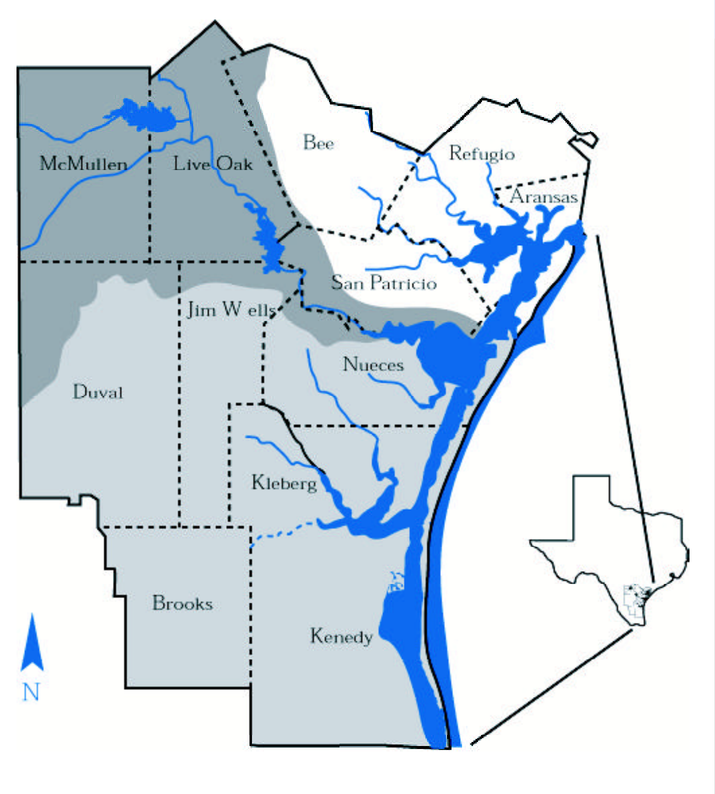
Characterization of Anthropogenic and Natural Disturbance on Vegetated and Unvegetated Bay Bottom Habitats in the CCBNEP Study Area



Volume I: Literature Review

Volume II: Assessment of Scarring in Seagrass Beds

Corpus Christi Bay National Estuary Program
CCBNEP-25 • May 1998

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CHARACTERIZATION OF ANTHROPOGENIC AND NATURAL DISTURBANCE ON VEGETATED AND UNVEGETATED BAY BOTTOM HABITATS IN THE CORPUS CHRISTI BAY NATIONAL ESTUARY PROGRAM STUDY AREA

Volume I:
Literature Review

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CORPUS CHRISTI BAY NATIONAL ESTUARY PROGRAM

The Corpus Christi Bay National Estuary Program (CCBNEP) is a four-year, community based effort to identify the problems facing the bays and estuaries of the Coastal Bend, and to develop a long-range, Comprehensive Conservation and Management Plan. The Program's fundamental purpose is to protect, restore, or enhance the quality of water, sediments, and living resources found within the 600 square mile estuarine portion of the study area.

The Coastal Bend bay system is one of 28 estuaries that have been designated as an **Estuary of National Significance** under a program established by the United States Congress through the Water Quality Act of 1987. This bay system was so designated in 1992 because of its benefits to Texas and the nation. For example:

- Corpus Christi Bay is the gateway to the nation's sixth largest port, and home to the third largest refinery and petrochemical complex. The Port generates over $1 billion of revenue for related businesses, more than $60 million in state and local taxes, and more than 31,000 jobs for Coastal Bend residents.
- The bays and estuaries are famous for their recreational and commercial fisheries production. A study by Texas Agricultural Experiment Station in 1987 found that these industries, along with other recreational activities, contributed nearly $760 million to the local economy, with a statewide impact of $1.3 billion, that year.
- Of the approximately 100 estuaries around the nation, the Coastal Bend ranks fourth in agricultural acreage. Row crops -- cotton, sorghum, and corn -- and livestock generated $480 million in 1994 with a statewide economic impact of $1.6 billion.
- There are over 2600 documented species of plants and animals in the Coastal Bend, including several species that are classified as endangered or threatened. Over 400 bird species live in or pass through the region every year, making the Coastal Bend one of the premier bird watching spots in the world.

The CCBNEP is gathering new and historical data to understand environmental status and trends in the bay ecosystem, determine sources of pollution, causes of habitat declines and risks to human health, and to identify specific management actions to be implemented over the course of several years. The 'priority issues' under investigation include:

- altered freshwater inflow
- declines in living resources
- loss of wetlands and other habitats
- bay debris
- degradation of water quality
- altered estuarine circulation
- selected public health issues

The **COASTAL BEND BAYS PLAN** that will result from these efforts will be the beginning of a well-coordinated and goal-directed future for this regional resource.

STUDY AREA DESCRIPTION

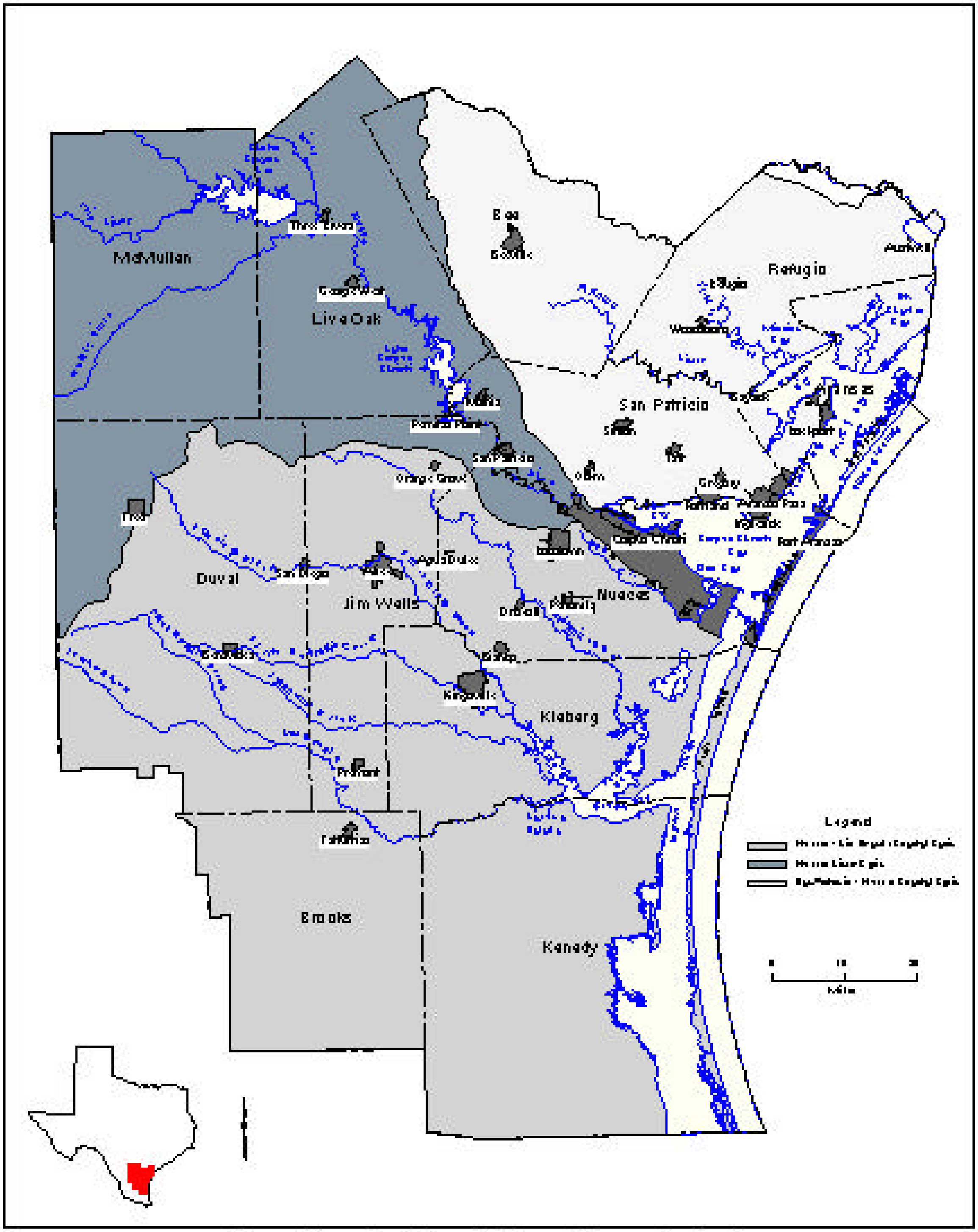
The CCBNEP study area includes three of the seven major estuary systems of the Texas Gulf Coast. These estuaries, the Aransas, Corpus Christi, and Upper Laguna Madre are shallow and biologically productive. Although connected, the estuaries are biogeographically distinct and increase in salinity from north to south. The Laguna Madre is unusual in being only one of three hypersaline lagoon systems in the world. The study area is bounded on its eastern edge by a series of barrier islands, including the world's longest -- Padre Island.

Recognizing that successful management of coastal waters requires an ecosystems approach and careful consideration of all sources of pollutants, the CCBNEP study area includes the 12 counties of the Coastal Bend: Refugio, Aransas, Nueces, San Patricio, Kleberg, Kenedy, Bee, Live Oak, McMullen, Duval, Jim Wells, and Brooks.

This region is part of the Gulf Coast and South Texas Plain, which are characterized by gently sloping plains. Soils are generally clay to sandy loams. There are three major rivers (Aransas, Mission, and Nueces), few natural lakes, and two reservoirs (Lake Corpus Christi and Choke Canyon Reservoir) in the region. The natural vegetation is a mixture of coastal prairie and mesquite chaparral savanna. Land use is largely devoted to rangeland (61%), with cropland and pastureland (27%) and other mixed uses (12%).

The region is semi-arid with a subtropical climate (average annual rainfall varies from 25 to 38 inches, and is highly variable from year to year). Summers are hot and humid, while winters are generally mild with occasional freezes. Hurricanes and tropical storms periodically affect the region.

On the following page is a regional map showing the three bay systems that comprise the CCBNEP study area.



Corpus Christi Bay National Estuary Program Study Area

VOLUME I. LITERATURE REVIEW
TABLE OF CONTENTS

LIST OF FIGURES

LIST OF TABLES

ACRONYMS

CCBNEP	Corpus Christi Bay National Estuary Program
CPUE	Catch-per-unit-effort
DGPS	Differential global positioning system
EPA	U.S. Environmental Protection Agency
GCPS	Ground control points
GIWW	Gulf Intracoastal Waterway
GIS	Geographic Information System
JFK	John F. Kennedy Causeway
MSL	Mean sea level
NBS	National Biological Service
NMFS	National Marine Fisheries Service
NOAA	National Oceanic and Atmospheric Administration
STAC	Science and Technical Advisory Committee of the CCBNEP
TGFOC	Texas Game, Fish and Oyster Commission
TNRCC	Texas Natural Resource Conservation Commission
TWDB	Texas Water Development Board
TPWD	Texas Parks and Wildlife Department
ULM	Upper Laguna Madre
USGS	U.S. Geological Survey
UTMSI	University of Texas Marine Science Institute

GLOSSARY

Anaerobic	Said of an organism, or life process, that does not utilize, or cannot exist in, the presence of oxygen.
Anoxic	Zero dissolved oxygen.
Anthropogenic	Refers to influences caused by man, e.g. cultivation.
Azoic	No living organisms present.
Benthic	Refers to the bottom of any body of water.
Benthos	Organisms which live on or in the bottom of the ocean of bodies of fresh water, from the water's edge down to the greatest depths.
Biocide	Compounds that can kill or harm living things.
Biodeposition	Depositing of organic matter of biotic origin on the sea floor
Biodiversity	Diversity
Biomass	The total quantity of all the species in a community.
Biotic	Living components of the ecosystem.
Bioturbation	Alteration of substrate through burrowing and feeding activities of benthic organisms such as crabs, polychaetes, and molluscs.
Brines	Discharged fluid from production well high in salt content.
Bycatch	Non-targeted organisms caught in trawls.
Commensal	One of the organisms living together with benefit usually to one and with injury to none.
Contaminant	Materials discharged into the environment by man's activities.
Correlation	Statistical relationship between two variables.
Demersal	Swimming organisms.
Detrital	Origin in detritus, or dead organic matter.
Diatoms	One-celled plants.
Disturbance	Abrupt change in the environment.
Diversity	Different kinds (or species) of organisms in a community.
Dredging	Digging of or in sediment from the bottom.
Ecosystems	The community, including all the component organisms together with the abiotic environment, forming an interacting system.
Effluent	The outflow of water from subterranean storage.
Epifauna	Organisms living on or near the sediment surface.
Euryhaline	Able to live in waters of a wide range of salinity.
Eustatic	Characterized by a worldwide change of sea level.
Eutrophication	Nutrient enrichment to the point that algal blooms occur.
Grain sizes	Sediment is composed of grains size classes defined as: Clay: <2 µm. Silt: 2 to 50 µm. Sand: 50 to 200 µm. Rubble (cobble and gravel): > 2000 µm.
Hydrocarbon	Compounds made of carbon and hydrogen only, such as, oil or petroleum.
Hydrographic	Physical-chemical conditions of the water.
Hypoxic	Low oxygen conditions.

Herbivores	Consumes plants, either unicellular or multicellular, that are primary producers.
Heterogeneous	Consisting of dissimilar or diverse ingredients.
Hydrology	The science dealing with water and snow, including their properties and distribution.
Hypersalinity	Salinities greater than that of seawater
Infaunal	Living in sediments.
Inflow	Freshwater that flows into an estuary. Sources can be diverse including river, rain, treated effluent, and runoff.
Invertebrates	An animal lacking a spinal column, e.g., insects.
Isoleth	A line drawn on a map or chart connecting places having the same value of a certain factor.
Macrofauna	Benthic organisms greater than 0.5 mm in size.
Meiofauna	Benthic organisms greater than 0.063 mm, but smaller than 0.5 mm in size.
Metazoan	Multicellular organisms.
Models	A description or analogy of something that can not be visualized. A system of postulates, data, or inferences presented as equations to describe or simulate a system.
Morphology	Study of form.
Nektonic	Strong swimming of animals in water.
Normoxic	Normal oxygen conditions.
Omnivores	A diet of both plants and animals.
Osmotic	Properties of osmosis, which is movement of a solvent through a semi-permeable membrane.
Oxygenation	Introduction of oxygen.
Parasites	Organisms that live in or on other organisms.
Pelagic	Inhabiting the mass of water of sea or lake in contrast to the sea or lake bottom.
Photosynthesis	Utilization of light energy to produce carbohydrates.
Phytoplankton	One-celled plants in the plankton.
Planktonic	The floating or weakly swimming animal and plant organisms occurring at any depth in lakes, ponds, streams, or seas: often microscopic in size.
Pollutants	A substance that makes the environment unpure or unclean.
Resuspension	Sediment or sediment particles in suspension in the water column.
Sediment	Bulk and water phase of sea floor composed of: interstitial water (about 50% by volume), inorganic phases, e.g., rock fragments and minerals (about 48% by volume), organic matter (about 1% by volume), and anthropogenically derived materials (<1% by volume).
Sessile	Organisms attached to a surface, or not free to move about.
Scarring	Tracks made by boat propellers is the sediment.
Silt	See grain size.
Spatial	Relating to space.
Stability	State of being stable or unchanging.
Stratification	The appearance of plants or plant parts, or their remains, in horizontal divisions.

Stress	Systemic. The condition of an organism where large parts of the body deviate from their normal resting state, either because of their activity or because of an injury.
Substratum	Layer beneath surface soil.
Succession	The replacement of one kind of a community by another kind.
Symbiotic	Living together in mutually beneficial relationship.
Synergistic	Interaction of discrete agents where total effects is greater than sum of effects.
Temporal	Relating to time.
Tertiary	Third order, as in a bay removed from the ocean by two other bays.
Trawling	Fishing by dragging a large net along the bottom.
Trophic	Refers to nutrition. Strata of a food chain.
Turbidity	The condition of a body of water that contains suspended material such as clay or silt particles, dead organisms or their parts, or small living plants and animals.
Xenobiotic	A chemical compound that is foreign to a living organism.

PREFACE

The purpose of this report is to characterize the impact of anthropogenic and natural disturbances on estuarine benthic habitats within the Corpus Christi Bay National Estuary Program (CCBNEP) study area. In general, all CCBNEP characterization reports include the following items: 1) determination of the current status and trends in the physical, chemical, and biological characteristics of the study area; 2) identification of probable causes of these trends through correlation with status and trends of human activities and natural events; and 3) identification of missing data critical to the management and monitoring of the CCBNEP study area. Information presented in this report will aid in identification of appropriate data acquisition programs and management activities needed to assure long-term maintenance of benthic habitats within the CCBNEP study area.

Three objectives were established to meet the project's goal of characterizing estuarine benthic disturbances in the CCBNEP study area: A) to assess relative contribution of anthropogenic and natural disturbances on benthic habitats; B) to quantify aerial extent of propeller damage in submerged aquatic habitats; and C) to determine effects of anthropogenic disturbances on biological processes.

Objective B is fulfilled by the acquisition of new data, while objectives A and C are fulfilled by a literature review. Because there are different methods and scopes of work in a review and a new field project, this report is prepared in two volumes. Volume I fulfills objectives A and C and Volume II fulfills objective B.

ACKNOWLEDGMENTS

The authors wish to thank the many people who helped in the preparation of the literature review presented in Volume I. Hudson DeYoe and Sandra Alvarado facilitated program management for the CCBNEP, Sheila Meyers provided quality control advice, and Laura Radde provide suggestions on improving the work plan. We also thank Mr. Lonnie Jones (Human Resources Report) for passing along dredge data, and the Port of Corpus Christi for recent shipping data. Several University of Texas Marine Science Institute staff were helpful in preparing the report: Heather Alexander provided input on the seagrass section, and Greg Street provided input on the bay bottom section.

The authors also had considerable help in preparing Volume II. We thank the GIS team at the Conrad Blucher Institute for Surveying and Science, led by Dr. Gary Jeffress, for their valuable and considerable contribution of time and equipment to this project. In particular we thank Oscar Cantu, Kirk Rutkofske and Dan Prouty for both their intensive efforts and valuable friendships, especially during the final hectic days of final report preparation.

Finally, the manuscript was improved considerably by suggestions made by the project review team. The authors thank Pam Baker, Paul Carangelo, Robyn Cobb, Billy Fuls, David Jensen, Larry McEachron, Laura Radde, Windle Taylor, and Kim Withers for their time and attention to editorial detail.

Table of Contents

CHARACTERIZATION OF ANTHROPOGENIC AND NATURAL DISTURBANCE ON VEGETATED AND UNVEGETATED BAY BOTTOM HABITATS IN THE CORPUS CHRISTI BAY NATIONAL ESTUARY PROGRAM STUDY AREA

VOLUME I. LITERATURE REVIEW

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EXECUTIVE SUMMARY

Benthic (i.e., bottom dwelling) communities are a dominant feature in the shallow, well-mixed estuaries of South Texas, and play a very important role in the functioning of these ecosystems. Benthos (a Greek word meaning “bottom dwelling”) encompasses creatures that dwell in or near the bottom of all fresh water, estuarine, and marine environments. Small invertebrates in the benthos are often referred to as “the invisible fauna” because they are hard to see, but are very important in marine ecosystems, especially shallow estuaries. Benthic habitats are important because they store energy for the entire ecosystem and regulate or modify most physical, chemical, geological, and biological processes. This energy is in the form of detritus, and becomes available when nutrients are regenerated by detrital decomposition. In this way, sediments act as the “memory” of the estuarine ecosystem. Recycled nutrients are key elements which support primary production in the overlying water column. Benthic organisms live in or on the sediment and are important in almost all aspects of living marine resources. Benthic organisms are important food sources for many bottom feeding fish and wildlife, such as drums and wading birds. Most benthic invertebrates have planktonic larval stages; these larvae are important food sources for planktonic and nektonic organisms. The benthos supplies economic resources and food for humans in the form of shellfish, such as shrimps, crabs, oysters, mussels, and clams. In summary, benthos are key components of the environmental health of shallow-water marine ecosystems.

Benthic organisms are especially useful in applied research. Benthos are usually the first organisms affected by pollutants or environmental degradation. Benthic invertebrates are sensitive to pollutants. Due to gravity, everything ends up in bottom sediments. Even pollutants in freshwater will be transported to the coastal sea bottoms. Everything dies and ends up in the detrital food chain, which is utilized by the benthos. Pollutants are usually tightly coupled to organic matrices, therefore benthos have great exposure through their niche (food) and habitat (living spaces) to pollutants. Benthos are relatively long-lived and sessile, so they integrate pollutants effects over long temporal and spatial scales. Bioturbation and irrigation of sediments by benthos affect mobilization and burial of xenobiotic materials. There are also ecological models that provide a scientific basis for interpreting data generated in benthic monitoring and detection studies. These approaches utilize many single species, community studies, and statistical models. One of the most important concepts is the succession model, which posits that normal sediments will have a diverse assemblage of

deeper dwelling organisms than a polluted or disturbed environment. In summary, benthos are well known indicators of environmental change in marine ecosystems.

Benthic habitats can be broadly classified as either vegetated or unvegetated. Seagrasses are the dominant form of submerged vegetation. Vegetated bay bottoms are generally more productive than unvegetated bottoms. Over 85% of seagrasses in Texas occur in estuaries of the Corpus Christi Bay National Estuary Program (CCBNEP) study area. Seagrass ecosystems are noted for high commercial and sport fishery landings. This correlation is often attributed to the high primary and secondary productivity associated with large seagrass beds in the CCBNEP study area. Disturbances, operating at many different scales, however, can profoundly affect benthic processes in estuaries and threaten fishery resources. For example, upper Laguna Madre appears to be undergoing a dynamic transition from a seagrass-dominated to a phytoplankton-dominated system as a result of a brown tide algal bloom. This large scale disturbance has resulted in pronounced changes in the trophic dynamics and relative importance among key plant and animal species within the system. What is not known is if the disturbance in Upper Laguna Madre is natural or influenced by man's activities, or if there are interactive effects due to natural fluctuations in the environment and anthropogenic input. Therefore, there is a need to identify natural and anthropogenic disturbances and characterize effects on benthic habitats.

The four most important benthic habitats in the CCBNEP study area are: seagrass beds, oyster reefs, open bay muddy bottoms and shoreline sandy bottoms. Disturbances to benthic habitats are well known. Seagrasses are stressed by nutrient enrichment, propeller scarring, and especially light reduction caused by brown tide, turbidity, and dredging. Oyster reefs are stressed by reduced freshwater inflows and concomitant higher salinities. Bay bottom habitats are a victim of their own bounty. The large economic benefit the CCBNEP region derives from shrimping and sport fishing, which is supported by bay bottom habitats, is at risk due to mechanized harvest. Trawling continually turns over sediment, keeping the benthos in a continual state of low abundance and biodiversity. Bycatch removes potential food for recreationally and commercially important fish species.

Numerous disturbances resulting from human activities and natural events that affect physical, chemical, and biological characteristics of estuarine benthic habitats have been identified within the CCBNEP study area. Examples of human activities include: shrimp harvesting with bottom trawls, commercial tug and barge operations, recreational boating, dredging, altered currents, altered inflow, nutrient and contaminant input, and hydrocarbon exploration and production. Each activity has variable effects on the different types of bay bottom habitats. Examples of natural disturbances include: wind-generated resuspension and deposition of fine sediments, abrupt salinity changes due to direct rainfall and runoff, erosion due to storms, fish mortality due to freezes, long-term climatic changes, and harmful algal blooms.

