limits, nutrient and sediment loading requirements, fishery harvest targets, and chance constraints on the statistical uncertainties. Some of the decisions are straightforward; several will require interaction with knowledgeable biological or hydrological experts, or specialists operating the model. A few of the policy decisions involve the overall management objectives for an estuary and raise issues of importance to many citizens. It seems appropriate that some guidelines should be established before regulatory use of the model occurs.

9.4 CONCLUSIONS

This study has focused on the effects that freshwater inflows have on the major components of estuarine ecosystems, and the development of a methodology to determine freshwater inflow needs, given natural resource management policies and objectives for an estuary. The tools, data, and knowledge to determine estuarine inflow needs are now available. While these tools and our understanding of estuarine relationships are imperfect, they are complete enough that they may be applied to produce answers. The scientists and engineers who developed the tools and data are confident that the techniques presented here capture the essential relationships between freshwater inflow and the productivity of fish, shellfish, and other estuarine life, and that the levels of beneficial inflows needed to maintain that productivity can be determined. The methods can be applied to Texas estuaries to improve the management of the renewable resources therein, so the resources these estuaries provide will be available for future generations. The same tools serve to guide further investigations and refinement to our understanding estuarine ecology and freshwater inflow requirements.

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