The major finding of this report is natural and anthropogenic disturbances are synergistic. That is, the net effect of both disturbances is greater than the sum of each disturbance. Synergistic interactions between multiple stressors (including both natural and anthropogenic disturbances) are very important, but quantification of these effects represents the largest data gap. Each kind of anthropogenic disturbance is important in specific sites where the disturbance occurs and if the disturbance affects an ecological process or community that is present in the site. Overall, reduced inflows appear to be the most severe problem, because the CCBNEP study area is already severely stressed by a naturally arid climate, which produces naturally low inflow rates to the estuaries. Effects of low inflow rates (and resulting high salinities) are low biodiversity and low benthic secondary production. Effects of low inflow rates are exacerbated by the naturally small tidal range and high residence times of water in the bays. Altered circulation further restricts flow rates and exacerbates effects of high salinities. This may be the cause of hypoxia in a portion of Corpus Christi Bay. Whereas altered inflows and circulation affect the entire study area, the main effects of two activities appear to be restricted. Dredging is of concern mostly in seagrass habitats. Shrimping is a concern mostly in open primary bays. Again, both activities occur in the setting of altered inflows and circulation, which increases net effects of both activities due to synergistic interactions. The largest potential source of environmental degradation is from accidents during transportation of oil products in pipelines , barges, and tankers. Estuaries are very valuable, sensitive environments, and long-term catastrophic effects have resulted due to accidents in other parts of the world. Although accidents are rare in the CCNBEP study area, habitat degradation due to produced water discharge is a current problem. Finally, further characterization and understanding of the effects of natural and anthropogenic disturbances on benthic habitats within the CCBNEP study area are needed to develop management strategies of South Texas estuarine resources.

Table of Contents

I. INTRODUCTION

A. Importance of Bay Bottom Habitats

Benthic (i.e., bottom dwelling) communities are a dominant feature in the shallow, well-mixed estuaries of South Texas, and play a very important role in the functioning of these ecosystems. Benthos (a Greek word meaning "bottom dwelling") encompasses creatures that dwell in or near the bottom of all fresh water, estuarine, and marine environments. Small invertebrates in the benthos are often referred to as "the invisible fauna" because they are hard to see, but they are very important in marine ecosystems, especially shallow estuaries. Benthic habitats are important because they store energy for the entire ecosystem and regulate or modify most physical, chemical, geological, and biological processes in the ocean. This energy is in the form of detritus, and becomes available when nutrients are regenerated by detrital decomposition. In this way, sediments act as the "memory" of the estuarine ecosystem. Recycled nutrients are key elements that support primary production in the overlying water column. Benthic organisms live in or on the sediment and are important in almost all aspects of living marine resources. Benthic organisms are an important food source for many bottom feeding fish and wildlife, such as drum and wading birds. Most benthic invertebrates have planktonic larval stages, and these larvae are important food sources for planktonic and nektonic organisms. Finally, the benthos supplies economic resources and food for humans in the form of shellfish, such as shrimp, crabs, oysters, mussels, and clams.

Analysis of benthic invertebrate communities has been widely used in pollution detection and monitoring studies. We expect indicator organisms to do for us today what canaries did for miners in the 18th and 19th century. Indicator organisms should have the following characteristics (Soule, 1988): 1) they should direct our attention to qualities of the environment, 2) they should give us a sign that some characteristic is present, 3) they should express a generalization about the environment, 4) they should suggest a cause, outcome or remedy, and 5) they should show a need for action. Benthic organisms have been especially useful in applied research and can be used as indicator organisms. Benthos are usually the first organisms affected by pollution. There are several reasons why benthos are good indicators of environmental stress. 1) Gravity insures that everything eventually ends up in bottom sediments. Even pollutants in freshwater will be transported to the sea bottoms. 2) Everything dies and ends up in the detrital food chain, which is utilized by the benthos. Pollutants are usually tightly coupled to organic matrices, therefore benthos have great exposure through their niche (food) and habitat (living spaces) to pollutants. 3) Benthos are relatively long-lived and sessile, so they integrate pollutants' effects over long temporal and wide spatial scales. 4) Benthic invertebrates are sensitive to pollutants. 5) Bioturbation and irrigation of sediments by benthos effect mobilization and burial of contaminated materials.

There are ecological theories and models that provide a scientific basis for interpreting data generated in benthic monitoring and detection studies. These approaches utilize many single species, community studies, and statistical models. One of the most important concepts is the succession model proposed by Rhoads et al., (1978). They applied scientific theories of ecological succession and its relation to productivity to suggest ways that dredge-spoil could be managed to enhance productivity. One important aspect of this theory is that normal sediments will have a more diverse assemblage of deeper-dwelling organisms than polluted or disturbed environments. Thus, we have a scientific justification for biological diversity studies. Since this classic study, numerous other studies have demonstrated that benthic biological diversity is an excellent indicator of environmental health.

B. Identification and Description of Natural and Human-Induced Disturbances

B.1. Natural resources

The CCBNEP study area is rich in natural resources. Perhaps the most economically important living resources are commercial and recreational fisheries, but oil and gas reserves beneath bay bottoms are also economically important. Oil and gas reserves are not discussed in this report, because they are non-living, subsurface resources. However, extraction of oil and gas often results in benthic disturbances from dredging, construction, and drilling.

Benthos can be defined in four different ways: by size, sub-habitat, sediment preference, or feeding type. There are four size distributions: megafauna (> 2 mm), macrofauna (> 0.5 mm), meiofauna (> 0.063 mm, but <0.5 mm), and microfauna (<0.063 mm). There are three sub-habitats: motile epifauna (which move about the sediment surface), infauna (which live in the sediment), and interstitial fauna (which live between the grains of sand). It is common for organisms to be restricted to or prefer either muddy, sandy, or hard bottoms. There are six feeding types: 1) suspension feeders that are non-selective trappers, 2) filter feeders that are a subset of suspension feeders using filters to capture particles from the water, 3) deposit feeders that ingest sediment and can either be non-selective or selective, 4) omnivores that are either raptors or scavengers, 5) herbivores that browse the surface of plants or eat diatoms, and 6) parasites and commensals that live in association with other animals.

Benthic epifauna are dominated by crustaceans (primarily shrimp and crabs), and some fishes. Densities range from <1 to over 100 per square meter. Benthic macroinfaunal communities are dominated by polychaetes, crustaceans (mostly amphipods), and molluscs (mostly bivalves and gastropods). Densities range from 2,000 to 100,000 per square meter, and biomasses range from 0.1 to 20 grams dry weight per square meter (Montagna, unpublished data). Meiofauna communities are dominated by Nematoda and Harpacticoida (Copepoda), but also include many other metazoan taxa (e.g., Hydroida, Turbellaria, Gnathostomulida, Gastrotricha, Oligochaeta, Halacaridae, and Bryozoa). Densities range from 500,000 to 2,000,000 per square meter, and biomass is about 0.1 to 1 grams dry weight per square meter. Smaller meiofauna have faster turnover rates than macrofauna, so productivity for the two groups is about equal. Benthic communities can influence physical and sediment properties by bioturbation, oxygenation of anaerobic sediments, carbon depletion, production of binding and stabilization agents (mucus), and biodeposition. Benthos can also reduce effects of eutrophication in estuaries by filter feeding on phytoplankton and other particulate matter.

Many benthic animals are economically important commercial and sport fishery resources. Examples include shrimp (*Penaeus* spp.), blue crab (*Callinectes sapidus*), eastern oysters (*Crassostrea virginica*) and flounder (*Paralichthys* spp.). These animals depend on organic resources in close association with the bay bottom. Other fishery resources, such as Atlantic croaker (*Micropogonias undulatus*), spot (*Leiostomus xanthurus*), and black drum (*Pogonias cromis*), depend on benthic animals as food. For example, adult black drum will eat bivalves (e.g., *Mulinia lateralis*). Juvenile spot, Atlantic croaker, and black drum, consume benthic mysids and harpacticoid copepods. Relationships between these fishery resources and benthic organisms will be discussed in further detail in section IV.

Benthic communities are dependent on benthic habitats. Habitats are elements of an environment that sustains a population or community. The three most important benthic habitats in the CCBNEP study area are: seagrass beds, open bay bottoms, and oyster reefs. Subtidal seagrass habitats support the richest benthic community, followed by oyster reefs, and bay bottom. The two richest habitats (seagrass beds and oyster reefs) have benthic structure in common. That is, where there are places to attach, or where structural complexity exists, certain habitat characteristics are amplified. In particular, food webs are more diverse and structure provides a refuge from predators. Open bay habitats are very extensive in the CCBNEP study area and are much more heterogeneous than they appear. This is due to the interactions of different sediment types, salinity regimes, and water depths (Mannino and Montagna, 1996; 1997).

A more detailed description of the living natural resources in the CCBNEP study area can be found in Tunnell et al. (1996). A complete description of benthic, as well as other, habitats in the CCBNEP study area can be found in Montagna et al. (1996).

B.2. Natural disturbances

Storms, floods, droughts, and freezes are examples of natural disturbances that affect estuarine biota. Disturbances by large storms can result in potentially massive sediment redistribution. But, storm events often have variable impacts on estuarine and coastal biotic communities. For example, Hurricane Andrew had no significant impact on seagrass beds in South Florida (Chris Dawes, University of South Florida, personal communication), but Hurricane Gilbert did impact turtle grass (*Thalassia testudinum*) communities in Puerto Morelos, Mexico (van Tussenbroek, 1994). Thus, the effect of large scale disturbances on benthic and epibenthic plants and animals can be quite different depending on a number of factors, e.g., storm frequency, intensity, and the local community present. Storms are also stochastic events that cannot be predicted and a changing climate may also affect storm frequency and intensity.

Turbidity, sedimentation, and bioturbation are other natural disturbances. Shading due to high turbidity can limit photosynthesis, and thus primary production. Sedimentation may bury organisms whereas turbidity may hinder suspension feeding organisms. Additionally, bioturbation can either hinder or enhance life styles of other animals. Turbidity and sedimentation are often associated with storm events, but also result from freshwater inflow and wind mixing. Sediment carried by freshwater flowing into bays and estuaries may be kept in suspension or resuspended by wind mixing. Turbidity is caused by the sediment load of the freshwater and the resuspension of sediment by wind and tidal mixing. Bioturbation results from the activities of benthic animals (e.g., burrowing, locomotion, and feeding). Unlike storms, all of these particulate related events are predictable, if the community structure, and certain rates (e.g., river flow) are known. Storms, however, can contribute both to the timing and intensity of resuspension events since high winds are often associated with storms.

B.3. Human influences

Numerous human influences exist within the CCBNEP area. Although nearly 75% of the land within the CCBNEP area is used for agriculture, many people live on the bays' borders. Corpus Christi (population over 250,000) is the largest city in the area and the largest city in Texas situated directly on the shores of a major bay. Therefore, the Nueces Estuary (which includes Corpus Christi and Nueces Bays) is generally more impacted that the Mission-Aransas Estuary and the Baffin Bay-Laguna Madre Estuary.

There are a variety of human uses of the bays within the CCBNEP study area (Montagna et al., 1996). The Gulf Intracoastal Waterway (GIWW) and the Corpus Christi Ship Channel are used for shipping, tug, and general maritime transportation operations. Shrimp, a natural resource associated with benthic habitats, supports a commercial fishing industry. Finfish such as, black drum (*Pogonias cromis*), red drum (*Sciaenops ocellatus*) and spotted seatrout (*Cynoscion nebulosus*) utilize benthic habitats and support an extensive recreational fishing industry. Oil and gas wells are regionally important economic resources that provide energy to support the nation's economy.

B.4. Anthropogenic disturbances

Anthropogenic disturbances have potential to greatly disrupt benthic ecosystems. Generally, these ecosystems are adapted to cyclic natural phenomena such as seasonal storms and climatic fluctuations. Organisms are also generally adapted to natural perturbations such as high winds and tides, and pulses of freshwater inflow. In contrast, human disturbances are generally continuous, non-cyclic events (e.g., trawling) for which organisms are not adapted. Some events are episodic, (such as oil and chemical spills) and can be toxic. Anthropogenic disturbances that impact bay bottom habitats include a variety of activities related to marine transportation, commercial and recreational fishing and tourism. Frequency of these activities and their impact increases with increasing human populations and human use of the ecosystem.

The Port of Corpus Christi is the sixth largest port in the United States, making marine transportation a dominant industry in South Texas. All ship traffic enters the CCBNEP study area via Aransas Pass, located in the center of the CCBNEP study area. Ship traffic destined to the Port of Corpus Christi crosses Corpus Christi Bay via the Corpus Christi Ship Channel. Barge traffic mainly uses the ICWW. Corpus Christi Bay is only 3-4 m deep while the Corpus Christi Ship Channel is dredged to a depth of 15 m. Commercial maritime traffic includes tankers, container ships, grain ships, barges, and associated tugboats. Another important maritime industry is the offshore supply and crew boats, which are primarily ported in Port Aransas and Aransas Pass. Commercial shipping operations do not directly influence the benthos. However, chemical discharges, often related to shipping and tug activities, may disturb benthic habitats and ship wakes can cause erosion or disturbance in shoals. The largest influence due to maritime traffic is maintenance dredging of channels and marine construction. Dredging disrupts benthic processes during digging and deposition of dredged material. The main concerns are death of organisms, habitat loss through dredging or burial, and increased turbidity caused by resuspension of fine sediments.

Hydrocarbon exploration and production is a very large business in South Texas, much of it occurring in public, submerged lands. Bay bottoms are subject to disturbance during all phases. Exploration often begins with seismic testing. Seismic testing causes large holes in the sediment, this is potential problem in rich habitats, e.g., seagrass beds. Drill cuttings often contain toxic amounts of heavy metals which can cause toxicity in sediment communities. Construction of semi-permanent production wells has two opposing effects: toxicity and habitat enrichment. There may not be a general net effect of production, because it is often a function of local environmental conditions. The largest concern with hydrocarbon production is with transportation. Pipelines leak or break, and accidental spills occur. Spills can be numerous, but small as occurs during loading and offloading operations, or large as when ships or barges ground or collide.

There are five primary commercial fisheries in South Texas: black drum, flounder, shrimp, oyster, and crab. Shrimping constitutes the largest commercial fishery in Texas, and there are several potential effects from activities related to shrimping. The shrimp fishery consists of the bait fishery and the food shrimp fishery. Bait shrimp are captured over open bottom with bottom trawls and are occasionally caught over seagrass beds using small boats equipped with push nets. Food shrimp are harvested only with bottom trawls. Trawls capture many other animals besides the targeted shrimp. These non-targeted organisms are bycatch. Trawls also disturb the bay bottom, leading to sediment resuspension and localized higher turbidity. It is possible that both the digging or turning over of sediment and resulting turbidity are causing biological effects. In addition, endangered species such as sea turtles are caught in trawls. This has led to adoption of regulations requiring use of turtle excluder devices in trawls. Oyster harvesting produces significant turbidity and the oyster boats may disturb the bottom near the relatively shallow reefs. The potential impact of the crabbing industry is disturbance or destruction of seagrasses when crab pots are placed in seagrass meadows.

Tourism and recreational fishing have increased during the last ten years. There has been an increased demand for bait shrimp, bait fish, marinas, and lodging associated with these industries. The main affects of these activities are related to bait shrimping, marine construction, and small localized oil spills in marinas. Recreational boating may disturb the benthos through propeller scarring of bottom habitats, especially seagrass beds. Tourism is increasing as use of the coastal zone increases. Therefore, it is likely that tourism related impacts will increase in the future.

C. Ecological Disturbance and Recovery Theory

Through time, populations and communities recover from disturbances via succession, a process studied by ecologists for more than 60 years. Numerous models describe ecological succession and the role of disturbance in determining community structure. Initially, these models were created for, and applied to, terrestrial ecosystems (Cooper, 1939; Keever, 1950; Krebs, 1972; Odum, 1969). Later, these models were refined for application to marine and estuarine ecosystems (Dauer, 1993; Pearson and Rosenberg, 1978; Rhoads et al., 1978). An inherent aspect of succession is initiation by some sort of ecosystem or community disturbance. Several generalities can be drawn from succession models to distinguish early stages of succession from later stages. For example, early succession is characterized by low diversity, opportunistic species, simple food chains and poor nutrient conservation. Late succession is characterized by high diversity, specialized slow- growing species, complex food web, and good nutrient conservation.

In general, when an ecosystem is severely disturbed, existing fauna and flora are destroyed and/or dislocated creating open niche space for colonization. This colonization occurs as a succession of species, beginning with pioneer species that typically have high biomass, high growth rates, and short generation periods (Rhoads et al., 1978; Walker and Alberstadt, 1975). Pioneer species are eventually replaced with intermediate, and then climax species. Species of later successional stages live longer and grow slower than pioneer species. As a result of these dynamic processes, disturbance is thought to play a critical role in maintaining high productivity for certain elements of the ecosystem. Generally, catastrophic disturbance dramatically reduces biodiversity. However, disturbance of intermediate frequency and intensity can maintain species diversity by continually opening new space for colonization by pioneer species, while other areas progress through successional stages.

Benthic estuarine environments of South Texas bays appear to be in a state of perpetual early succession. Benthic estuarine communities in this area are characterized by low diversity and opportunistic species (Montagna and Kalke, 1992; Martin and Montagna, 1995). There are several possible reasons for this. First, estuarine succession models developed for application in other areas may not apply to South Texas estuaries due to physical (e.g., depth and tides) and climatological (e.g., rainfall and wind speed) differences. Second, benthic communities of South Texas may be in a state of constant disturbance. Examples of disturbances that could play a role in community development in this region include: trawling, sediment resuspension, wide salinity ranges (including hypersalinity), low dissolved oxygen, and pollution.

C.1. Defining Succession

The concept of community succession was initially developed to describe changes in terrestrial vegetation. J. E. B. Warming and H. C. Cowles (1899; 1901) are credited with inception of this concept (Krebs, 1972), which they used to describe changes in sand dune vegetation as the community developed. Succession theory was formalized by F. E. Clements (1916; 1936) who referred to it as the “monoclimax hypothesis” (Krebs, 1972). In Clements’ theory, succession was viewed as a linear process that begins with a pioneer stage that facilitates colonization of subsequent species. Species replacement continues until the climax stage is reached.

The term "succession" is used to describe community dynamics. A community is "... a group of organisms occurring in a particular environment, presumably interacting with each other and with the environment, and separable by means of ecological survey from other groups (Mills, 1969; p. 1427)." Succession has been used as a means of characterizing, and sometimes predicting, community changes through time (Odum, 1969; Horn, 1974; Pearson and Rosenberg, 1978; Anderson, 1986). Odum (1969) defined succession by three parameters:

> "(i) It is an orderly process of community development that is reasonably directional and, therefore, predictable. (ii) It results from modification of the physical environment by the community; that is, succession is community-controlled even though the physical environment determines the pattern, the rate of change, and often sets limits as to how far development can go. (iii) It culminates in a stabilized ecosystem in which the maximum biomass (or high information content) and symbiotic function between organisms are maintained per unit of available energy flow (Odum, 1969, p. 262)."

General features described in Odum's (1969) model are fairly uniform throughout the literature. It is generally agreed that succession is a directional process in which species alter the environment to culminate in a terminal climax, or equilibrium, community. However, the role disturbance plays in the general successional process is not a consistent feature in the application of the theory.

C.2. Defining Disturbance

Though many studies use a concept of disturbance, it is difficult to find a universal definition describing characteristics of disturbance. Disturbance is often used synonymously with stress and perturbation when applied to ecosystem dynamics (Rykiel, 1985). However, these words may confuse the issues of cause and effect. In Rykiel (1985), disturbance is defined as "A cause; a physical force, agent, or process, either abiotic or biotic, causing a perturbation [an effect of disturbance] (which includes stress) in an ecological component or system; relative to a specified reference state and system; defined by specific characteristics (p. 364)."

Disturbance is often defined as a physical process, e.g., as dredging, storms or construction (Wilson, 1987; Horn, 1974; Rhoads et al., 1978). Focus on the physical nature of disturbance probably arises from early applications of succession theory to the terrestrial environment, where key disturbances were fire and cattle grazing. As succession theory has become more frequently applied to marine and estuarine environments, the definition of disturbance has been broadened to include chemical alterations of the environment, e.g., nutrient enrichment and industrial pollution (Simboura et al., 1995; Warwick and Clarke, 1994).

Chemical disturbance may be more important in aquatic environments than in terrestrial environments. Estuarine communities are subject to two critical chemical stresses that do not influence terrestrial communities: fluctuations in salinity and oxygen. Both of these stresses can cause mortality and declines in community diversity. Other chemical disturbances may arise from spills of oil or chemical products and the discharge of municipal and industrial effluent.

Zajac and Whitlatch (1982a) define disturbance as "...any stochastic event initiating species population change either from density-independent mortality and/or change in the resource base of the community... (p. 1)." This definition is sufficiently broad to incorporate both physical and chemical disturbances. It also may include alterations in food availability, which is a "disturbance" that is not ordinarily considered in succession models.

C.3. Models of Succession

At least five different disturbance and succession models have been proposed to describe and predict community development through time. These models describe: 1) the general successional sequence, 2) the influence of a disturbance on a community, 3) the influence of disturbance on the progression of succession, 4) the physiological influence of a disturbance, and 5) the various means by which early succession species influence subsequent colonization.

The conceptual model presented in Figure I.1 is adapted from Odum (1969). In this model, early succession species are the first to colonize a substratum. These species modify the physical environment making it more suitable for species later in the successional sequence. Succession continues until a stabilized ecosystem is achieved. Many of the characteristics Odum (1969) associated with either end of the successional sequence are presented in Table I.1.

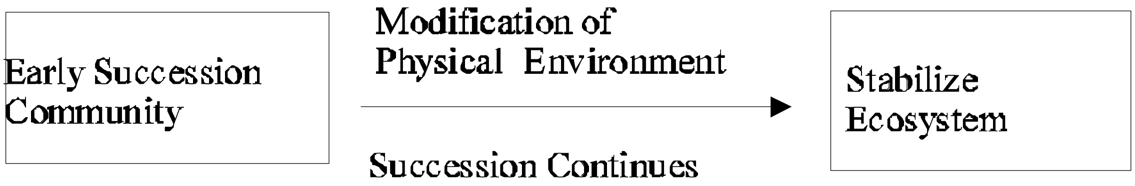

Figure I.1. Conceptual model of succession. Adapted from Odum (1969).

Table I.1. General characteristics of early and late succession**.**

Characteristic	Early Succession	Late Succession
Food chain complexity	linear grazing based[1] moderate/low[2]	complex detrital based[1] moderate/high[2]
Total organic matter	small[1]	large[1]
Species diversity	low[1, 2]	high[1, 2, 4]
Pattern diversity/stratification	poorly organized[1] low[2]	well organized[1] high[2]
Niche specialization	broad[1] low[2]	narrow[1] high[2]
Organism size	small[1, 3]	large[1, 3]
Life cycle	short[3] simple[1]	long[3] complex[1]
Role of detritus in nutrient regeneration	unimportant[1] unclear[3]	important[1] more important[3]
Growth form	rapid (*r*-selection)[1]	feedback controlled (*k*-selection)[1]
Growth rate	high[2, 3]	low[2, 3]
Nutrient conservation	poor[1, 3]	good[1, 3]
Stability (resists external perturbations)	poor[1]	good[1]

[1]Odum (1969), [2]Walker and Alberstadt (1975), [3]Rhoads et al. (1978), [4]Dauer (1993)

The disturbance model presented in Figure I.2 is adapted from Dauer (1993). It is based on the assumption that "...healthy benthic communities can be characterized by high biomass estimates dominated by long-lived, often deep-dwelling, species and high species richness (Dauer 1993, p. 252)." As depicted in the model, a healthy community is exposed to stress, such as low dissolved oxygen, contaminated sediments (in this case stress is a synonym of disturbance), which alters the community. Characteristics of the altered community, compared to the healthy counterpart include lowered community biomass, decreased species richness, a lower percentage of biomass representing deep-dwelling and equilibrium species, and a greater percentage of biomass representing fast growing opportunistic species.

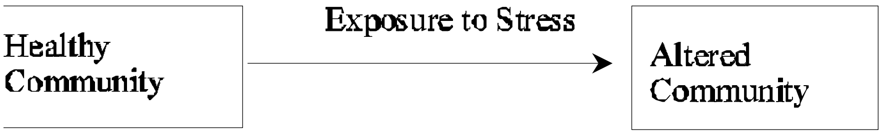

Figure I.2. Conceptual model of disturbance. Adapted from Dauer (1993).

Rhoads et al. (1978) described the role of disturbance in the successional sequence. A conceptual diagram adapted from Rhoads et al. (1978) is presented in Figure I.3. In this model, the occurrence of a disturbance prevents a community from reaching the equilibrium stage. However, in the absence of disturbance, community succession will progress until a stable equilibrium is reached.

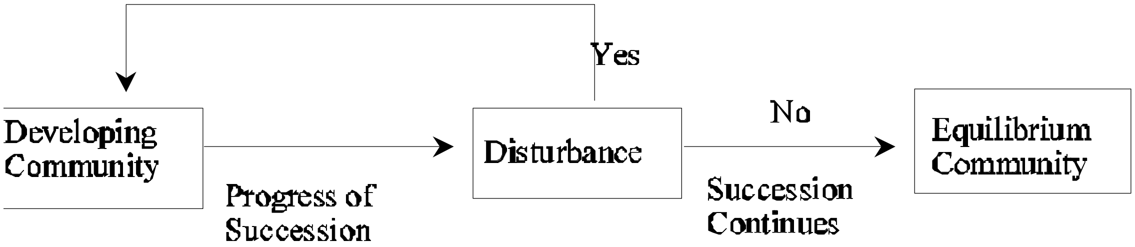

Figure I.3. Conceptual model of the role disturbance plays in the successional process. Adapted from Rhoads et al. (1978).

The fourth model (Figure I.4) illustrates how a disturbance may induce a physiological response in an organism. This diagram is adapted from Forbes et al. (1994) who investigated the influence of the disturbances nutrient enrichment and low oxygen on capitellid polychaetes. Organic enrichment and low oxygen caused physiological responses in the polychaetes that resulted in decreased feeding and decreased conversion efficiency. The physiological-based model illustrates how a community response to a disturbance may represent the sum total of the physiological responses of individual species.

Figure I.4. Conceptual model of the physiological response of an organism to disturbance. Adapted from Forbes et al. (1994).

Connell and Slatyer (1977) developed a successional model in which early succession colonists influence community development through three different mechanisms. They referred to these three mechanisms as the: 1) facilitation, 2) tolerance, and 3) inhibition models. Conceptual diagrams of each model are presented in Figure I.5.

In the facilitation model (Figure I.5A; Connell and Slatyer, 1977), early succession species modify the environment so that it is less suitable for species earlier in the sequence, but more suitable for later species. Thus, succession continues until environmental modification no longer facilitates community development and a stable community is attained. This model is analogous to Odum's (1969) model presented in Figure I.1.

In the tolerance model (Figure I.5B; Connell and Slatyer, 1977), early succession species modify the environment so that it is less suitable for species earlier in the successional sequence. However, it differs from the facilitation model in that the environmental modifications have no affect on the recruitment of, and colonization by, species later in the sequence. Thus, succession continues until no species can invade and grow in the presence of resident species.

The inhibition model (Figure I.5C; Connell and Slatyer, 1977) is different than the facilitation and tolerance models. In the inhibition model, early succession species modify the environment, making it less suitable for any species to colonize the area. Thus, early colonists exclude new recruits as long as they persist.

C.4. Application of Models

Applying disturbance and succession models to benthic estuarine communities is not easy. Problems arise due to specificity of some models to certain environmental conditions or habitats, and over-generalization of ecological mechanisms modeled. With these cautions, it is possible to interpret and predict changes in benthic habitats.

Though generalized, succession models were developed with respect to specific conditions, disturbances, and habitats making the broad application of a universal theory difficult. For example, Dauer's (1993) disturbance model was developed based on the effects of low oxygen and contaminated sediments as disturbances. Another potential problem is the possibility of geographic bias in the succession and disturbance literature (Pearson and Rosenberg 1978). This is due to the predominance of data from North America and Western Europe. Furthermore, succession models were initially developed for, and applied to, terrestrial ecosystems.

It is difficult to apply terrestrial succession models to estuarine ecosystems. In an estuary, communities are subjected to two primary stressors that do not exist in the terrestrial environment: fluctuations of ambient salinity and dissolved oxygen. These fluctuations can be large in Texas bays and estuaries that are shallow and subjected to pulses of freshwater inflow, high evaporation, and high temperatures. Thus, numerous natural disturbances (e.g., high salinity, low oxygen) may intermittently and repeatedly affect benthic organisms making it difficult to differentiate between anthropogenic and natural disturbances.

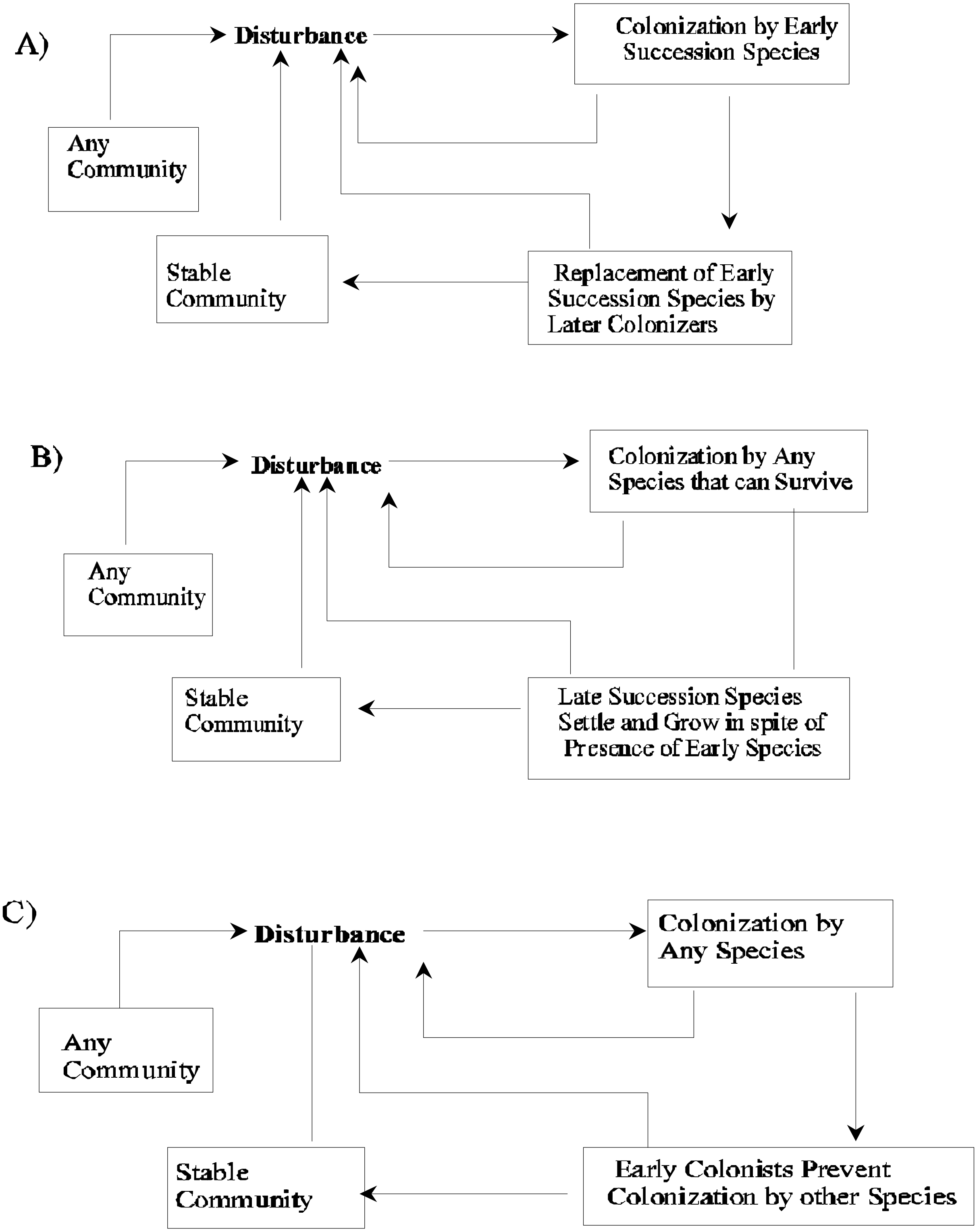

Figure I.5. Conceptual models illustrating the influence of early colonist species on the progression of succession. A) the facilitation model, B) the tolerance model, and C) the inhibition model. Adapted from Connell and Slatyer (1978).

Few organisms can tolerate broad salinity fluctuations. The number of species in an aquatic habitat decreases with salinity below 30‰ until about 6‰ (Britton and Morton, 1989). At salinities less than 6‰, freshwater species begin to tolerate saline conditions and diversity sharply increases (Britton and Morton, 1989). Thus, broad salinity fluctuations have the effect of a disturbance in estuaries, especially when pulses of freshwater cause sudden salinity declines. The effect is to cause a successional sequence.

Decease in diversity with decreasing salinity is due to the inability of euryhaline species to cope with the osmotic influx of water into their bodies. Euryhaline species are able to tolerate broad salinity changes by one of two mechanisms. Some allow their bodies to equilibrate with the surrounding environment. Others maintain tolerable ionic concentrations within their bodies through osmoregulation (Britton and Morton, 1989).

Few organisms are able to survive hypoxic (< 2 mg O_2 l^{-1}) or anoxic (<1 mg O_2 l^{-1}) conditions that occur in some estuarine environments. Estuarine water is susceptible to sudden increases in nutrient concentrations due to influx of detritus, nutrients and animal wastes transported by terrestrial runoff. Some estuaries also receive point source inputs from sewage outfalls and industrial effluent discharges. These nutrient inputs may lead to increased respiration, which consumes available oxygen, creating hypoxic, and sometimes anoxic, conditions.

Though succession and disturbance models are derived with respect to specific environmental conditions and habitats, mechanisms described by models are sometimes overgeneralized. For example, influence of species' life histories and physiological responses to disturbance on community succession are not incorporated directly into models. In addition, influence of interspecific interactions, e.g., competition, predation, and food availability for colonizing species are omitted from models. These elements are directly related to the progression of succession and are important factors influencing community dynamics and ultimate outcome of succession.

Frequency and intensity of disturbances may also influence the outcome of succession. Frequency of disturbance may maintain ecosystem productivity by inhibiting a community from reaching equilibrium (Rhoads et al., 1978). In addition, disturbance frequency at various stages may interrupt the progress of succession and reset progression back to the beginning of the sequence (Connell and Slatyer, 1977). It is obvious that magnitude of a disturbance may influence succession, but we are not aware of a succession model in which disturbance intensity has been incorporated.

C.5. Community Succession

Community succession is often described as exhibiting distinct patterns (Cooper, 1939; Odum, 1969; Horn, 1974; Rhoads et al., 1978; Dauer, 1993). Succession is generally initiated by a disturbance that creates an empty patch for colonization. This open patch is then colonized by early succession species with the characteristics presented in Table I.1. Species that occur early in the succession sequence may facilitate, inhibit, or not affect subsequent species colonization (Connell and Slatyer, 1978). Community succession then continues through a variety of mechanisms until a stable climax community is attained.

The emphasis of most succession models is directed toward early succession versus climax communities. This is thought to arise from the variety of mechanisms that affect community colonization during the progression of succession between the two extremes. Examples of such mechanisms include: species' life histories, food resource availability, predation, competition, species interactions with the environment, disturbance frequency, and intensity of disruption. These mechanisms may act in concert and to varying degrees influence progression of succession. Thus, characterization of succession stages between early succession and climax communities is difficult and probably geographically and temporally specific.

C.6. Estuarine Benthic Succession

Many studies apply disturbance and succession models to estuarine benthic communities. Some studies focus on colonization of newly deposited, defaunated sediment (Rhoads et al., 1978; Zajac and Whitlatch, 1982a; 1982b Gallagher et al,. 1983). Others investigate disturbance based succession arising from organic enrichment (Pearson and Rosenberg, 1978; Levin and Smith, 1980; Dauer, 1993; Forbes et al., 1994), hypoxia (Santos and Simon, 1980; Dauer, 1993; Forbes et al., 1994), industrial pollution (Moran and Grant, 1989a; 1989b; Simboura et al., 1995), and bioturbation (Brenchley, 1981). Other benthic succession studies concerned deep sea communities (Levin and Smith, 1984; Grassle and Morse-Porteous, 1987), fouling communities (Sutherland and Karlson, 1977; Moran and Grant, 1989a; 1989b), rocky intertidal communities (Lubchenco and Menge, 1978; Soussa and Connell, 1992), and meiofauna (Sherman and Coull, 1980; Fegley, 1988; Alongi and Christoffersen, 1992).

Most estuarine benthic succession studies focus on early stages of succession. This is probably due to the relatively short duration of a scientific study compared with length of time it may take a community to fully develop. Few studies have a duration longer than two or three years due to funding constraints. Another less obvious barrier to long-term studies is the time limitation of a graduate students degree plan.

It is not known how long it takes a marine benthic community to reach a climax state. There is evidence that estuarine benthic communities may progress through succession faster than those of the deep sea benthos (Levin and Smith, 1984). The best evidence is from studies on succession following dredging, effects of large storms, and recovery from accidental spills. These studies indicate from one to five years may be necessary for full recovery.

Many succession models focus on defining characteristics of an early succession community. The key characteristic of an early succession species is community dominance by opportunistic species, e.g., *Mediomastus ambiseta*, *Streblospio benedictii*, and *Mulinia lateralis* (Dauer, 1993). Other characteristics include lower biomass and diversity compared to a climax community (Dauer, 1993). Composition of an early succession community will largely depend on species' life histories, time of year, prevailing weather patterns, and larval recruitment dynamics. Presence of early succession communities can be an indicator of recent disturbance.

Opportunistic species, characteristic of early succession benthic communities, can alter the environment to affect further colonization. For example, spionid polychaete tubes have been found to facilitate colonization of azoic sediment (Gallagher et al., 1983). Infaunal polychaetes can play an important role in controlling community structure (Commito, 1982). This can occur directly by preying on newly settling larvae or indirectly when bioturbation or bioirrigation alters sediment.

Large infauna (e.g., Enteropneusta), often associated with climax communities, may facilitate oxygenation of deeper sediments by bioturbation (Flint and Kalke, 1986). With oxygenation of deeper sediments, infauna are able to live more deeply in sediments, enhancing colonization by new infaunal species (Flint and Kalke, 1986). Thus, colonization of a community by larger infauna appears to facilitate an increase in community diversity and promotes progression of succession.

It has been suggested that we have a poor understanding of characteristics of a benthic climax community (Rhoads et al., 1978). However, there is evidence indicating a benthic climax community consists of larger, deep-dwelling animals (Flint and Kalke, 1986), high biomass, high diversity, and longer-lived species (Rhoads et al., 1978; Dauer, 1993;).

It is uncertain whether benthic climax communities have been observed in the field. Many study areas lack climax species altogether (Sutherland and Karlson, 1977; Santos and Simon, 1980; Gallagher et al., 1983; Alongi and Christoffersen, 1992). Concurrent with these findings, estuarine benthic communities of South Texas tend to be dominated by opportunistic species (Montagna, unpublished data). Opportunistic species are characteristic of early succession and highly disturbed communities. Areas supporting only these kinds of communities may be in a state of continual disturbance.

Absence of climax species has been associated with tidal and storm disturbances (Alongi and Christoffersen, 1992), and experimental effects such as tray avoidance and hydrodynamic interactions with equipment (Levin and Smith, 1984). Other disturbances that may prevent a climax community from being attained include hypoxia, salinity fluctuations, and resuspension of sediments. Each type of disturbance may individually act to inhibit the progress of succession. However, it is more likely they interact synergistically in space or time to maintain the community at its early development stage.

D. Objectives

Two objectives of the present study are to: 1) determine effects of anthropogenic disturbances on biological processes, and 2) assess relative contribution of anthropogenic and natural disturbances on benthic habitats. The approach to fulfill these objectives was to perform a literature survey, and synthesize available data in the context of disturbance and succession theory. First, anthropogenic and natural influences are identified. Then, status and trends of disturbances, as well as potential environmental impacts, are characterized. Together, influences and trends represent study results. The discussion section compares disturbance effects on different bottom types and identifies key resource management concerns. Finally, data information gaps were identified.

Table of Contents

II. METHODS

A. Historical Data

Effects of anthropogenic and natural disturbance on benthic flora and fauna were evaluated based primarily on available literature and databases. The evaluation includes special emphasis on assessment of anthropogenic disturbances resulting from: 1) shrimp harvesting, 2) commercial tug and barge operations, 3) recreational boating, 4) maintenance channel dredging, and 5) open water placement of dredged material and effects of natural processes including: 1) storms, 2) fronts, 3) floods, 4) freshwater inflow, 5) suspended sediments, 6) subsidence, 7) hypoxia, 8) brown tide, and 9) bioturbation on the physical, chemical, and biological processes of bay bottom habitats. Qualities of disturbances (depth, duration, and frequency) were considered when comparing influences of each type of disturbance.

A.1. Literature review

Available literature relating to anthropogenic and natural disturbances of estuarine benthic communities within the CCBNEP study area was reviewed. We identified and incorporated pertinent findings from CCBNEP projects that were completed or in progress. Because some reports were not published, draft copies were reviewed. Primary scientific literature on disturbance and succession ecology, as well as other relevant topics, was reviewed. Literature sources included those identified in current CCBNEP characterization reports. We compiled detailed information on status and trends of anthropogenic disturbances including: shrimp harvesting, commercial tug operations, recreational boating, and dredging practices. Several literature resources were identified *a priori* for issues related to: freshwater inflow (Texas Department of Water Resources, 1982; Montagna and Kalke, 1992; Longley, 1994), commercial fish and shrimp harvesting (Matlock, 1982; National Marine Fisheries Service, 1990; Nichols et al., 1990; Hoar et al., 1992; Boyd et al., 1994; Food and Agriculture Organization, 1994; Robinson et al., 1994; Campbell and Choucair, 1995; Fuls, 1995), sportfish harvesting (Warren et al., 1994), and dredging (Windom, 1972; 1975; James, et al., 1977; Rhoads et al., 1978). Recreational boating trends, based on angler surveys, were obtained from the Texas Parks and Wildlife Department (TPWD). Information on commercial tug operations was obtained from the U.S. Department of Commerce and the Corps of Engineers (COE). The COE also provided information regarding channel dredging and the deposition of dredged material. Information regarding the construction of structures (e.g., piers, docks, cabins, oil and gas facilities) over state-owned submerged land was available from the Texas General Land Office (GLO).

A.2. Assessment of disturbances

Natural and anthropogenic disturbances to submerged aquatic habitats were identified based on historic and current levels, as well as projected trends. Key sources for identification and assessment of disturbances were characterization reports completed in Year 1 of the CCBNEP. A second source was members of the CCBNEP Scientific and Technical Advisory Committee. Status and trends of natural disturbances (e.g., storms, floods, bioturbation, salinity effects, ambient turbidity, siltation rates, subsidence) and impacts on estuarine benthos within the CCBNEP study area were identified.

Particular attention was given to natural disturbances that may result in impacts similar to those of human activities. Commonalities or differences that may exist between effects of natural and anthropogenic disturbances were identified.

Influence of anthropogenic and natural disturbances on benthic plant and animal communities was assessed in the context of ecological disturbance and succession theory. Several models have been developed to describe benthic disturbances and subsequent succession of benthic organisms (Walker and Alberstadt 1975; Pearson and Rosenberg 1978; Rhoads et al., 1978). These models served as the basis for assessment of disturbance effects within the CCBNEP study area. A conceptual model of the CCBNEP ecosystem and food web (Montagna et al., 1995) served as the basis for food web analysis. This model contains a diagrammatic representation of the study area's estuarine food web and habitats. In addition, specific anthropogenic and natural disturbances that may affect trophic dynamics and energy transfer were illustrated. Alterations in the food web were assessed using disturbance and succession theory. Magnitude, frequency, and areal extent of anthropogenic disturbances on each bottom type (e.g., vegetated, unvegetated, mud, sand, and shell) were compared.

Relative level of disturbances was difficult to compare based on literature reviews. In some cases direct comparisons between anthropogenic and natural disturbances were possible, because the same habitat or biota was affected, and endpoints were in the same units. Examples of these disturbances include hypoxia, freshwater inflow, and shipping effects on macrofaunal abundance and biomass. However, comparing these disturbances to other types of disturbances was more problematic because ecological processes affected were different or endpoints were in different units. For example, it was desirable to compare effects of dredging and hypoxia on macrofauna, but it was not possible, because time scales of dredging events and hypoxic events were discordant. Also, the primary ecological process affected by light is photosynthesis (because increased turbidity reduces light), but the primary ecological process affected by hypoxia is respiration (because of reduced availability of dissolved oxygen). Clearly, these are not comparable. Another problem was dredging and hypoxia do not occur in the same areas or habitats. The only way effects of different disturbances could be compared to one another was in a large, complex ecosystem model with standardized units. By linking all processes in one model, changes in levels of a disturbance in one compartment is easily comparable to changes in another compartment. However, construction of such a model was well beyond the scope of the current study. Therefore, our approach was to characterize extent and/or trends in disturbances, contrast those that were directly comparable to each other, and determine which benthic habitats were affected by each type of disturbance.

A.3. Public databases and gray literature

Appropriate local, state, and federal agencies (e.g., Port of Corpus Christi, GLO, TPWD, Texas Natural Resource Conservation Commission (TNRCC), Texas Water Development Board (TWDB), National Biological Service (NBS), U.S. Geological Service (USGS), and COE) were contacted to identify pertinent databases and publications that were not readily available.

B. New Data

Members of the CCBNEP Science and Technical Advisory Committee (STAC) were consulted to compile a list of resource management concerns. Specific members, whose expertise complemented that of the authors, were targeted for interviews. Interviews were conducted using a set of standard questions (Appendix A). Questions were framed to illicit responses on concerns about benthic habitats.

Information collected during interviews was also used to identify information gaps that were pertinent to long-term management and monitoring of the CCBNEP study area. These gaps were analyzed to determine how they may influence findings or conclusions related to effects of anthropogenic and natural disturbances on the physical, chemical, biological, and ecological components of estuarine benthos.

Table of Contents

III. RESULTS

A. Status and Trends of Anthropogenic Disturbances

This section provides results from literature reviews on status and trends of man's activities that can affect bay bottom habitats. Activities reviewed include: shrimping, commercial maritime transportation, dredging, and recreational boating.

A.1. Shrimping

A.1.1 Description of shrimping activities

The commercial shrimping industry in Texas has evolved from a man-powered fishery which used cast nets, haul seines, sail-powered vessels, and other primitive methods to become the most valuable commercial food fishery in Texas. Landings in 1993 totaled 74 million pounds and the total economic impact to the Texas economy was estimated to be at least $500 million per year (TPWD, 1995b). Fishermen began using trawls towed behind motorized vessels for catching shrimp around 1920 (TGFOC, 1923). Two primary factors that determine the extent to which shrimp trawling will affect benthic habitats are time spent trawling (i.e., effort) and area covered per unit time (i.e., net size multiplied by towing speed). Data on effort are compiled for the entire Texas coast but data are not available specifically for the CCBNEP study area. Neither historical nor current data are available on area covered by trawling. Commercial harvest data might serve as a measure of effort and area covered but there are numerous limitations to using harvest data for this purpose. Ponwith and Dokken (1996) described limitations of using harvest data to measure status of the target resource and most of those limitations apply to our use of the data as well. Primary among them are: inaccurate/incomplete reporting, changes in technology (i.e., catch per unit effort), and changes in regulations. Other data which might provide information on amount of trawling effort over time include: seasons and closed areas, kinds of boats and gear used, number of licenses issued, and areas fished. These factors are reviewed below in an effort to assess changes or patterns in trawling effort within the CCBNEP study area.

A.1.1.1. Seasons, times, and closed areas

The first comprehensive management plan regulating shrimp harvest was contained in the 1959 Shrimp Conservation Act, which established licenses for each of the user groups. This legislation set size, bag, possession, time and gear limits and established fishing seasons and areas for fishing (e.g., major bays, bait bays, nursery areas, Gulf waters). In 1985, the 69th Texas Legislature delegated to the Texas Parks and Wildlife Commission authority to regulate the shrimp fishery (Cody et al., 1989). This led to development of the Texas Shrimp Fishery Management Plan, which was published in 1989. This plan allows the Texas Parks and Wildlife Commission to regulate catching, possession, purchase, and sale of shrimp.

For the purposes of regulating shrimping activity, estuarine waters are divided into three zones: major bays, which in the CCBNEP area includes Aransas and Corpus Christi Bays; bait bays, which includes most of Copano, Nueces, Upper Laguna Madre, Baffin, and Alazan Bays; and nursery areas, which includes all tributary bays, bayous, inlets, lakes and rivers. Shrimping is permanently excluded in nursery areas. Areas supporting submerged vegetation within major bays or bait bays are not excluded from shrimping except by virtue of shallow water depth, which excludes most shrimp boats. A table showing size restrictions and the commercial bay and bait shrimp regulations for the CCBNEP area for the 1979, 1990 and 1994 seasons is given in Ponwith and Dokken (1996). Regulations for 1996 are almost identical to those for 1994. Regulations have become more restrictive over the years, especially relative to 1979, and might have served to reduce trawling effort.

A.1.1.2. Kinds of boats and gear used

Commercial shrimp boats are divided into three length classes for management and data reporting purposes: < 7.6 m (25') = part-time commercial bay boats, 7.6 - 16.8 m (25' - 52') = commercial bay boats, and > 16.8 m (52') = commercial Gulf vessels (Crowe and Bryan, 1986; 1987). For the purposes of this report, only the first two size classess will be addressed. Of these smaller boats, bait shrimpers use the smallest, usually < 9.1 m (30') in length (Iversen et al., 1993). Bay shrimping boats are usually flat-bottom, shallow-draft boats, which make it easier to work in shallow bay waters. In the CCBNEP area, depths range from 0.6 m (2') in Mission Bay to 4.0 m (13') in Corpus Christi Bay. Mean depth in the Laguna Madre is about 1.2 m (4') (Baird et al., 1996).

Commercial bay boats may use only a single otter trawl or a beam trawl as the main net and a small version of the otter trawl as a “try” net (to sample shrimp in a given area). The shrimp net is basically a large-mesh, wide-mouth funnel with lateral wings on each side. These wings are attached to top and bottom ends of doors, which are flat and weighted. Trawls are towed along bottom and as water pressure forces the doors upright they act as kites that hold the net open and keep it on the bottom. A tickler chain, located on the bottom between the wings, disturbs the shrimp, which jump up off the bottom and are captured as the net passes (Iversen et al., 1993). A beam trawl is a rectangular frame, typically suspended from the bow of the vessel and does not touch the bottom. A try net may not exceed 6.4 m (21') in total width and its doors may not exceed 0.29 m^2 (3 sq. ft.) each. A beam trawl used as a try net may not exceed 3.0 m (10') and, if used as a main net, may not exceed 7.6 m (25'). Two doors used to keep the main net open may range from 0.9 - 3.0 m (3' - 10') lengthwise, and total width of the net and doors may range from 12.2 - 15.8 m (40 - 52'), depending on door size (TPWD, 1995a). During the fall shrimping season only (see below), the maximum width of the net and doors may not exceed 19.0 m (62'). Most commercial bay boats use nets that are near the maximum allowable size (Fuls, 1995). Bait-shrimp boats may also use either an otter trawl or beam trawl as the main net. Net size restrictions in the bait shrimp fishery are the same as for bay shrimping, except that a try-net used by bait shrimp boats may not exceed 3.7 m (12') in total width (TPWD, 1995a).

Tow times for the bait shrimp boats are usually about 30 minutes to insure the retrieval of shrimp in live condition, while bay shrimp boats will pull trawls an average of one to three hours (Page Campbell, personal communication, TPWD, Rockport, Texas). Tow speeds for bay boats are generally 5.5 km hr^{-1} (3 knots), with larger bay boats trawling at around 7.4 km hr^{-1} (4 knots) (Terry Cody, TPWD, Rockport, Texas, personal communication).

A.1.1.3. Number of licensed boats

Four types of shrimp boat licenses are available in Texas: Gulf, bay, bait, and recreational (sport) trawl. For the purposes of this report, we are only concerned with three: bay, bait and recreational trawling. Sale of sport trawl license tags statewide declined from 9,000 in 1959 to 2,000 in 1990. This is probably attributable to the fact that in 1979 daily poundage limit for sport shrimpers in the bays decreased from 100 pounds to 15 pounds (900 to 7 kg) (TPWD, 1995b). Numbers of commercial bay licenses issued state-wide rose steadily from 1965 until 1973 when numbers declined (Figure III.1), probably due to higher fuel prices. Likewise, state-wide numbers of commercial bait licenses also dropped. Numbers increased for all types of shrimp licenses from 1976-1983, then began a steady decline which continued to 1993, partly due to increasing costs of boat operations and fuel (TPWD, 1995b). Statewide, combination licenses increased 41% from 1979 - 1983, while number of boats increased only 12% (Crowe and Bryan, 1986). The increase in combination licenses came mostly in commercial bay boats.

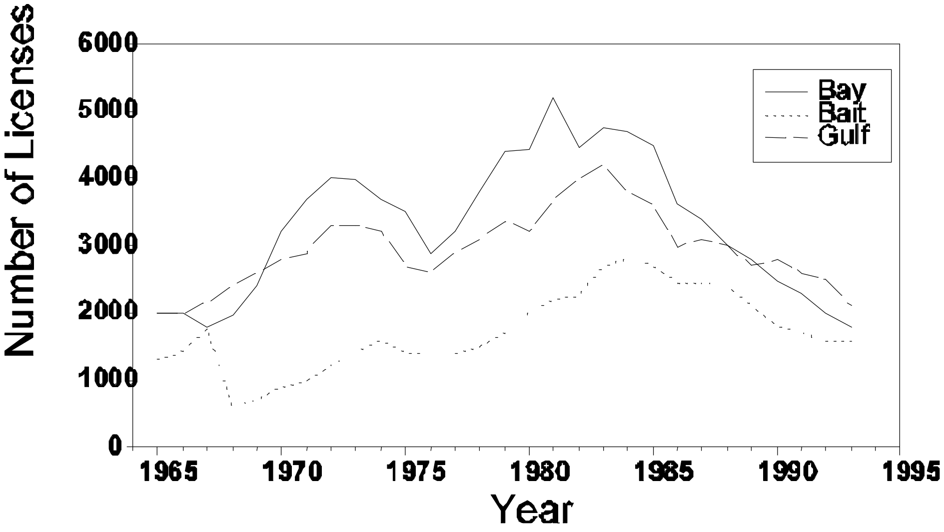

Figure III.1. Number of licenses issued on the Texas coast for bait, bay, and gulf shrimping between 1962-1993. Redrawn from TPWD (1995b).

Within the CCBNEP study area, there are separate data on number of licensed boats for the Fulton/Rockport/Aransas Pass area and from the Corpus Christi area including Upper Laguna Madre (Crowe and Bryan, 1986; 1987; Page Campbell, unpublished data, TPWD, Rockport, Texas). There has been a general decline in number of licensed boats in the CCBNEP area since the peak in 1980 - 1983 (Figure. III.2). In Aransas Bay area, licensed boats increased from around 200 to almost 300 in the mid 1980's, but have since declined. There were 46 fewer licensed boats in 1993 than in 1979. An even greater decline occurred in Corpus Christi Bay area, from 482 in 1983 to 128 in 1993, a 73 % decline. Some licenses shift from one bay system to another. The high number of licenses sold prior to 1981 may be due to anticipation of the 2-year moratorium on new bay and bait boat license sales that began 1 March 1981 (Crowe and Bryan, 1986). Changes in license sales probably reflect increased demand for shrimp, changes in license cost and regulations, and changing economic conditions in Texas (Crowe and Bryan, 1987).

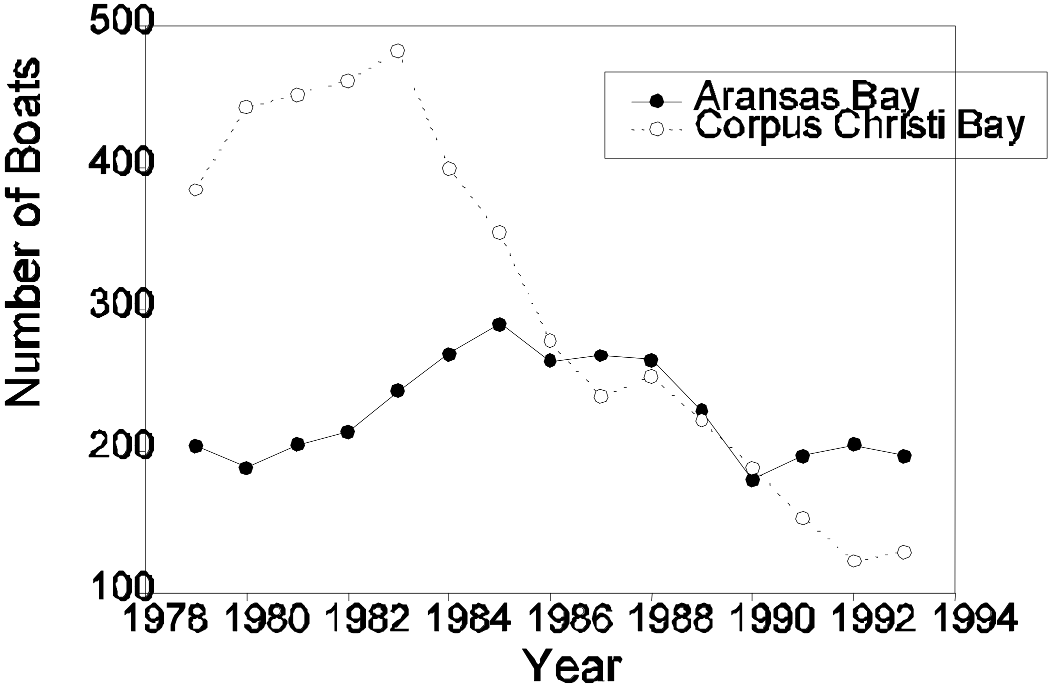

Figure III.2. Trend in number of licensed boats in Aransas and Corpus Christi Bays during the period 1979-1993. Adapted from Crowe and Bryan (1987) and unpublished data (Page Campbell, TPWD, Rockport, Texas, personal communication).

A.1.1.4. Effort

State-wide fishing effort on shrimp stocks increased 95% in the Gulf and 400% in the bay fishery since 1961 (Figure III.3). This growth has led to over-fishing in the bay fishery (TPWD, 1995b). Over-fishing was indicated by overall declines in catch rates (i.e., pounds per trawling hour) of brown shrimp in the bays. Average weight of shrimp landed for each hour of shrimping effort decreased by 40% between 1972 and 1993 (Figure III.4), however, there has been a dramatic increase in the pounds of bay shrimp landed over that period (TPWD, 1995b). The net result for the individual shrimper has probably been a decline in catch for each hour of shrimping effort. "Thus, shrimpers are fishing harder to catch smaller shrimp that are worth less in value, forcing shrimpers to fish even harder" (TPWD, 1995b). If the change in state-wide effort reflects changes in effort in the CCBNEP study area then trawling effort has increased substantially in recent years.

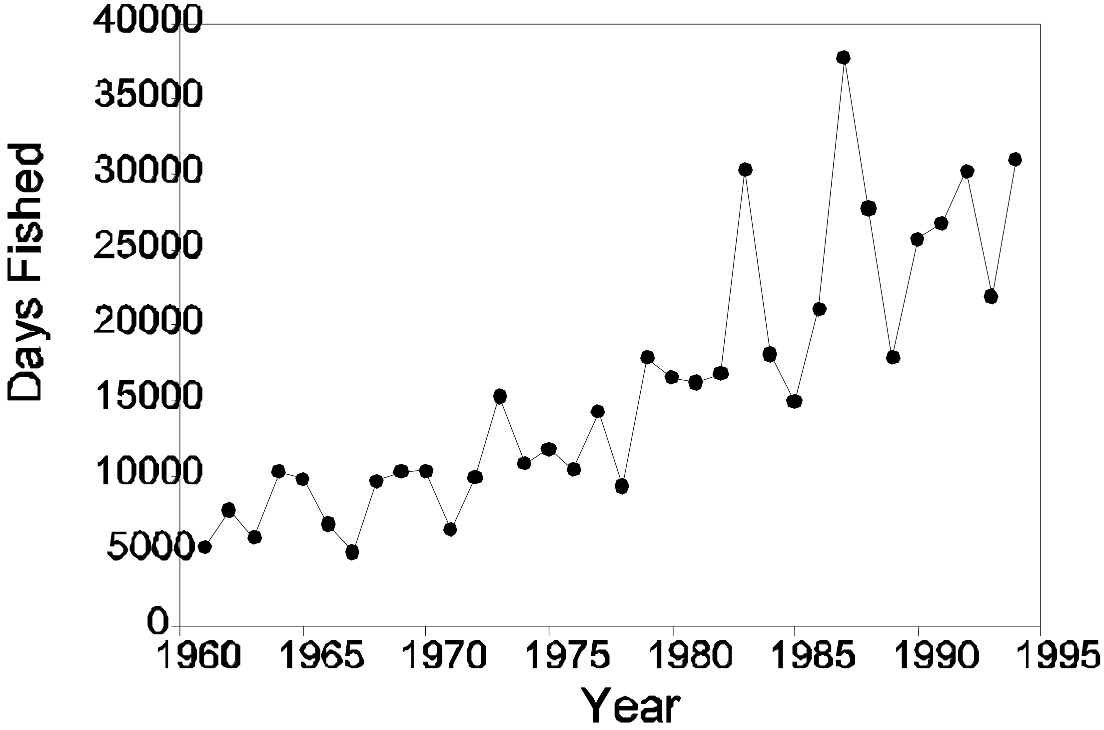

Figure III.3. Number of days fished by shrimp boats along the Texas coast. Redrawn from TPWD(1995b).

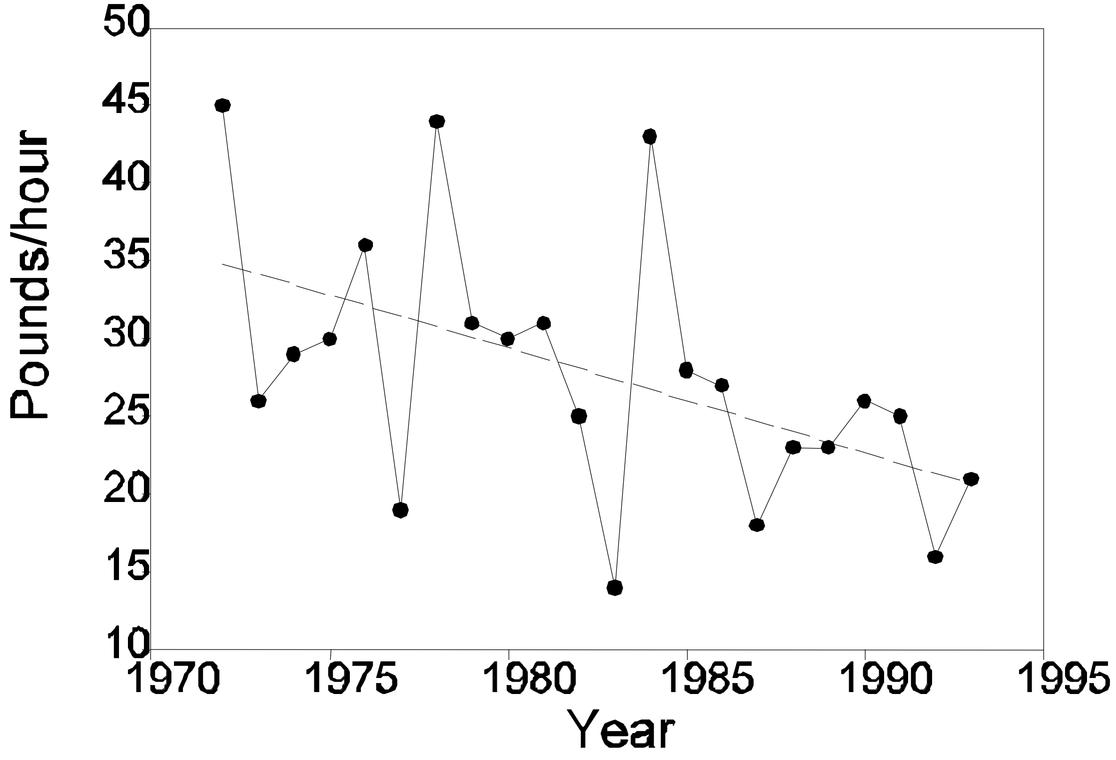

Figure III.4. Coast-wide trend in pounds of brown shrimp caught per hour of trawling. Redrawn from TPWD (1995b). Conversion: 1 pound = 0.454 kg)

On June 8, 1995, the Texas Legislature enacted Senate Bill 750, a limited entry plan for bay and bait shrimp fisheries. This bill established a license program with the intent of limiting numbers of shrimp boats working in Texas bays (TPWD, 1995b). To be eligible for a commercial license for participation in the fishery beginning September 1, 1995, an individual must have held a valid bay and/or bait license on April 1, 1995 (TPWD, 1995b). There are also restrictions concerning vessel upgrades and license transfers. The TPWD is engaged in a license by-back program as an additional means of reducing numbers of boats fishing. In the long-term, limited entry would cap numbers of vessels and institute equitable methods to reduce amount of shrimping effort (TPWD, 1995b). If successful, this plan could result in less trawling in the bays in the future assuming effort by remaining license holders does not increase, or boats from other areas do not shrimp in the controlled areas.

A.1.1.5. Harvest

Combined commercial harvests for both bay and bait shrimp catches for each bay system in the CCBNEP study area from 1979 to 1993 are shown in Figure III.5. A history of commercial fisheries harvests (including shrimp) for the CCBNEP study area was summarized by Ponwith and Dokken (1996). Since 1972, landings of both brown and white shrimp from the Aransas Bay system have been about 50 - 80% higher than landings from the Corpus Christi Bay system and almost an order of magnitude higher than those from Upper Laguna Madre. Quantitative data on shrimping effort (i.e., days trawled per year) are not available specifically for the CCBNEP study area. Therefore, there are no estimates of catch-per-unit-effort (CPUE) and no way to measure the surface area of bay bottom impacted by trawls, either historically or currently.

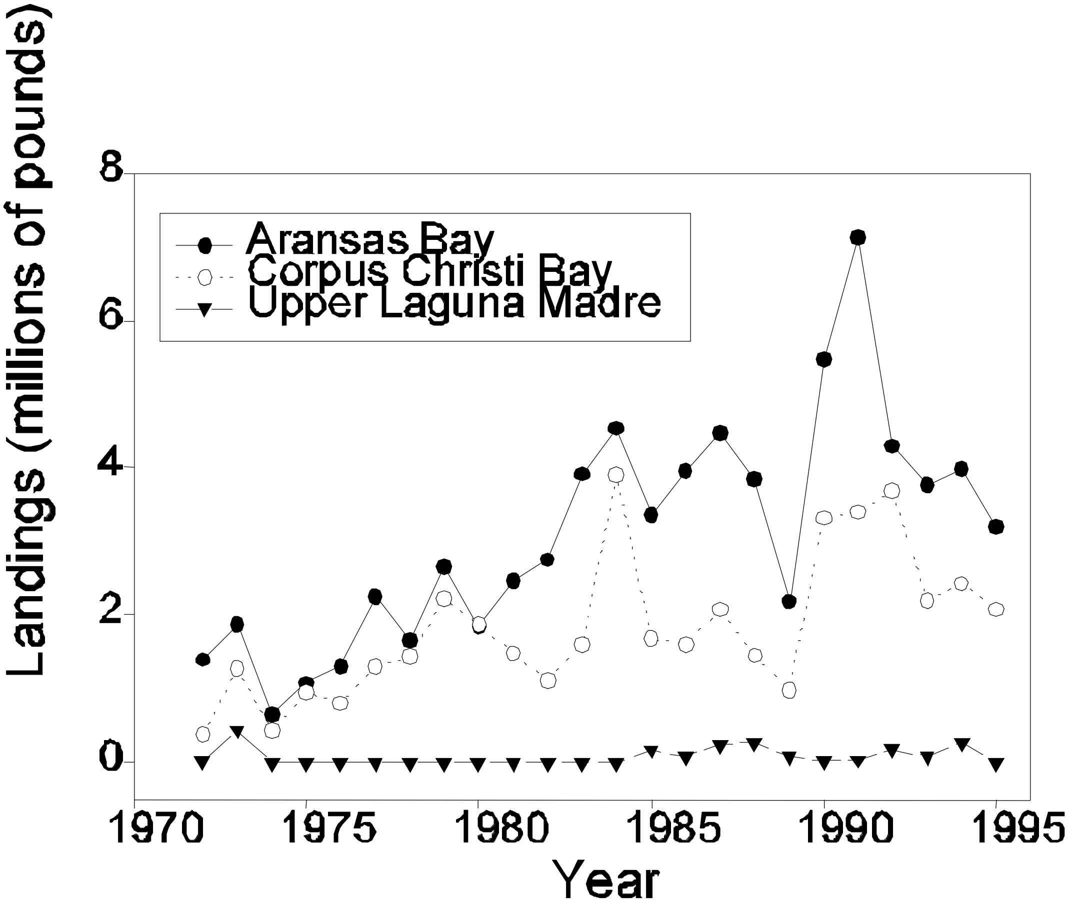

Figure III.5. Total landings of shrimp for each of the major bays in the CCBNEP region. Adapted from Robinson et al. (1995) with additional unpublished data (Page Campbell, TPWD, Rockport, Texas, personal communication)

A.1.1.6. Historical and current shrimping areas

There are no quantitative assessments of either historical or current usage patterns by shrimp harvesters in the CCBNEP study area. There are anecdotal references to concentrated fishing effort in the GIWW and the Corpus Christi Ship Channel in Corpus Christi Bay, but there are no definitive data. It is also known that effort in Aransas Bay shifts seasonally as the shrimp move out of the estuary, but quantitative data do not exist.

A.1.1.7. Bycatch

Virtually every fishery produces an incidental catch or bycatch as part of harvesting. Bycatch is defined as incidental catch of non-target organisms due to fishing activities. In shrimping, many finfish and invertebrates other than shrimp are captured. Bycatch includes crabs, non-commercial shrimp species, jellyfish, starfish, squids, molluscs, and various fishes. Blue crabs and some fishes in the catch are marketable, but the largest portion of catch is usually discarded (Cody et al., 1989). Bycatch in commercial bay shrimp fishery in Aransas and Corpus Christi Bays was analyzed by Fuls (1995) during the 1993 spring and fall open seasons. Bycatch was composed almost entirely of finfish and those results were summarized in Ponwith and Dokken (1996). There were virtually no infaunal organisms in the bycatch and only a few epifaunal organisms were caught. Most of these were various crabs and shrimps and with the exception of blue crab (*Callinectes sapidus*) and lesser blue crab (*Callinectes similis*), none of them ever made up more than 1% of the total bycatch within a season and few were ever taken in numbers greater than 10 per hour. There are no other similar data available for the CCBNEP study area to estimate trends in bycatch of infaunal or epifaunal organisms.

A.1.2. Potential impacts of shrimping activities

Concern that fishing practices, especially those that involve dragging fishing gear over the bottom, are harmful to the environment is centuries old. Graham (1955) reports that in 1376, fishermen petitioned the King of England "*that the great and long iron of the wondyrchoun runs so heavily and hardly over the ground when fishing that it destroys the flowers of the land below water there.*" Given this early insight and numerous studies on effects of several bottom-fishing gear types, it is surprising there is still no consensus regarding impacts of those activities on benthic environments.

The dominant type of gear used in the CCBNEP study area is the otter trawl. We could not find any studies on direct effects of otter trawls on the sediment structure or benthic infauna in the CCBNEP study area. However, Schubel et al. (1978) reported high levels of suspended sediment concentrations behind shrimp trawls and found levels comparable to that caused by dredge disposal activities. Data on trawling effects presented here were drawn from studies conducted in marine environments around the world. The vast majority of studies are from coastal and oceanic sites rather than bays and estuaries. These studies have focused on effects of three similar types of fishing gear: otter trawls, beam trawls, and scallop dredges. Beam trawls referred to here are heavy, rigid structures towed over the bottom rather than the relatively lightweight nets fished at the surface in the Upper Laguna Madre shrimp fishery. Most studies focused on effects of scallop dredges and beam trawls rather than otter trawls. Dayton et al. (1995) conducted an extensive evaluation of the environmental impact of marine fishing, and contrasted effects of various bottom-fishing gear. Effects of three gear types were similar in his study. However, in another study, intensity of damage to the bottom was generally proportional to weight of the gear (Jones, 1992). Otter trawls are generally lighter weight and thus produce the least amount of physical damage. Assessment of trawling effects presented here will cover impacts of all three gears, and differences between otter trawls and the other two gears will be pointed out where appropriate.

Impacts of trawls and dredges on marine environments range from direct and immediate to indirect and long-term. Type and degree of impact depends on a multitude of factors. Jones (1992) reviewed the literature on trawling effects up through 1990. He divided effects into five categories: scraping and ploughing, sediment resuspension, destruction of non-target benthos, dumping of processing waste, and indirect effects. Messieh et al. (1991) also reviewed studies on trawling impacts through about the same period and divided potential impacts into physical and biological. Some possible impacts of trawling listed by Messieh et al. (1991) are:

- Incidental mortality or damage to target and non-target species by direct contact with mobile fishing gear.
- Increased predation pressure in the dredge track resulting from exposure of infaunal species.
- Alteration of sediment chemistry and texture, rendering the seabed less suitable for larval and adult stages of valuable fishery resources. This alteration can also lead to changes in community structure.
- Sediment resuspension can reduce quality of available food for filter feeders, smother spawning areas, detrimentally affect feeding and metabolic rates of benthos, and cause damage to gills of marine organisms.
- Resuspension of sediments containing toxic contaminants (i.e., heavy metals) may increase their bioavailability to marine organisms.
- Increased rates of benthic/pelagic nutrient flux may stimulate phytoplankton production in shallow or well-mixed areas.

A major determinant of degree of impact is nature of substratum being fished. Hard-bottom habitats, such as inshore sponge-coral habitats of the South Atlantic Bight, are subject to substantially greater and more long-term damage than soft-bottom habitats (Van Dolah et al., 1987). Hard-bottom habitats support numerous species of ascidians, sponges, and soft corals that provide excellent habitat for many demersal fish species. Trawling causes damage or loss of these epifaunal communities and habitat they provide. Recovery rates are relatively slow. Even areas of gravel and boulders suffer substantial long-term damage due to loss of sessile organisms. There are virtually no hard-bottom areas impacted by trawling in the CCBNEP study area, so impacts to hard-bottom communities will not be considered further.

A.1.2.1. Sediment resuspension

A substantial sediment cloud is generated by turbulence behind trawl doors and may enhance catch rates by herding fish into the net (Main and Sangster, 1981). Sediment clouds may also contribute significantly to total suspended sediment load of the water column, especially in relatively clear waters. Sediment resuspended by trawling reached up to 10 m above bottom and remained detectable for up to a day over the Mud Patch, a Mid-Atlantic Bight fishing site where sediment composition is > 25% silt and clay (Churchill, 1989). Even in fine sand sediments, sediment clouds may reach 3.0-3.5 m high and 4.5-6.0 m wide 50 m behind trawl doors (Main and Sangster, 1981). Contribution of trawling to increased turbidity varies depending on location and ambient conditions. Churchill (1989) concluded trawling was the primary initiator of sediment resuspension in the Middle Atlantic Bight at depths of > 100 m, but the largest contribution to annual average sediment loads in shallower waters was due to currents during January-April storm events.

Total suspended solids immediately behind a shrimp trawl working in Corpus Christi Bay have been estimated to be more than 5000 mg l^{-1} if disturbed sediments were uniformly distributed throughout the water column (Schubel et al., 1978). However, observed suspended sediment concentrations ranged from 100 to 500 mg l^{-1} in the top 2 m of the water column at a distance of 100 m behind a shrimp trawl. The discrepancy between estimated and observed values is due to a lack of initial uniform distribution of suspended material and to rapid settling of heavier particles. Aerial photographs of bays in the CCBNEP study area regularly show substantial sediment plumes behind working shrimp trawlers (Schubel et al., 1978).

A.1.2.2. Impacts on benthic habitats

Physical damage to soft-bottom habitats has been well documented. Otter doors plow a distinct groove along the sea floor that can vary from a few cm up to 0.3 m deep. In addition, tickler chains between the otter doors may skim off the sediment surface (Jones, 1992; Krost et al., 1990). Depth of impact varies with sediment structure and is deepest in soft mud. A recent study by Schwinghamer et al. (1996), using high resolution acoustics, showed substantial changes in sediment structure to a depth of at least 4.5 cm over the entire area traversed by an otter trawl towed over a hard-packed sandy bottom. Disturbance to greater depths might be expected in silt and clay sediments of the CCBNEP study area. This could be as deep as 30 cm (Jones, 1992; Krost et al., 1990), but Schubel et al. (1978) estimated disturbance depths to be only about 5 cm in Corpus Christi Bay. Using data for shrimping effort from 1975, they estimated that between 25×10^6 and 209×10^6 m^3 of in-place sediment are disturbed by shrimp trawling each year.

Rate of physical recovery of sea beds after disturbance varies in relation to sediment structure and energy level (i.e., tides, waves, and currents) in the environment. Tracks left by trawl doors disappear in as short as a few hours on relatively high-energy sandy bottoms, but may persist as long as five years in low-energy areas with a sandy-mud bottom (Krost et al., 1990). Another critical factor regulating recovery rate of sediment structure is recurrence of disturbance. Even where the relative amount of fishing effort is known, it is difficult to determine exactly how often a particular parcel of bottom is disturbed because trawling effort is not uniformly distributed (Messieh et al., 1991). In heavily fished areas of the open ocean, area disturbed by trawls can be substantial. Caddy (1973) estimated 3-7% of the sea floor in Chaleur Bay, Gulf of St. Lawrence was disturbed annually by otter trawls and 21% of the sea floor of Georges Bank (7700 km^2) was swept annually by scallop dredges. In a more extreme example, Floderus and Pihl (1990) estimated that in a 1300 km^2 area of the Kattegat Sea, in the eastern North Atlantic, mean time of recurrence of otter trawl disturbance was 28 days. Rate of recurring disturbance can have a significant biological effect, because it affects rate of recovery.

A.1.2.3. Direct impacts on benthic and epibenthic organisms

There are extensive studies on direct physical effects of bottom fishing on both epibenthic and infaunal organisms from regions outside the CCBNEP study area, primarily in oceanic rather than estuarine habitats. Other potential effects often ascribed to bottom fishing gear, such as modifications to microbial activity, resuspension and remobilization of contaminants, and increase on the benthic/pelagic nutrient flux have received little experimental attention. Direct impacts of sediment resuspension due to trawling activities have not been investigated but implications can be derived from studies of natural and other anthropogenic sources of sediment resuspension. A general discussion of turbidity follows in section A.5.

The most obvious impacts are seen on large organisms associated with hard-bottom habitats. Impacts on smaller, soft-bodied organisms, especially in soft-bottom habitats, are less clear. There are few studies of effects of trawling on soft-bottom communities. It is often difficult to distinguish fishing effects from natural variation in time and space (Dayton et al., 1995). Furthermore, fishing disturbances have been occurring for many decades, or even centuries, so the original nature of many of these bottom communities is unknown. Long-term data available for portions of the German Bight indicate a shift in abundance from molluscs and crustaceans to polychaetes has taken place since the 1920's (Messieh et al., 1991). However, long-term, pre-fishing observations are rare.

Theoretical considerations suggest deep-water communities may be more at risk than shallow-water communities. Deep-water communities are often characterized by animals with slow growth, extreme longevity, delayed maturation, and low mortality (Dayton et al., 1995), all of which contribute to low recovery rates. Organisms from shallow-water communities generally have life-history strategies adapted to more frequent disturbance (e.g., large ranges in salinity and temperature, and storms). Therefore, shallow-water communities may recover more rapidly. Small-scale or short-period disturbance events may increase diversity of functional groups (Hall et al., 1994), but chronic disturbance has substantially different effects. Recurrent and large-scale trawling activities that remove surface dwelling organisms and modify surface topography will likely reduce heterogeneity of the benthic community and result in development of short-lived deposit-feeding associations, effectively reducing functional diversity (Dayton et al., 1995).

Results of experimental studies of trawling effects on soft-bottom benthos demonstrate the issue is complex and dependent on both the community present and fishing methods and intensity. Sediment structure and environmental conditions may influence impact of a disturbance. Trawling with heavy tickler chains damaged *Subularia* worm tubes and other fragile projecting structures, but there was little damage to benthic species preyed on by fish (Graham, 1955). For example, detrimental effects on benthic communities can be detected on stable sediments, but not mobile sediments (Kaiser and Spencer, 1996). Brylinski et al. (1994) found only minor impacts to an intertidal benthic community due to flounder trawling in the Bay of Fundy. Van Dolah et al. (1991) could find no differences in abundance, diversity, or species composition of benthic infaunal assemblages due to shrimp trawling in a South Carolina estuary. Conversely, Thrush et al. (1995) found experimental scallop dredging produced significant changes in benthic community structure, despite the benthic assemblage being dominated by small, short-lived species prior to dredging. They further argued that their assessment of impacts was “quite conservative” because commercial fishing occurs over a much larger area than that studied and repeated fishing occurs over the same area of seabed on any one trip, producing higher levels of disturbance than experimental fishing. Changes in sediment structure have been detected, even in cases where no significant impact on benthos was demonstrated (Schwinghamer et al., 1996). Trawl-induced changes in sediment structure will reduce structural and dynamic unpredictability, or chaos, which may result in a reduction of material and energy transfer within the system. This may result in long-term changes in ecosystem function even when immediate changes in community structure cannot be demonstrated (Schwinghamer et al., 1996). Furthermore, it is often difficult to extrapolate results of a small-scale, short-term study to large-scale, long-term impacts (Thrush et al., 1995).

The power to detect significant differences between treatment effects in experimental fishing studies was generally not considered in studies where no effect was found (Thrush et al., 1996). Dayton et al. (1996) argue that managers and others evaluating trawling impacts should pay more attention to potential impacts of Type II errors, or failure to find differences among treatments when differences actually exists. Studies that report no treatment effect may have not detected the effect because of small sample size or poor experimental design. The consequence of ignoring Type II errors (e.g., continued fishing at the same level or in the same manner when it is actually having an impact) can include loss of a resource and serious ecosystem effects (Dayton et al., 1996).

A.1.2.4. Ecological impacts of bycatch

There are multiple biological and ecological effects due to shrimp trawl bycatch, but the most obvious effects are seen in finfish which make up the bulk of bycatch. The only significant impacts of bycatch that might affect benthos would be through food webs, either through removal of significant predation pressure on benthic organisms or through altered nutrient cycles. Ecosystem level impacts of shrimp trawling due to killing and discarding a wide range of species may range from influencing competitive interactions, predator-prey dynamics, and other cascading effects throughout food webs. Kennelly (1995) suggests “consequences of such interactions are difficult to comprehend, let alone quantify, and there exists very few examinations of such effects”. Up to 55% of finfish and virtually 100% of crustacean bycatch sinks to the bottom (Rothlisberg, 1992). As much as 33% of the diet of Australian sand crabs (*Portunus pelagicus*) is made up of fishes discarded by shrimp trawlers (Wassenberg and Hill, 1987) and success of the Australian sand crab fishery may be related to this enhanced food supply (Wassenberg and Hill, 1987). Discarded bycatch that reaches the sea floor is largely consumed by bottom scavengers and detritus feeders (Andrew and Pepperell, 1992), enhancing the detrital food web. Ultimately, this matter enters sediment microbial food webs, and would be recycled to surface water. The microbial loop may be a sink and not source of food for benthos (Pomeroy and Wiebe, 1988). Estimates for rates of bycatch recycling via benthic microbes are unavailable.

Sheridan (1984) and Browder (1991) used a theoretical approach to estimate ecosystem level impacts of shrimp trawl bycatch through an energy flow model (Figure III.6). This model contained numerous estimated parameters based largely on information from the eastern Gulf of Mexico, but it serves as an initial attempt to estimate ecosystem effects. Browder (1991) ran a five-year simulation to estimate impact of reducing bycatch by 50% and compared results with simulation runs of current baseline conditions where bycatch discard rate is 100%. If a 50% reduction was accomplished by retaining bycatch and removing it from the system, the model predicted that shrimp biomass would decline and never return to baseline conditions. The decline may have been triggered by an initial decline in high-nitrogen organic material (due to removal of discards from the system) that changed nitrogen cycling rates. If a 50% reduction were accomplished by the use of BRDs (i.e., bycatch was never caught), shrimp biomass declined initially, but rebounded quickly and grew beyond baseline level. Counter-intuitively, biomass of high-nitrogen material increased substantially over baseline, apparently providing more food for shrimp populations. Browder's explanation was that reduction in bycatch mortality left more living animals in the system to deposit waste. She suggested living animals contributed more to nutrient recycling by eating, growing, and depositing waste than by dying and being discarded. This model is currently being updated (NMFS 1995) and will be used to reassess the impact of bycatch reduction on shrimp populations as well as other components of the ecosystem.

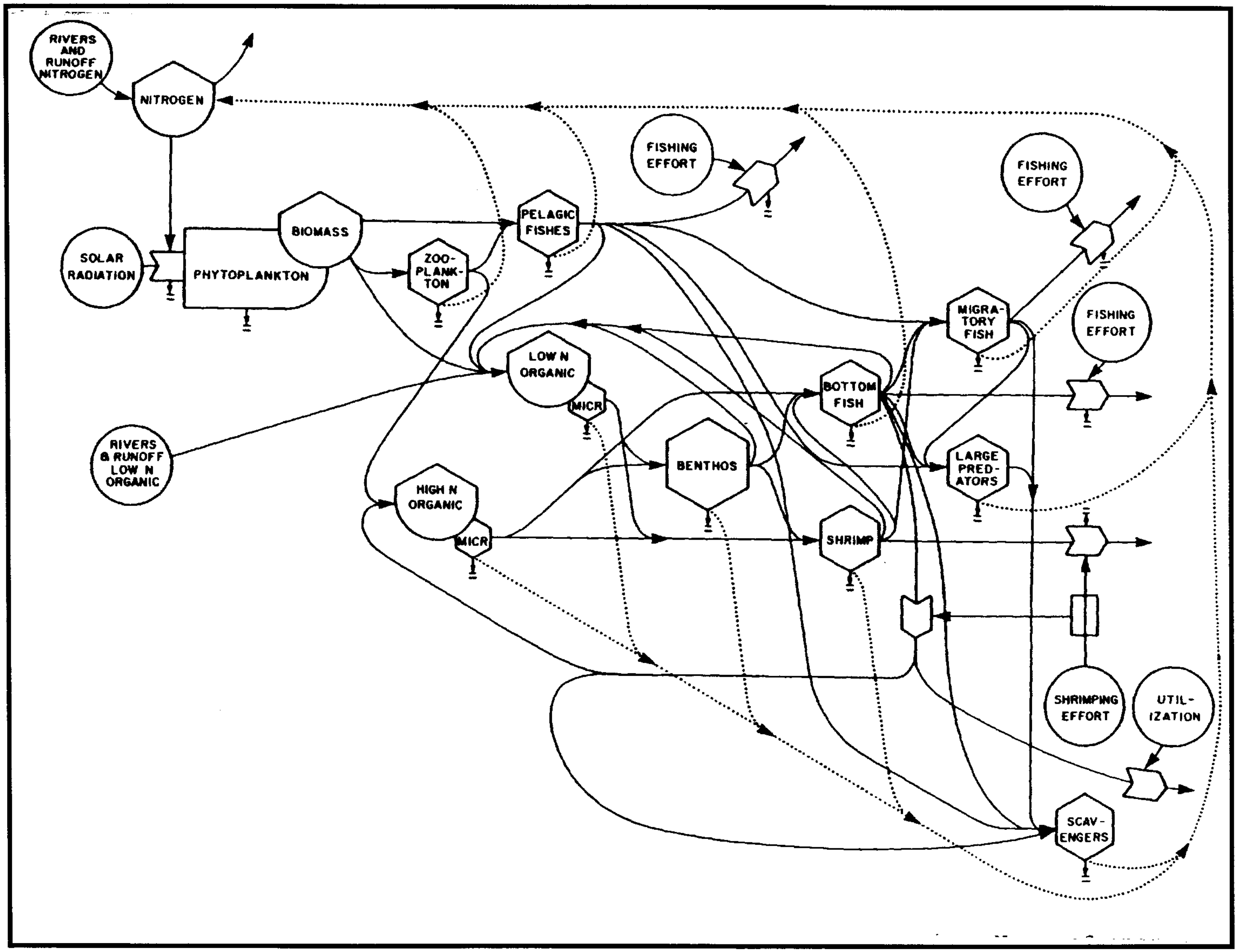

Figure III.6. Energy-flow diagram of the north-central Gulf of Mexico. Compartments connected by energy (solid lines) and nitrogen flows (dotted lines) represent major trophic units of the system. From Browder 1991.

A.2. Commercial ship/boat operations

A.2.1. Operations

The Port of Corpus Christi and associated waterways contribute substantially to economic growth. Numerous industries (e.g., oil and gas, petroleum refining, chemical manufacturing, and agriculture) depend on waterways within the CCBNEP study area for transportation of products and materials. Waterborne transportation is more cost effective than overland transportation. Thus, access to passable waterways helps keep manufacturing and retail costs down, and people in related jobs employed.

The Port of Corpus Christi has experienced steady increases in tonnage of materials transported through the harbor between 1985 and 1994 (Figure III.7). Tonnage transported at Harbor Island stayed relatively constant during this period, ranging between 1200 and 2300 tons (Figure III.7).

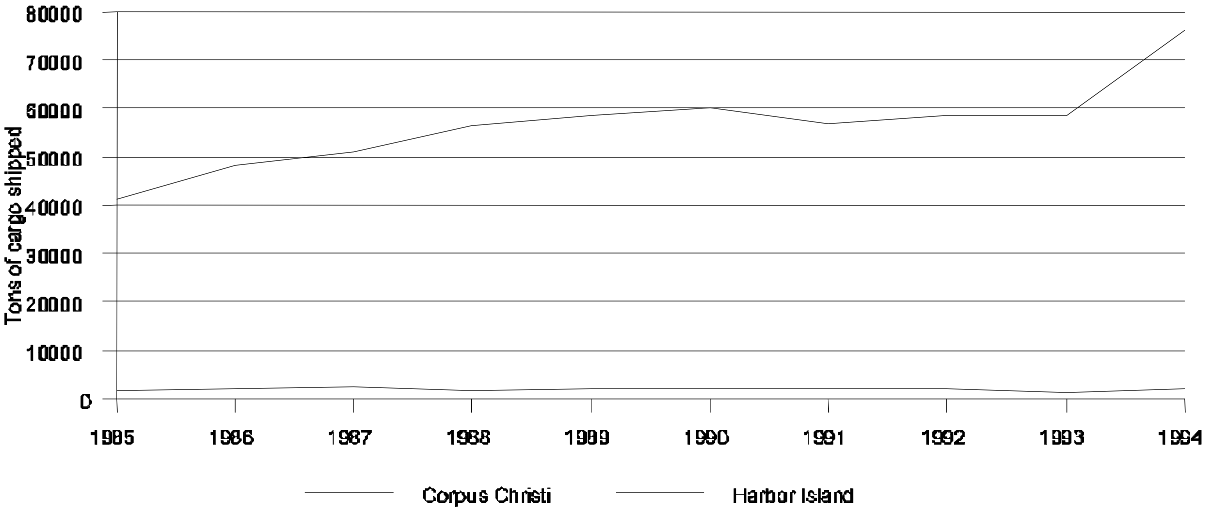

Figure III.7. Tonnage transported each year between 1985 and 1994 in Corpus Christi and Harbor Island ports.

Shipping within the CCBNEP study area is composed predominantly of petroleum and petroleum products (e.g., gasoline and crude oil) (Figure III.8). Other items shipped in this area include crude materials (e.g., rubber, lumber, and ore), food and farm products (e.g., wheat, and vegetable oil), and chemicals and related products (e.g., fertilizers, hydrocarbons, and alcohol). In 1979, more than 49 million tons of commodities were transported via the Corpus Christi Harbor. Petroleum and petroleum products composed 72% of this total. Percentages of other commodities transported in 1979 are given in Figure III.8. By 1994, shipping in the area had risen to 76 million tons. Petroleum and petroleum products had gained 10% of the share of total tonnage, rising to 82% (Figure III.8).

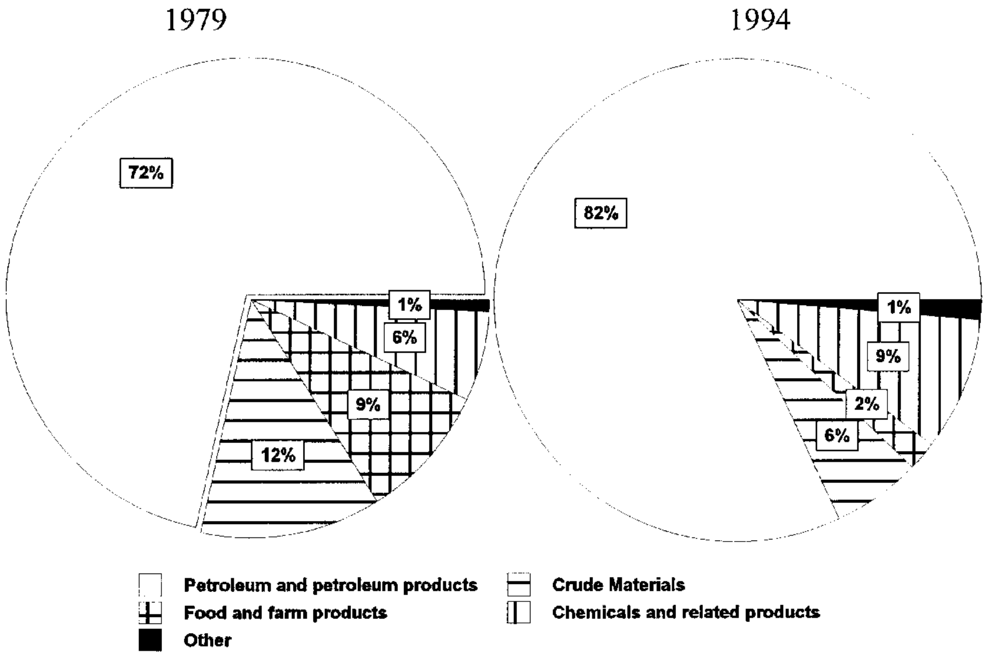

Figure III.8: Percentage of total tonnage transported in 1979 (left pie) and 1994 (right pie) for commodities of regional importance (U.S. Army Corps of Engineers, 1979; 1994).

A.2.2. Potential effects of commercial ship/boat activities

Distinct differences in benthic community structure exist between shoal bottoms and bottoms of shipping channels. For example, channel station communities had lower and more variable population abundances and lower diversity than shoal communities in Corpus Christi Bay (Flint and Younk, 1983). However, channel communities were not composed of distinct and characteristic group of species. These differences were thought to be caused by disturbances associated with continuous ship traffic. Shipping activities can also cause localized effects on benthic habitats due to wakes from ships.

There are several ways shipping activities may physically disturb benthos. For example, erosion of bay or channel margins may be caused by wakes of boats and ships. Wakes may also resuspend bottom sediment increasing turbidity and settlement of sediment on the bottom. In addition, a ship running aground may create disturbance similar to dredging. Shipping activities may also contaminate estuarine environments, especially in enclosed harbors and marinas. Contamination can occur through 1) product spills, 2) improper disposal of human waste, 3) use of antifouling agents on ship and boat hulls, and 4) use of treated lumber in dock and pier structures. These activities can indirectly impact benthic communities through toxic effects.

A.2.3. Effects of marine construction

Treated lumber, created by pressure-treating wood with toxic compounds of copper, chromium, and arsenic (CCA) has been in widespread use since 1933 (Brooks, 1996). Wood is used to build structures in and over water to support the marine transportation industry (e.g., pilings, piers, bulkheads, marinas, and docks). However, it has long been known small amounts of these metals leach out of wooden structures into aquatic environments (Brooks, 1996). These toxic metals accumulate in fine grained sediment and in benthic animals near the structure (Brooks, 1996; Weis and Weis, 1996).

There are lethal and sublethal effects due to CCA lumber (Weis and Weis, 1996). Benthic infauna exhibit reduced diversity and species richness in the vicinity of CCA-treated wood structures. The mud snail (*Ilyanassa obsoleta*) exhibits a sublethal response in which it retracts into its shell and becomes inactive when exposed to CCA leachates. If snails were placed in clean water they recovered, but after a few days in water with CCA wood they died. In addition, laboratory studies have demonstrated copper, chromium and arsenic are trophically transferred metals. Weis and Weis (1996) found "...the extent and severity of effects of pressure treated wood in an estuary depends on the amount and age of the wood and the degree of dilution by water movements." They suggested areas with more CCA-treated wood structures and restricted water flow may permit greater accumulation of contaminants. CCA wood allowed to leach for one month did not deter organism settlement as much as newly placed CCA wood. Wood allowed to leach for two months did not influence community patterns but metals did accumulate in animals (Weis and Weis, 1996).

Areas that have naturally occurring epifaunal assemblages can be severely impacted by marina operations and boating activities (Turner et al., 1997). There was a loss of cover by abundant and spatially dominant solitary ascidians at sites inside New Zealand marinas. This loss led to open space that was increased the cover of sponges, hydroids, erect and encrusting bryozoans, and colonial ascidians. The net effect was a total change in community structure of epifauna in marinas.

A computer model was developed to assess environmental risks associated with bulkhead construction (Brooks, 1996). The model is based on known leaching rates and sediment quality criteria developed by the U.S. Environmental Protection Agency (EPA). The model predicts most projects constructed in well flushed bodies of water will not have severe impacts. However, exceeding regulatory levels will occur in closed bodies of water, where circulation is poor, or where the surface of CCA-treated wood is significant in proportion to the water body (Brooks, 1996). In general, water circulation in the CCBNEP study is poor because of the microtidal (< 1 m) range of tides in the area. This implies care must be taken in engineering marine construction in the CCBNEP area.

A.3. Dredging

A.3.1. Dredging activities

Dredging is mechanical removal of sediment to maintain or create a navigable waterway or harbor. Accumulation of sediment in existing channels is a natural process. The COE conducts regular maintenance dredging to remove accumulated sediment. In particular, the COE is responsible for construction and maintenance dredging of the Corpus Christi Ship Channel and GIWW. The COE is also responsible for permitting private dredge and fill-related activities. Examples of private dredging activities include dredging residential canals and access channels to oil and gas production sites.

In addition to purposeful channel and canal dredging, inadvertent (or accidental) channelization may occur. For example, a barge may run aground in shallow water, “plowing” through sediment. Accidental groundings occur due to high winds, mechanical failures, negligence, or miscalculation of water depth or tidal cycle. Another example is running boat propellers through sediment. Both activities create disturbances similar to dredging, but on different scales.

Oyster shell dredging was a common practice in bays north of the CCBNEP study area prior to 1980 (Ward, 1997). Although there has been commercial oyster shell dredging in the CCBNEP study area in the past, it is not presently an issue. Shell dredging was active in the CCBNEP study area from 1960 to 1969. During these years, between 0.8 - 1 million m^3 (1 - 1.3 million yd^3) of shell material was dredged from Nueces Bay. Approximately 96,000 m^3 (125,000 y^3) was dredged from Copano Bay in 1965. (Rollin MacRae, Texas Parks and Wildlife Department, Personal Communication, May, 1997). It is estimated that a total of 18.7 million m^3 (24.4 million y^3) was dredged from Nueces Bay in this century (Ward, 1997)

The CCBNEP Human Uses Report (Jones et al., 1997) provides a detailed history of the Port of Corpus Christi and the GIWW. It has been estimated a total of 262 million m^3 (343 million yd^3)of dredge material has been removed from or redistributed within the CCBNEP study area since 1945. However, this does not reflect initial amounts dredged for creation of the GIWW in the early 1940's or the Corpus Christi Channel in 1926. It also does not reflect private dredging. Because initial dredging included large amounts of sediment, the historic trend, in cubic yards of sediment removed, has declined between 1946 and 1992 (Figure III.9).

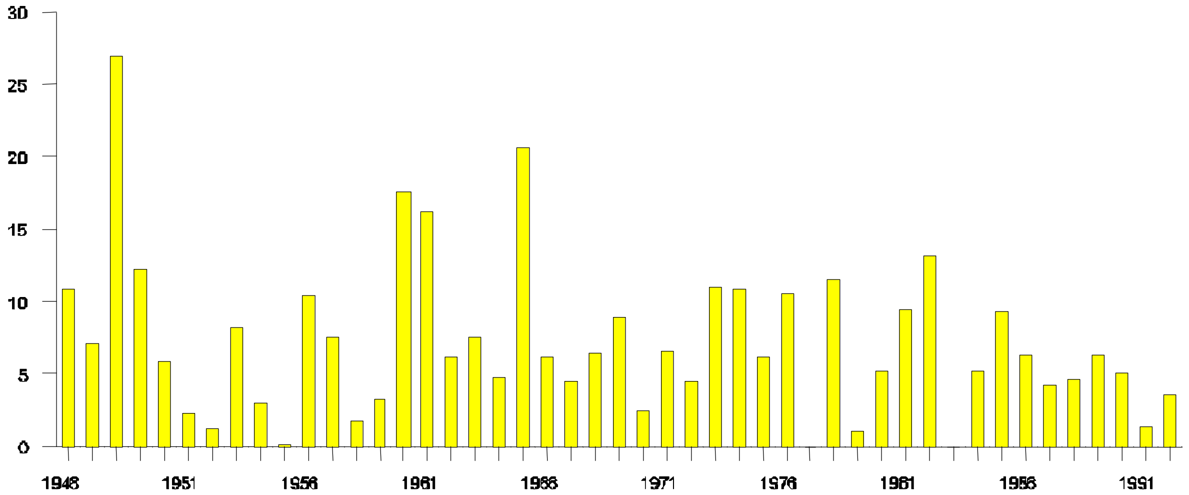

Figure III.9. Cubic yards of sediment dredged from the CCBNEP study area each year between 1946 and 1992. From Jones et al. (1997). Conversion 1 $yd^3 = 0.7646\ m^3$.

A.3.2. Dredge disposal effects on soft-bottom benthos

Dredging represents two problems: excavation and disposal. Disposal has the potential to be more deleterious than excavation. Excavation removes and buries organisms, but organisms can rapidly recolonize a hole. Disposal affects benthic habitats in other ways, e.g., smothering existing communities.

Repeated dredging in one place may prevent benthic communities from fully developing (Dankers and Zuidema, 1995). Excavation destroys the community that previously existed but creates new habitat for colonization. Excavation can actually maintain high rates of macrobenthos productivity (Rhoads et al., 1978). By repeatedly creating new habitat via disturbance, new recruits continually settle and grow. However, these new recruits are always small, surface-dwelling organisms with high growth rates. Large, deep-dwelling organisms that grow slower (and live longer) are lost to the system. In this way, excavation may not cause a decrease in production, but a rather large shift in community structure.

Smaller meiobenthic organisms are particularly resilient to sediment disturbances. It has been shown that after hand-turning benthic sediment, meiofauna communities return to pre-disturbance conditions within one tidal cycle (Sherman and Coull, 1980). Meiofauna are also known to rapidly recolonize oil seep sediments in coastal ocean waters (Palmer et al., 1988). Because of their small size and rapid recolonization rates, meiofauna are probably not as affected by dredging as macrofauna. However, we are not aware of direct studies of the effect of dredging on meiofaunal communities.

Disposal of dredged material may cause ecological damage to benthos. Dredge disposal can adversely affect benthos in three ways: 1) excavation and placement of dredge material creates physical disturbance to benthic ecosystems; 2) moving and turning dredge material may mobilize sediment contaminants making them more bio-available; 3) dredging activities increase amounts of suspended sediment in the water column. Organisms that are buried must vertically migrate or die (Maurer et al., 1986). However, in the top 2 cm of sediment dissolved oxygen decreases and ammonia and sulfide increases at disposal sites. Although vertical migration is possible, most organisms do not survive (Maurer et al., 1986). Open-water disposal in Mobile Bay, Alabama resulted in reduced benthic biomass, reduced redox potential discontinuity depth, and altered sediment relief. However, effects were confined to within 1,500 m of the discharge point and benthos recovered within 12 weeks (Clarke and Miller-Way, 1992).

Dredged material is disposed of in numerous ways. It can be broadcast, unconfined, over open bay bottom; placed in confined areas built on submerged, intertidal or upland areas; or placed unconfined on upland areas (called thin-layer disposal). Dredge material deposited in confined upland disposal sites poses little or no threat to submerged benthos. Thin-layer disposal on uplands may impact intertidal or even submerged areas if rainfall washes material back into the bay. Open bay disposal of unconfined dredge material may have wide-spread ecological effects (e.g., burial and increased turbidity) because material is not confined behind levees. The COE routinely considers alternative means of disposing of dredged material. Alternatives being considered include confined bay disposal, pumping into the open Gulf for beach nourishment, and transport to the Gulf by barge, among others.

Dredge material is typically composed of sand, silt, and clay (Engler, 1990). Other common components include gravel and organic matter. It is estimated 2 - 10% of material dredged in the U. S. is contaminated. Contaminants are especially prevalent in sediments of industrialized harbors. Examples of common contaminants include heavy metals, organohalides, petroleum and chemical by-products. Once contaminated sediment is dredged, contaminants can be dispersed and transported by currents and accumulate in previously uncontaminated areas. Ecological effects of contaminated dredge material depend on the nature of the contaminants and bio-geochemical environment. Contaminants bound to sediment may quickly settle to the bay bottom where they may be in direct contact with benthic animals and may be ingested (Engler, 1990). Characteristics of deposited sediment influence contaminant mobilization and bioavailability (e.g., redox potential, and iron, manganese, and organic matter concentrations). Generally, coarse sediment with low organic matter content, will exhibit lower contaminant mobilization (Engler, 1990).

Corpus Christi Bay sediments contain high concentrations of zinc and cadmium, which is thought to be due to the remobilization of deposited harbor sediments (Holmes et al., 1973). Bacteria may aid deposition of zinc and cadmium from the water column during summer in Corpus Christi Harbor. During winter, bacterial activity is reduced and circulation between bay and harbor increases. Thus, zinc and cadmium previously deposited in the harbor may redissolve and be transported into the bay to be adsorbed by suspended material. Contaminated dredge material can directly affect benthic organisms. Organisms exposed to sediment containing as little as 5% contaminated dredge material exhibit reduced immune capability (Smith et al., 1995). It has also been suggested that dredged sediment was responsible for high amounts of arsenic, 10-146 ug l^{-1} (ppb) in surface water and 1-82 mg kg^{-1} (ppm) in sediments, and was related to toxic responses in eastern oyster, *Crassostrea virginica* (Wirth, et al., 1996). In addition, contaminated harbor sediment has been shown to impair sea urchin embryonic development (Pinto, et al., 1995).

Dredge material may be used to serve numerous beneficial purposes. For example, it has been used to elevate farmland while providing organically rich soil. It has also been used to create rookery islands for colonial water birds. One of the more ecologically important uses of dredged material is wetland creation. Companies that adversely impact wetland habitat may mitigate for these losses by creating new habitat. In some cases, such mitigation involves use of dredge material to increase elevation of bay bottom to a level that will support desired vegetative growth (e.g., seagrasses or marsh grasses). A very large created wetland has just been built in Galveston Bay with dredged material. The largest engineering challenge to be solved in created wetlands is elevation and creek meanders. An important feature of wetlands, to which habitat utilization is directly related, is linear distance of creek edges. Wetlands with straight channels would not provide the same ecological functions as natural meandering creeks. Beneficial use of dredge material is a promising approach, but there is a need for better understanding of landscape ecology of wetlands and how to engineer a wetland.

A.3.3. Dredging effects on submerged aquatic vegetation

Dredge and fill activities are widely recognized as one of the major anthropogenic disturbances contributing to destruction of seagrass meadows. Direct and immediate effect of dredging seagrass meadows is mortality due to removal or burial. In addition, there are indirect losses resulting from disturbance of sediments during dredging operations. Seagrasses have high light requirements (Dunton, 1994), and decreased light availability associated with sediment resuspension has been associated with losses of areal coverage in Laguna Madre (Onuf, 1994). Furthermore, spoil areas are usually not suitable for colonization and growth of seagrasses (Zieman, 1975a). Dredging may also result in hypoxia by increasing biological oxygen demand as organic material exposed by dredging operations undergoes decomposition, which in turn can lead to changes in the redox potential of sediments within meadows (Zieman, 1975a; Nessmith, 1980). Seagrass meadows may also undergo erosion as a result of changes in hydrological conditions due to dredging of navigational channels.

There is evidence that suggests dredging is a causative factor of seagrass loss within the CCBNEP study area. Odum (1963) found turtle grass beds near a dredged area in Redfish Bay had low productivity and biomass in spring and summer following initiation of dredging operations in 1959 and attributed it to decreased light penetration. Direct losses of areal coverage as a result of burial were also reported. In 1960, however, productivity rates were exceptionally high (Figure III.10). Amount of chlorophyll *a*, on a g m^{-2} basis, was also remarkably low during the immediate post-dredging period when compared to the following year (Table III.1).

Table III.1. Chlorophyll *a* in *Thalassia testudinum* beds at varying distances from a newly-dredged navigational channel in Redfish Bay, Texas. Values represent averages of values presented in Odum (1963).

Distance from new channel	Chlorophyll *a* (g m^{-2})	
	Summer 1959	Summer 1960
0	0.003	Station out of water as spoil island
0.25 mile east	0.011	Beds covered with 30 cm of silt; no plants
0.50 mile east	0.058	1.35
0.75 mile east	0.045	0.41
1.0 mile east	0.031	0.25

There is a relationship between changes in seagrass distribution and location of dredging operations in the Laguna Madre (Onuf, 1994). Increased turbidity results from resuspension of dredged sediments from spoil banks by wind-generated waves. Wind-induced wave action is common in the CCBNEP study area due to prevailing southeasterly winds and polar frontal passages. Therefore, dredged sediment leads to decreased light availability to seagrass meadows. Multivariate analysis was used to determine if variations in light attenuation measurements in 1988 and 1989 could be attributed to dredging operations after other variables (wind speed, wind stress, depth and distance from shore, disposal site, and seagrass beds) were accounted for by the model. Light attenuation was greatest in the 1-3 months following dredging operations. In addition, attenuation coefficients were above predicted values for up to 15 months following the disturbance. Although effects of dredging on light attenuation were most pronounced in the vicinity of dredged areas, increased turbidity was evident up to 1.2 km away (Onuf, 1994).

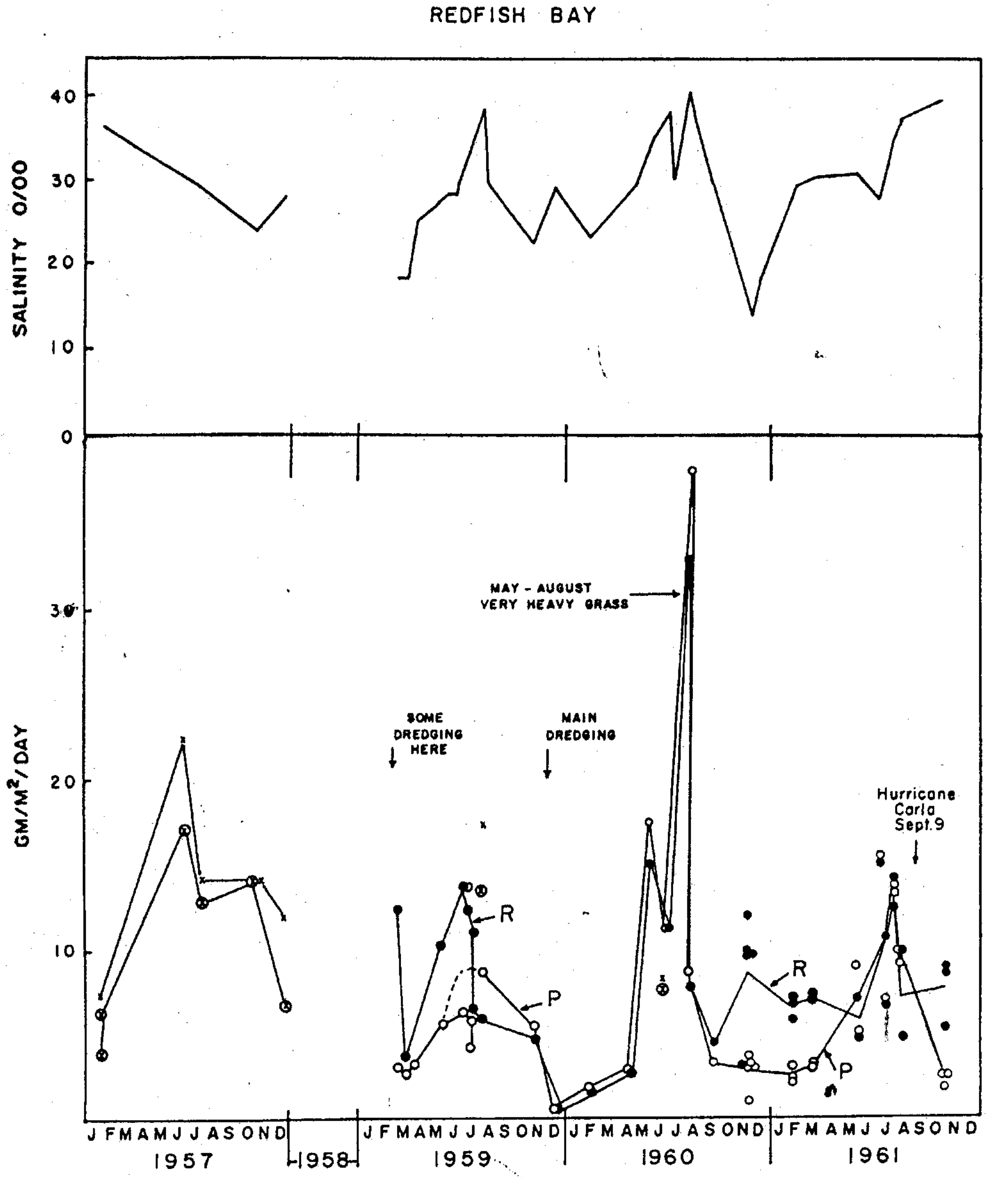

Figure III.10. Record of salinity, gross photosynthesis and total respiration in *Thalassia testudinum* meadows in Redfish Bay, Texas. After Odum 1963.

Deleterious impacts on seagrass beds are of concern because they may be the most important benthic habitat in the CCBNEP study area. This habitat harbors the highest density of invertebrates (Table III.2) and supports high diversity and biomass as well. These habitats support a large and diverse food web including many recreational and commercial species, are nurseries for juvenile fish, provide a refuge from predation, and are a source of food for migrating waterfowl. The most extensive seagrass habitats are in Laguna Madre. Smaller, yet still significant, areas of seagrass habitat can be found in Redfish Bay and along the margins of Corpus Christi and Aransas Bays.

Table III.2. Macrofauna abundances in vegetated habitats. Abundance (individuals m^{-2}) in seagrass beds adapted from Orth et al. (1984).

Location	Seagrass	Vegetated	Unvegetated	Source
Florida	*Thalassia testudinum* and *Halodule wrightii*	33,485	8795	Santos & Simon, 1974
Belize	*Thalassia testudinum*	12,167 6,476	16,750 8,000	Young & Young, 1982
Bermuda	*Thalassia testudinum*	13,580	3,145	Orth, 1971
North Carolina	Zostera marina	923	170	Thayer et al., 1975
Virginia	*Zostera marina*	51,343	1,771	Orth, 1977
North Sea	*Zostera noltii*	5,088	1,043	Reise, 1978
Florida	*Thalassia testudinum & Syringodium filiforme*	3,185	1,754	Stoner, 1980
Australia	Zostera muelleri	1,039	156	Poore, 1982
Florida	Thalassia testudinum	1,611	399	Lewis & Stoner, 1983
Florida	Thalassia testudinum & Halodule wrightii	17,479	5,844	Virnstein et al., 1983
North Carolina	Zostera marina & Halodule wrightii	3,223	720	Summerson & Peterson, 1984
Texas	Halodule wrightii	63,700 26,900 36,900	22,200 12,700 16,300	Conley, 1996

A.4. Recreational boating activities

A.4.1. Boating activities

In 1986, individuals made over 6 million trips to the Texas coast for recreational fishing; 43% of these were to the Nueces and Mission-Aransas estuarine systems, while 21% were to the Laguna Madre (Fesenmaier et al., 1987). Since the mid-1970's, recreational fishing pressure has generally increased in Aransas Bay and Corpus Christi Bay, while decreasing in Upper Laguna Madre

(Ponwith and Dokken, 1996). Presence of recreational boats in the CCBNEP study area is ubiqutous in space and time, because of shallow water habitats for fishing and mild climate.

A.4.2. Effects of recreational boating on benthic habitats

There is little evidence that recreational boating has any significant impact on open-bay bottoms, but the negative impact of recreational boating activities on seagrass habitats are well documented in Florida and the northeastern Gulf of Mexico (Phillips, 1960; Zieman, 1976; Eleuterius, 1987). Recreational boating activity causes direct damage to seagrasses through physical destruction of seagrass leaves and below-ground tissues (roots and rhizomes) by boat propellers. Prop scars tend to occur in areas with < 1 m (3.2') depth at low tide (Zieman, 1976), and are readily visible in seagrass beds from the water surface and through low altitude aerial photography. Eleuterius (1987) indicated that once a propeller scar is created, wave action leads to erosion within the channel resulting in scouring and deepening of the disturbed area. Similarly, Zieman (1976) reported reduced proportions of fine sediments within propeller scars.

Long-term effects of propeller scarring are a function of extent of disturbance and species impacted. There are five seagrass species in the Texas coast: shoal grass (*Halodule wrightii*), turtle grass, widgeon grass (*Ruppia maritima*), manatee grass (*Syringodium filiforme*) and clover grass (*Halophila engelmanii*). Seagrass recolonization rates vary depending on the ecological role of the species. Shoal grass is considered a pioneer species and is a rapid recolonizer of disturbed areas; this capability is attributed to a rapidly growing root and rhizome system. Widgeon grass and manatee grass are also characterized by less developed rhizome systems; therefore, colonizing abilities are similar to those of shoal grass. In contrast, turtle grass is a mature successional, or climax species, characterized by extensive root and rhizome systems that can comprise up to 85% of total plant biomass. Although leaf production rates for turtle grass are usually very high (500-1000 g dry wt m^{-2} yr^{-1}), growth rates of below-ground tissues are very slow.

Rates for seagrass recolonization vary by species. Turtle grass did not recolonize denuded plots 10 months after disturbance (Phillips, 1960). In contrast, shoal grass quickly invaded bare plots. Recolonization rates of turtle grass in prop scars in south Florida and found two-years were necessary before recolonization began and recovery took between three to five years (Zieman, 1976). The slow recolonizing ability of turtle grass was attributed to slow rhizome growth and potentially unsuitable environment for growth in the disturbed area. In an experimental study where 0.25 m (10") wide channels were dug within seagrass beds in Florida, recolonization to normal shoot densities occurred 0.9 - 1.8 years for shoal grass and 3.6 - 6.4 years for turtle grass (Durako et al., 1992). Although shoal grass to recolonization was quicker, recolonization at sparse edges of beds was slower (2.3 - 4.6 years).

The fraction of seagrass habitat impacted by prop scarring can be quantified through analysis of low altitude aerial photographs (scales of 1:2,400 to 1:24,000). In Florida, it is estimated 173,000 acres (70,000 ha) of the 2.7 million acres (1.09 million ha) of seagrass coverage is lightly to severely scarred (Sargent et al., 1995, Table III.3).

Table III.3. Acreage of scarred seagrasses in each region of Florida. Light scarring is < 5 % of the area within a sample polygon, moderate scarring is 5 - 20 %, and severe scarring is > 20 % (Sargent et al., 1991).

Region	Total Seagrass	Light Scarring	Moderate Scarring	Severe Scarring	Total Scarring
Panhandle	48,170	9,970	2,090	200	12,260
Big Bend	826,770	58,630	6,180	540	65,350
Gulf Peninsula	110,260	16,140	21,330	4,580	42,050
Atlantic Peninsula	69,360	250	3,030	490	3,770
South Florida	1,603,700	19,270	15,990	9,650	44,910

The project in Florida was large, to determine the extent of seagrass and propeller scar coverage in the entire Florida coast. Therefore, analysis was based on large scale aerial photography (1:24,000). At the scale of 1:24,000 < 1 % of the propeller scars, which can be identified at a scale of 1:2,400, are recognized. Therefore, extent of scarring in Florida is underestimated by one to two orders of magnitude (Sargent el al., 1991).

Few data exist regarding extent of prop scarring of seagrass beds within the CCBNEP study area. As part of the current report, data regarding the areal extent of seagrass coverage and propeller scarring within the CCBNEP study area were collected in winter 1996-1997. Extent of areal coverage of propeller scars and seagrass habitat was determined from aerial photography at a scale of 1:2,400. Estes Flats, near Rockport, has the greatest area impacted in the CCBNEP study area. Complete details of the study are found in Volume II.

A.5 Effects of turbidity from anthropogenic sources

Coastal and estuarine waters often have relatively high turbidity due to sediment load from terrestrial runoff and subsequent resuspension of those sediments by winds, tides, and currents. Various anthropogenic factors, especially shrimp trawling and dredge disposal, can produce significant suspended sediments loads resulting in very high local turbidities. Natural variations in turbidity can substantially influence distribution of estuarine fishes (Cyrus and Blabler, 1992), even though many species are adapted to variable or even consistently high turbidity levels. Few effects of turbidity on eggs or larvae were demonstrated in laboratory studies during short-term experiments. Savino et al. (1994) exposed lake herring (*Coregonus artedi*) eggs and larvae to 20 - 24 mg l^{-1} (ppm) turbidity typical of shipping activity in the Great Lakes. Jokiel (1989) exposed eggs or larvae of or mahimahi (*Coryphaena hippurus*) to test levels as high as 8000 mg l^{-1} (ppm), whereas 5 - 50 mg l^{-1} (ppm) turbidity is typical of marine mining activities near Hawaii. High suspended sediment levels > 1000 mg l^{-1} (ppm) severely impacted feeding rate of mahimahi, indicating potential long term impacts of high suspended sediment loads. Jokiel (1989), working on dredge spoil disposal plumes in Hawaii, suggested larvae were rarely subjected to suspended sediment loads > 1 mg l^{-1} (ppm), because higher levels were quickly diluted by currents or settled out of suspension. Data of Schubel et al. (1978) indicates fine sediments of bays in the CCBNEP study area may remain in suspension much longer, increasing potential impacts on larval fish. Eventual settlement of resuspended sediments can have significant impacts on benthos, especially in areas of relatively clear water. As little as 1 mm (0.04") of silt has been shown to inhibit settlement of oysters (Galsoff, 1964) and scallop (Stevens, 1987) larvae. Even small resuspension levels evoked by deposit-feeding polychaetes can inhibit suspension feeding molluscs (Rhoads, 1973).

A.6 Effects of hydrocarbon exploration and production

Bay bottoms in the CCBNEP study are utilized for hydrocarbon exploration and production. Exploration activities include anthropogenic disturbances associated with maritime traffic, drilling and seismic testing. Production activities that could cause environment disturbances include drilling, marine construction, maritime traffic, produced waters, and product transportation. Modes of product transportation includes pipeline construction, pumping, and maritime ship and barge traffic.

Many studies have addressed effects of offshore oil and gas development activities on continental shelves, and especially in the Gulf of Mexico. The Gulf of Mexico is responsible for 95% of all offshore oil and gas production in the United States. Long-term effects of offshore oil and gas activities have been reviewed by Boesch and Rabalais (1987). Concerns and issues related to these activities include: 1) chronic biological effects from persistent hydrocarbons in sediment; 2) residual damage form oil spills on coastal wetland, reefs, and vegetated habitats; 3) effects of channelization for pipeline routing and navigation in coastal wetlands; 4) physical fouling of birds, mammals and turtles; 5) cumulative effects on benthos through field development rather than exploratory drilling; 6) effects of produced water discharges into nearshore rather than shelf environments; 7) effects of noise disturbance on birds, mammals and turtles; 8) reduction of fishery stocks due to egg and larval mortality during oil spills; and 9) effects artificial islands. Note two of these concerns deal directly with estuaries even though the list was developed for offshore environments. All issues listed are also concerns in shallow water ecosystems.

Blasts for seismic testing during exploration disturbs bay bottom habitats. Although no formal studies have been found, Chris Onuf (U.S. Geological Survey) has inspected seismic holes in Upper Laguna Madre. Onuf reports (via personal communication) blasts create circular holes in sediments with a raised lip on the outside. The center is excavated to a depth of about 4 m (13'). After one year, three holes were revisited. One hole was completely refilled and seagrass had overgrown the hole. Two other holes had an identifiable 10 cm (4") deep depression, which was filled with dead drift material. The dead material retards re-establishment of seagrasses. The total effect of seismic testing would be a function of density of test holes. At this time, insufficient data exists to assess effects of seismic testing. However, testing in seagrass habitats would be of greatest concern.

Exploratory and production drilling generates drill cuttings. Drill cuttings contain oil- or water-based drill mud lubricants and minerals from deposits. Cuttings are a source of hydrocarbons when oil-based drill mud is used and trace metals when water-based muds are used. Studies on effects of oil or gas platforms in the CCBNEP study area have not been found. However, it is known that platforms in the Gulf of Mexico directly offshore of the CCBNEP study area are contaminated with trace metals (Kennicutt et al., 1996) and benthic organisms are negatively affected within a 100 m (328') radius around the platform (Montagna and Harper, 1996).

Produced water discharges are sources of brines and hydrocarbons. A separator platform was studied in Trinity Bay, Texas (Armstrong et al., 1979; Carr et al., 1996). Oil field brine effluent at this site was responsible for sediment contaminated with hydrocarbons and that were toxic to sea urchin embryos. A produced water discharge in New Bayou, Texas was found to depress macrobenthic populations 107 m (350') downstream and 46 m (150') upstream (Nance, 1991). Produced waters can be especially harmful to rich seagrass habitats. *Thalassia testudinum* was not affected by experimental exposures to produced water, but benthic invertebrate populations were depressed (Weber et al., 1992). The physical disturbance related to greater turbidity and sedimentation reduced light, which causes lower productivity, whereas toxicity causes declines in plant and animal epiphytes (Kelly et al., 1987).

Produced waters have been discharged into Nueces Bay since the turn of the century (D’Unger et al., 1996). As recently as 1995, a total of 16 water discharge sites were producing 2,477,426 l d^{-1} (654,538 gal d^{-1}) in Nueces Bay and Nueces River Tidal areas (Caudle, 1995). The effluents were toxic as were sediments near the discharges (D’Unger et al., 1996). Produced water in Nueces Bay is characterized by low dissolved oxygen and high salinity and temperature, relative to receiving water (Caudle, 1995). In addition, produced water contained high amounts of toxic ammonia, trace metals, aromatic hydrocarbons, oil, and grease. The contaminants result in degraded valuable habitats and biological communties of Nueces Bay.

Transportation of hydrocarbon products poses the greatest environmental risks (Boesch and Rabalais, 1987). There is potential for spills when hydrocarbon products are moved. Since 1984, there have been at least two major spills: an oil pipeline break in Nueces Bay and a cumene spill from a barge collision in the Corpus Christi Ship Channel near Ingleside. Oil spills are especially devastating to valuable vegetated habitats (Levings and Garrity, 1994; Wood et al., 1997) and beaches (Amos et al., 1983).

Pipeline construction in vegetated habitats is another source of concern. Plants can recover due to clonal growth as well as seed production. Salt marshes in South Carolina recovered from pipeline construction in 34 to 36 months. Seagrasses in Upper Laguna Madre may take 14 to 17 years to fully recover ecological functions after a construction project (Montagna, 1993).

A.7 Kills and spills

An indicator of potential disturbance to bay bottoms is reports by the public of fish and wildlife kills and pollution. The TPWD, Resource Protection Division, Kills and Spills Team maintains a data base on reports of fish and wildlife kills and pollution complaints and incidents. The program is popularly known for its “kills and spills” hotline, which receives complaints from the public. People can report kills and spills to the 24-hour TPWD law enforcement dispatcher at (512) 389-4848, or call any local game warden, or one of the regional offices (during business hours). The regional biologist for the Corpus Christi area is Ken Rice at (512) 980-3245. The data base contains about 4,000 records dating back to 1969. The following data was prepared for the CCBNEP study area by Cynthia H. Contreras (TPWD). The data base was queried for fish and wildlife kills and pollution complaints in aquatic environments of the CCBNEP study area. Causes of kills and pollution complaints included: diseases, inorganic compounds, low dissolved oxygen, organic compounds physical damage, scum, solid waste, low temperature and other unknown environmental conditions. Spatial segments in the database are based on Texas Natural Resource Conservation Commission geographical classifications (Figure III.11). Although reported together, kills and spills are different.

Nueces Estuary (Nueces Bay, Corpus Christi Bay, Corpus Christi Inner Harbor, and Redfish Bay) had the highest number (58.1%) of 458 reports from 1969 to 1997 in the CCBNEP study area (Figure III.11). Mission-Aransas Estuary (Copano Bay, St. Charles Bay, and Aransas Bay) had 26.4% of reports, and Upper Laguna Madre and Baffin Bay had only 15.5% of reports. In the TNRCC system, Baffin Bay includes the tertiary bays: Alazan, Laguna Salada, and Cayo del Grullo. The number of reports correlates with population density along the shorelines of the three estuaries in the CCBNEP study area.

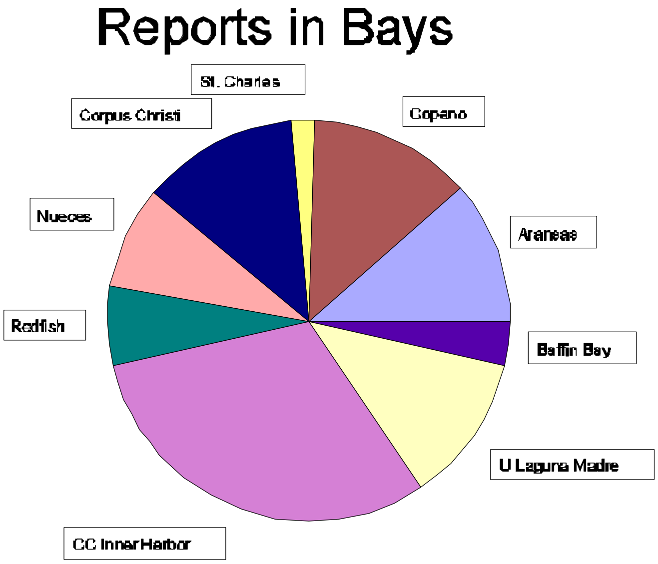

Figure III.11. Reports of kills and spills in bays of the CCBNEP study area. Data provided by Cindy Contreras (TPWD).

Trends of kills and spills reports increased dramatically in the early 1980's and has since remained relatively constant (Figure III.12). There was a spike of increased reports in 1992 and 1993, but recently reports have remained at levels similar to the 1980's. There has been an average of 23 annual reports of spills and kills in the CCBNEP study area since 1980. Most of these incidences can affect water column habitats, but can affect bay bottom habitats as well.

Table of Contents

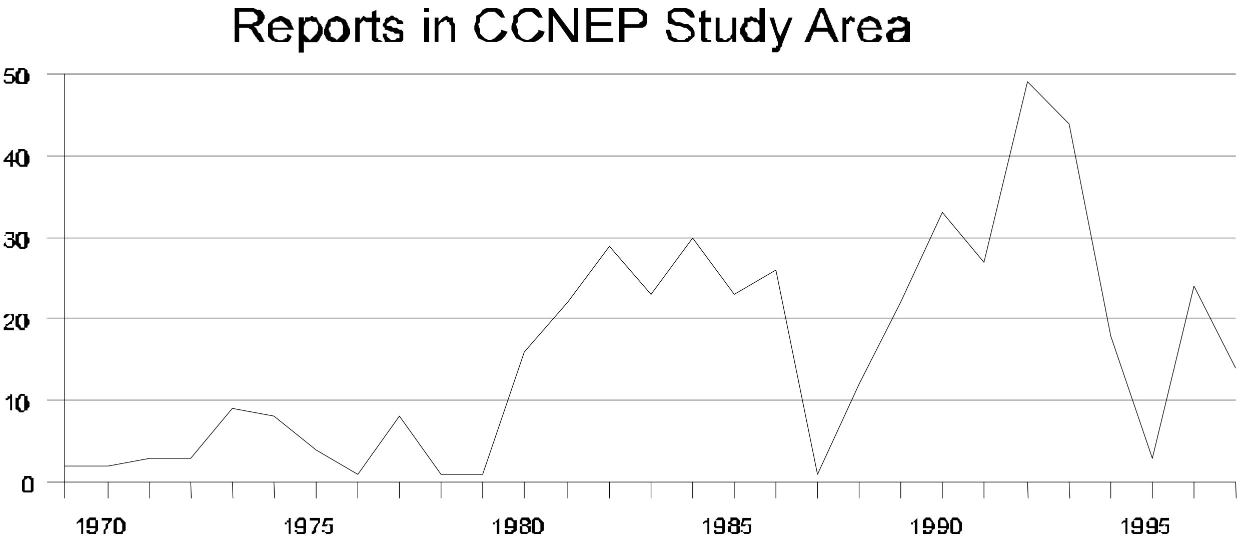

Figure III.12. Total number of kills and spills incidents reported annually in the CCBNEP study area. Data from Cindy Contreras (TPWD).

B. Status and Trends of Natural Disturbances

B.1. Episodic events

B.1.1 Hydrographic setting

Texas bays are typically lagoonal embayments with limited access to Gulf waters. Morphology and hydrodynamics of bays plays a significant role in response of estuarine embayments to natural disturbances, such as storms and floods, which in turn influence salinity, turbidity and siltation regimes. The following summary of hydrodynamic characteristics of Gulf of Mexico estuaries, as applicable to the CCBNEP study area, is adapted from a review by Ward (1980, 1997). A brief discussion of the factors that influence estuarine circulation is also included.

Gulf of Mexico bay systems, including the Mission-Aransas, Nueces and Laguna Madre Estuaries, are naturally broad and shallow with maximum depths of < 5 m (16 ft). However, dredged navigation channels 10 - 12 m (33 - 39 ft) in depth and 50 - 150 m (164 - 492 ft) in width transverse the bays. Access to Gulf waters is limited to narrow inlets.

Freshwater inflow is greatest toward the central area of the Gulf coast (Table III.4). Mean discharge to Mobile Bay is 1820 m^3 s^{-1} (65,000 cfs), while Corpus Christi Bay receives only 25 m^3 s^{-1} (893 cfs). Discharge per unit watershed area (Q/DA), an indicator of aridity, is 2 cm y^{-1} (0.75 in y^{-1}) for Corpus Christi Bay, compared to 41 and 50 cm yr^{1} (16 - 20 in y^{-1}) for Pensacola and Mobile Bays, respectively. Likewise, the ratio of bay volume to discharge (V/Q) is a rough descriptor of the influence of freshwater inflow into a bay; Corpus Christi Bay has a comparatively high value of 620 days.

Table III.4. Physical and hydrological characteristics of selected Gulf of Mexico bays. Adapted from Ward (1980). Conversions: mile = 1.609 km, ft^3 = 0.028 m^3, mi^2 = km^2, 1 in = 2.54 cm.

Bay	Surface Area (km^2)	Volume (10^6 m^3)	Mean Discharge Q (m^3 s^{1})	Drainage Area DA (km^2)	Q/DA* (cm yr^{1})	V/Q** (days)
Tampa	880	2900	40	5800	22	800
Pensacola	265	840	300	22900	41	30
Mobile	1020	3060	1820	114000	50	20
Sabine Lake	240	440	490	48900	32	10
Galveston	1420	4300	275	56500	15	180
Matagorda	1070	3300	115	121900	10	1750
Corpus Christi	435	1340	25	41100	2	620

*Discharge per unit watershed area.
**Bay volume to discharge ratio.

Four primary factors governing bay hydrography are: meteorological forcing, tides, freshwater inflow, and density currents. Density currents are average currents from bay-mouth to bay-head resulting from a salinity gradient along an estuary. The single most important factor influencing bay circulation in Gulf estuaries is meteorological forcing (Ward, 1980). Importance of meteorological forcing is attributed to the relatively small influence of tides. The tidal range is typically 0.3 - 0.7 m (1' - 2') and large surface area to volume ratios of Gulf estuaries (Table III.4). Large bay surface area translates into a long fetch, which is conducive to formation of large wind-driven waves that induce mixing of surface and bottom waters as well as erosion. The Azores-Bermuda High Pressure Zone is responsible for prevailing onshore southeasterly winds during most of the year. In late summer and early fall, the western Gulf region (including the CCBNEP study area) is subjected to tropical disturbances that usually lead to highest rainfall periods of the year. During winter, and to a lesser extent during spring and fall seasons, weakening of the Bermuda High renders the Gulf region more susceptible to northern frontal systems (fronts) resulting from mid-latitude westerlies.

Freshwater inflow to Gulf estuaries is variable within and between years. Inflow tends to be higher during late summer and early fall due to tropical disturbances. Excluding sporadic events such as hurricanes and floods, freshwater inflow to arid regions such as the CCBNEP study area is limited and not usually capable of driving significant currents within an estuary. Nevertheless, input of freshwater to the system affects estuarine salinity gradients and is responsible for formation of density currents. Because intensity of density currents varies as a function of depth cubed, presence of deep navigation channels in Texas estuaries increases their intensity, and affects general circulation patterns. Lastly, large areas of bays are very shallow, therefore evaporation is an important component in determining salinity and hydrographic regime of estuaries in the CCBNEP study area.

B.1.2. Storms

B.1.2.1. History and frequency

The Texas coast is subject to impact of tropical storms and tropical cyclones (hurricanes), particularly during June through November. Hurricanes are storms with winds in excess of 74 mph, whereas tropical storms have winds less than hurricane intensity. During the 114-year period between 1871 and 1984, 26 tropical storms and 40 hurricanes made landfall on the Texas coast (Bomar, 1983, Table III.5).

Based on hurricane occurrence between 1871 and 1974, average interval between hurricane landings in Texas is about two years, while tropical storms are expected every three years (Henry et al., 1975). Any 50-mile section of the coast will experience a tropical storm or minor hurricane every three to six years (Henry, 1975) and an extreme hurricane every 15 to 20 years (Bomar, 1983). Within a one-year period, likelihood of a storm making landfall is 12.5% for the area between Port Arthur and Freeport, 17% for Freeport to Port O’Connor, 12.5% for Port O’Connor to King Ranch, and 14% for King Ranch to Brownsville (Bomar, 1983). Likewise, annual probability of a tropical storm striking the Texas coast in the vicinity of the CCBNEP study area is estuarine-specific. For the Mission-Aransas and Nueces estuaries, there is 10% chance of a tropical storm, a 16% chance of a minor hurricane, and a 7% chance of a major hurricane within a one-year period. For the Laguna Madre estuary, there is an 11% chance of a tropical storm, an 18% chance of a minor hurricane and a 5% chance of a major hurricane (Henry et al., 1975).

Table III.5. Number of tropical storms and hurricanes making landfall on the Texas coast between 1871-1984. Adapted from (Bomar, 1983).

Storm	Month					
	June	July	August	September	October	TOTAL
Tropical	7	6	2	9	2	26
Hurricane	7	6	14	10	3	40

Despite probabilities, hurricane occurrences are sporadic. Between 1903 to 1908, no significant hurricanes made landfall on the Texas coast. In addition, no major hurricanes have impacted the CCBNEP study area since 1980 as of this writing (1997). In contrast, five major hurricanes made landfall in Texas between 1961 and 1980: Hurricane Carla (1961), Hurricane Beulah (1967), Hurricane Celia (1970), Hurricane Fern (1971) and Hurricane Allen (1980) (Table III.6). Hurricane

Carla was the largest, with maximum wind speed, surge height, and lowest barometric pressure among these hurricanes (Table III.6). Beulah had the highest amount of rainfall.

Table III.6. Local effects of landfall of major hurricanes on the Texas coast from 1961-1980. Compiled from ACOE, 1972; Brown et al., 1974; ACOE, 1981; Bomar, 1983; Ellis, 1986. Conversions: mi = 1.609 km, ft = 0.3048 m, in = 2.54 cm.

Year	Date	Name	Location	Max. Wind speed (mph)	Surge Height (ft)	Max. Rainfall (in)	Lowest Pressure (in)
1961	Sept. 11	Carla	Pt.O'Connor	175	22	16.2	27.49
1967	Sept. 20	Beulah	Brownsville	140	12	30.0	27.98
1970	Aug. 3	Celia	Corpus Christi	130	9	6.3	27.80
1971	Sept. 10	Fern	Rockport	90	6	22.7	28.98
1980	Aug. 10	Allen	Pt. Mansfield	138	12	20	27.91

Although the exact location of hurricane landfall may vary, the consequences affect large areas of the Texas coastline. Hurricane conditions can be experienced at distance from the eye of the storm. Except for Hurricane Fern, the major hurricanes since 1961 caused hurricane strength winds greater than 86 mph (138 km h^{-1}) and surge heights greater than 6.8' (2 m) in Corpus Christi Bay (Table III.7).

Table III.7. Effects of major hurricanes on Corpus Christi, Texas between 1961 and 1980. Adapted from (COE, 1968; COE, 1971; COE, 1972; Brown et al., 1974; Bomar, 1983; Ellis, 1986.

Year	Date	Name	Location	Max. Wind speed (mph)	Surge Height (ft)	Max. Rainfall (in)	Lowest Pressure (in)
1961	Sept. 11	Carla	Pt. O'Connor	86	6.8	1.2	28.88
1967	Sept. 20	Beulah	Brownsville	86	7.3	14.4	29.40
1970	Aug. 3	Celia	Corpus Christi	130	9.0	6.3	27.80
1971	Sept. 10	Fern	Rockport	70	3.9	6.5	28.98
1980	Aug. 10	Allen	Port Mansfield	92	9.0	13.2	29.26

B.1.2.2. Effects of storms

Oppenheimer (1963) did not find observable damage to seagrass beds of Redfish Bay (60 miles south of the storm's eye) after passage of Hurricane Carla at Port O'Connor. However, he noted silt had been redistributed within the bay as a result of 3 foot waves caused by the storm, and bays were turbid four days after storm landfall. Thomas et al. (1961) reported large quantities of turtle grass along the shoreline of Biscayne Bay as a result of Hurricane Donna (1960). However, there did not appear to be significant damage to seagrass beds. In contrast, van Tussenbroek (1994) documented a decrease in number of short shoots in some populations of turtle grass in Puerto Morelos, Mexico after passage of Hurricane Gilbert (1988). This was attributed to depositing or removal of sediments during storm passage. Eleuterius (1987) made references to destruction of seagrass meadows in Mississippi Sound due to scouring or burial caused by Hurricane Camille (1969) and along the Alabama coast as a consequence of Hurricane Frederick (1979). Therefore, if a hurricane passes directly over seagrass habitat, burial or removal is likely.

B.1.3. Fronts

B.1.3.1. Occurrence of fronts

From 15 to 30 Arctic or Pacific fronts pass through Texas between December and February each winter (Brown et al. 1976). Wind stress associated with north or northeasterly winds produces significant water movement in shallow Texas bays. As wind pushes water toward southern areas of a bay, higher water levels develop toward the bay-mouth accompanied by a depression of water level at the bay-head. As a result of the pressure gradient formed, water flows from bay-head to bay-mouth, and up to half of the estuary's volume can be emptied within a 24 hour period, potentially affecting the salinity regime (see below; Ward, 1980). This phenomenon is called "setup" and is responsible for water-level changes associated with frontal passage. Fronts also contribute to formation of wind-driven circulation patterns about which limited information is available.

B.1.3.2. Effects of fronts

There is no data on effects of fronts on bay bottoms or seagrass habitats. However, due to extensive root and rhizome system of seagrasses, it is unlikely fronts cause significant damage such as uprooting of plants. However, fronts cause massive resuspension of sediments and high turbidity, which may decrease light availability to seagrasses. Effects of reduced light on productivity are well known and discussed elsewhere.

Another possible effect of fronts is burial of seagrass beds by detritus. The effect is due mainly to wind, which moves seagrass detritus. Concentration of dead seagrass over a live meadow can bury the living shoots and kill them. It is possible that bare patches in a seagrass bed is due to this phenomenon.

B.1.4. Floods

B.1.4.1. Flooding due to hurricane storm surge and onshore winds

Flooding due to storm surge and onshore winds caused by hurricanes can be extensive along coastlines with wide, gently sloping shelves with concave geography, e.g., Texas, (Brown et al., 1974). As a hurricane approaches the coast, sea level may rise for 32 - 64 km (10 to 20 mi) to the right of the storm's track. In addition, gently sloping bottoms characteristic of Texas bays may lead to sea level increases > 3 m (10 ft). Furthermore, wind stress can also increase sea level within bays by as much as 1 m (3 ft) upwind of the storm track; an increase of 1 - 1.2 m (3 - 4 ft) in the vicinity of the storm is reflected in high sea levels along several hundred miles of coast (Brown et al., 1974).

Pronounced effects of storm surges and hurricanes winds in the CCBNEP study area were observed in 1961 when Hurricane Carla swept through Port O'Connor, Texas. Hurricane Carla generated waves greater than 1 m (3 ft) in Redfish Bay; and tides were 30 cm (1 ft) above mean sea level for four days following landfall (Oppenheimer, 1963). As a result, 526 km^2 (203 mi^2) in and around Corpus Christi were inundated and tidal flooding extended 16 km (10 mi) up the Mission, Aransas and Nueces river valleys. Together, Hurricanes Beulah and Carla flooded 82 ha (3,164 mi^2) with seawater; most of San Jose, Mustang and Padre Islands were flooded. According to Brown et al. (1974), if Carla had made landfall in Port Aransas, an estimated 10-15 % more area in the vicinity of Corpus Christi would have been flooded by tides 4.5 - 6 m (15 - 20 ft) above mean sea level.

Unlike Hurricanes Carla and Beulah, the most dramatic effects of Celia (1970) were not due to flooding from rains or storm surge, but from extreme winds as it made landfall in Corpus Christi. Rainfall associated with Hurricane Carla was confined to a 80 km (50-mi) radius around the storm's center; many cities proximal to Corpus Christi received little or no rainfall. The highest storm surge recorded was only 2.7 m (9 ft). However, winds gusting at 259 km h^{-1} (161 mph) were recorded in Corpus Christi an hour after the storm made landfall. Hurricane strength winds were seen as far inland as Del Rio and gusts over 80 km h^{-1} (50 mph) occurred in Big Bend (Bomar, 1983).

B.1.4.2. Flooding due to hurricane and tropical storm precipitation

In addition to storm surge, hurricanes and tropical storms often produce heavy rainfall. Precipitation due to a single hurricane in August or September may lead to greater rainfall within that month than during the entire spring season. Therefore, average monthly precipitation values tend to be biased toward months with greatest hurricane frequencies. For example, in September 1967, Hurricane Beulah deposited 81 cm (32 in) of rain over three days on much of South Texas (Brown et al., 1974).

Rainfall associated with hurricanes and tropical storms lead to peak late summer or early autumn freshwater inflows from rivers into estuaries. Following hurricanes Beulah (1967) and Fern (1971), inflows of 981 million m^3 and 755 million m^3 (796,000 and 612,000 acre-ft), respectively, were recorded from the Mission and Aransas Rivers into the Mission-Aransas Estuary, compared to average values of 150 million m^3 (122,800 acre-ft) (Tunnell et al., 1996). Similarly, hurricane Carmen (1974) made landfall in Louisiana and influenced Texas weather for two weeks following landfall. A total of 43 cm (17 in) of rain were recorded in Papalote Creek drainage, a tributary of Aransas River (Brown et al., 1974).

B.1.4.3. Non-hurricane related flooding

Flooding events not directly related to tropical cyclone activity also occur along the Texas coast. In 1935, major flooding due to abnormally high rainfall decreased salinities of Texas bays, converting them into “freshwater lakes”. This event reduced salinity of Gulf waters for several miles offshore. During 1948-1956, a severe drought caused record-high salinities in bay waters of the CCBNEP study area; the drought was ended by a major flood in 1957. As a result of the 1957 flood, influx of freshwater dropped salinities in Mesquite and Aransas Bays from 40 ‰ to 2 - 4 ‰ (Parker, 1959). In addition, inundation of low lying areas such as tidal flats, south sides of bays and barrier islands occurs with passage of fronts during winter and early spring (Brown et al., 1974).

B.2. Long-term events and natural processes

There are events that occur over relatively long time scales, comparable to the time scales of storms and floods, which can nevertheless be considered disturbances. Long-term events occur at scales of years and decades. Among these events are: variations in freshwater inflow, turbidity, hypoxia, and brown tide; all of which may have both natural and anthropogenic causes. Scientists at the University of Texas Marine Science Institute (UTMSI) have been monitoring soft-bottom benthic communities in vegetated and unvegetated habitats in the CCBNEP study area for many different purposes since 1987. Studies have been primarily focused on defining effects of freshwater inflow (Montagna and Kalke, 1992; 1995; Mannino and Montagna, 1995; Montagna and Li, 1996), but other studies have been performed on effects of potential pollutants and channelization (Martin and Montagna, 1995), brown tide (Montagna et al., 1993; Conley, 1996; Buskey et al., 1997), and hypoxia (Ritter and Montagna, unpublished). Together, these studies are combined and reviewed to assess status and trends of unvegetated bay bottom habitats as indicated by changes in benthic macrofaunal community structure, biomass, and production. Many samples have been taken and analyzed in the Nueces and Laguna Madre-Baffin Bay estuaries, but unfortunately, very little information is available about the Mission-Aransas Estuary.

Measurement of long-term benthic community changes have been made since 1987 in the Nueces Estuary (Figure III.13) and since 1988 in Laguna Madre-Baffin Bay Estuary (Figure III.14). Biomass is given in terms of g dry weight m^{-2}. Diversity is given as Hill’s N1 value, which is calculated as the exponentiated form ($^{H'}$) of Shannon-Weaver’s diversity index (H’). The N1 diversity index is useful, because it is interpreted as number of dominant species, and low values approach 1. There are both similarities and differences in trends of data between the two estuaries. In general, highest biomass, abundance and diversity is found in Laguna Madre, lowest values are found in Baffin Bay, and intermediate values are found in Corpus Christi and Nueces Bays.

There is a great amount of year-to-year variability, but certain changes are correlated to specific long-term events or trends. For example, decreasing diversity in Laguna Madre from 1990 is associated with onset of brown tide. There was an increase of infaunal biomass, abundance and diversity in

Corpus Christi Bay during the 1992 - 1993 El Niño. The increase is due to lower salinities, which follow high precipitation due to El Niño. The effect of El Niño is less evident in Laguna Made and Baffin Bay. However, increased abundance and diversity was found in Laguna Madre in 1992 - 1993, and an increasing trend in diversity was found from 1992 - 1995 in Baffin Bay. In general, El Niño had positive long-lasting trends in both ecosystems.

The focus of Section III.B.1.2 has been on the effects of storms. It is obvious that storms can act as a natural disturbance. However, an interesting question is: "what effect does a lack of storms have?" We have found no studies that could be used to answer this question. However, based on disturbance and succession theory (Section I.C), it is likely that storms have a beneficial role in ecosystems akin to resetting a clock. A natural disturbance, like a storm, can reopen niche space, thus making habitat available to pioneering species. Although community structure would change, system productivity would increase. In addition, it is likely that nutrient regeneration would be enhanced by creation of additional detritus. In contrast, stasis in a system might lead to stagnation. Is it possible that lack of a major hurricane was part of the cause of the brown tide bloom?

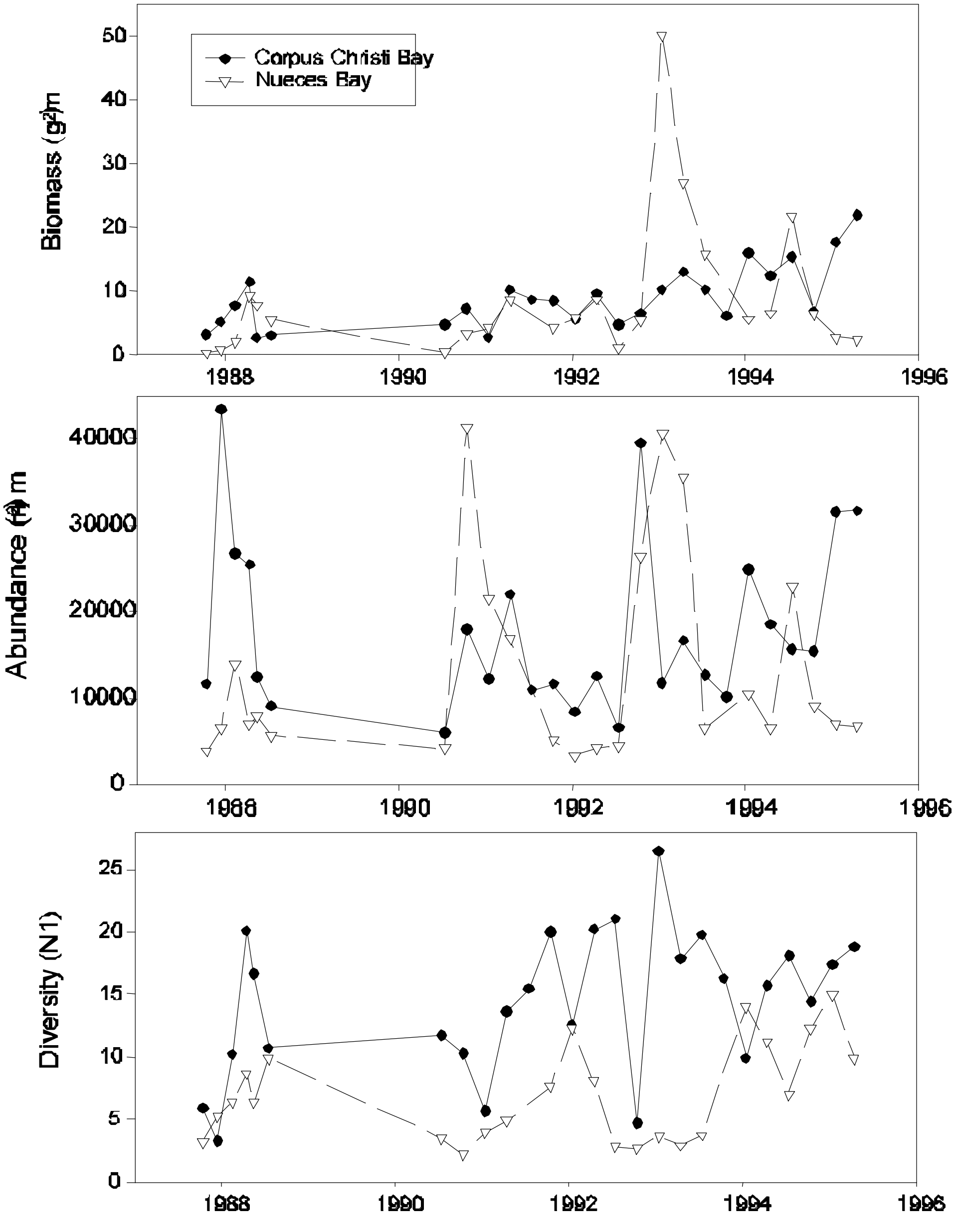

Figure III.13. Macrofauna long-term trends for the Nueces Estuary. Units for diversity are Hill's N1 value. Biomass is dry weight. Montagna, unpublished data.

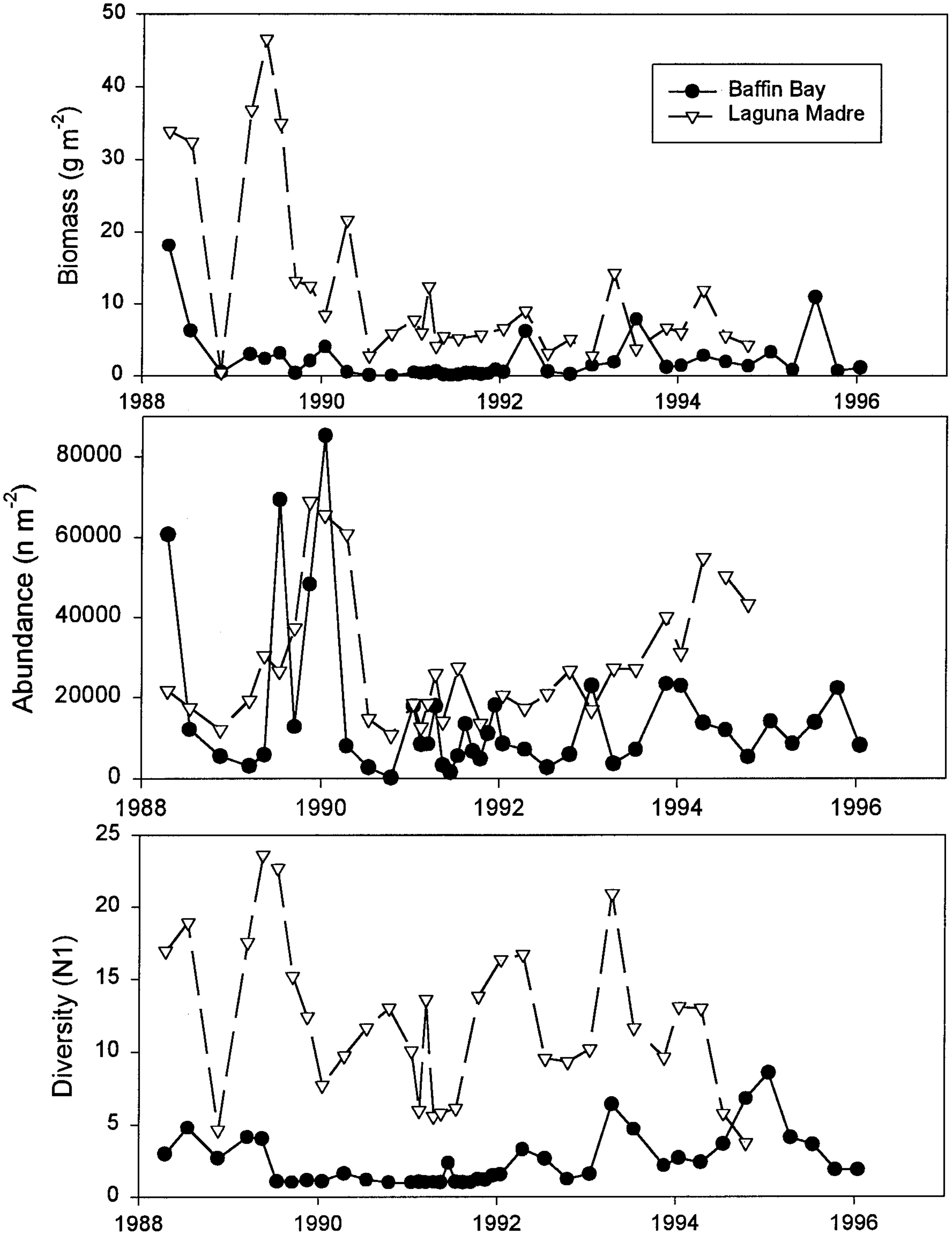

Figure III.14. Macrofauna long-term trends for the Laguna Madre-Baffin Bay Estuary. Units for diversity are Hill's N1 value. Biomass is dry weight. Montagna, unpublished data.

B.2.1. Freshwater inflow and salinity

B.2.1.1 Status and trends

Within the CCBNEP study area, most significant changes in salinity are due to isolated freshwater pulses rather than seasonal variations of freshwater inflow. Most freshwater pulses occur in early fall and coincide with tropical cyclone activity; however, rainfall due to passage of frontal systems in late spring may also result in noticeable salinity changes (Orlando et al., 1991).

Average input of freshwater to the Mission-Aransas, Nueces and Upper Laguna Madre estuarine systems is reported in Table III.8. Amount of gauged freshwater inflow for Mission-Aransas and Laguna Madre estuaries accounts for 15% and 17%, respectively, of total annual freshwater input; for the Nueces Estuary gauged inflow comprises about 60% of total. Precipitation has a greater influence in Mission-Aransas and Laguna Madre systems than in Nueces Estuary (Orlando et al., 1991).

Table III.8. Sources of freshwater input into the three estuarine systems of the CCBNEP study area. Estimates of the relative contribution of gauged and ungauged inflow and direct precipitation are presented (Orlando et al., 1991).

Freshwater Inflow Source	Percent Contribution of Source to Each Estuary (%)		
	Mission-Aransas	Nueces	Laguna Madre
Gauged	15	60	17
Ungauged	39	28	N/A
Direct Precipitation	46	8	65

Precipitation and evaporation rates vary among the three estuarine systems within the CCBNEP study area. On average, precipitation in the Mission-Aransas estuary is 89 cm y^{1} (35 in y^{-1}), while the Nueces and Laguna Madre estuaries receive 74 cm yr^{1} (29 in y^{-1}). In addition, average annual evaporation rates in Laguna Madre is slightly higher than in Mission-Aransas and Nueces estuaries, at 158 cm yr^{1} (62 in y^{-1})(Table III.9).

Average yearly surface and bottom salinities are variable among estuaries, as well as having large year to year variations (Figure III.15). On average, salinities in Mission-Aransas and Nueces estuaries are lower and display higher variability than in Laguna Madre (Table III.9). Mean surface salinity during this period was about 18 ‰ for Mission-Aransas Estuary, 28 ‰ for Nueces Estuary, and 34 ‰ for Upper Laguna Madre Estuary (Figure III.15). Higher salinities in Laguna Madre result from a large surface area and shallow average depth, as well as a lack of major river discharge into the estuary and limited access to exchange with Gulf waters (Montagna et al., 1996). Evaporation often exceeds inflow in Laguna Madre, so it is characterized by hypersalinity.

Table III.9. General characteristics of bays in the CCBNEP study area. Compiled from Tunnell et al., 1996 and Orlando et al. 1991. Conversions: mi = 1.609 km, ft = 0.3048 m, in = 2.54 cm.

Characteristic	Estuary		
	Mission-Aransas	Nueces	Laguna Madre
Size (km^{2})	540	500	1500
Depth (m)	2	2	1
Rainfall (cm yr^{1})	89	74	74
Evaporation (cm yr^{1})	151	151	158
Surface Salinity (‰)	11.2-17	14.8-31	30.3-34.4
Bottom Salinity (‰)	12.3-19.3	16.6-30.6	31.3-37.0

In the three estuaries of the CCBNEP study area, salinity differences between the low-inflow period (June through August) and the high-inflow period (September through November) are small. Between 1970 and 1988, average surface and bottom salinities in the Mission-Aransas estuary were only 5‰ higher in the low-inflow period than in the high-inflow period. Likewise, during the low-inflow period, average salinity was 2‰ higher in the Nueces estuary and 3‰ higher in the Laguna Madre estuary (Orlando et al., 1991). Lower salinities in the high-flow period result from lower evaporation rates and more frequent freshwater pulses.

Orlando et al. (1991) used existing salinity data to evaluate relative influence of freshwater inflow, tides, wind evaporation, tropical systems and navigational channels to salinity regimes of Texas bays Each of these forcing functions have relatively different importance in different estuaries (Table III.10).

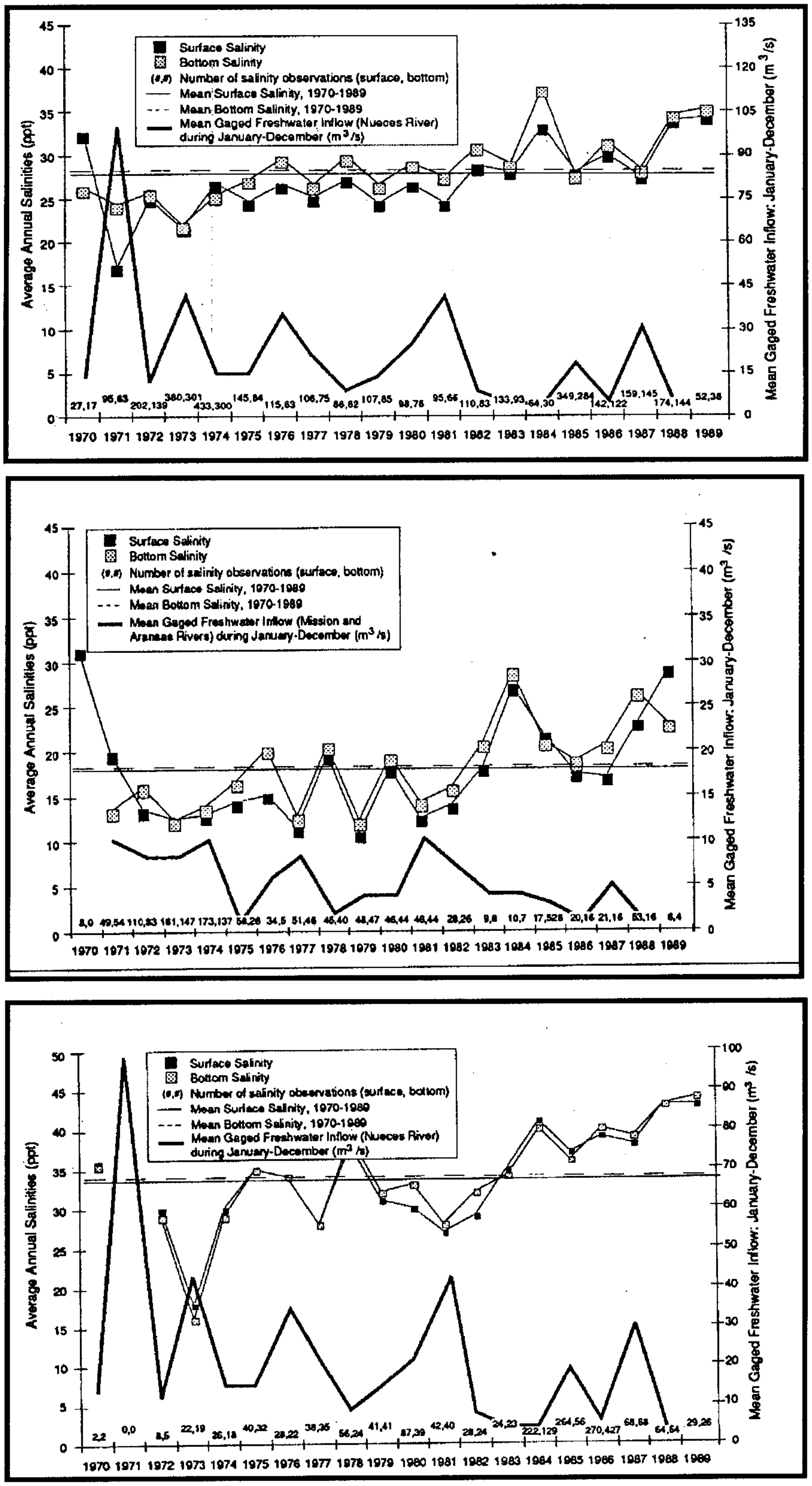

Figure III.15. Average annual salinity and gauged freshwater inflow in Mission-Aransas, Nueces and Upper Laguna Madre Estuaries. After Orlando et al. (1991).

Table III.10. Response of salinity to forcing mechanisms. Abbreviations: D: Dominant factor accounting for greatest range in salinity variability. S: Secondary factor having an influence on salinity variability. M: Minor factor having a detectable influence on salinity variability. Adapted from Orlando et al., 1991.

Estuary	Mechanism	Days	Weeks	Months to Seasons	Year to Year
Mission Aransas	Inflow		**M** (freshet)	**S** (long flushing)	**D** (wet/dry)
	Tide				
	Wind	**M** (frontal passages)	**M** (pressure systems)		
	Evaporation			**M**	
Nueces	Inflow		**M** (long flushing)	**M**	**D** (wet/dry)
	Tide				
	Wind		**S**	**M** (seasonal)	
	Evaporation			**D**	
	Channel				
	Tropical Storm	**M**	**M**	**M**	
Upper Laguna Madre	Inflow				**M** (wet/dry)
	Tide			**M** (seasonal)	
	Wind	**M**	**S** (pressure systems)	**D** (seasonal)	
	Evaporation			**S**	
	Tropical Systems		**M**	**S**	

Due to morphology and hydrology of South Texas estuaries, sporadic events of abnormally high or low freshwater input may have a large effect on salinity regimes within an estuarine system. The following is a historical compilation of events which resulted in abnormally low or high salinities in the CCBNEP study area.

- Between 1950 and 1957, a severe drought caused minimal freshwater inflow and high evaporation rates in Aransas Bay, leading to hypersaline conditions. The drought was terminated in the spring of 1957 when extensive floods caused salinities in Mesquite and Aransas Bays to drop from 40 ‰ to a range of 2 - 4 ‰ within three weeks (Parker, 1959).

- In 1974, a freshwater pulse estimated to occur every 10 to 20 years caused salinities in Mission-Aransas Estuary to decrease to 5 ‰ (Orlando et al., 1991).

- In August 1980, Hurricane Allen resulted in 50-year peak flows in San Fernando Creek. In addition, the storm lowered salinities to 3 ‰ in Cayo de Grullo (within Baffin Bay) during August, and salinities remained less than 20 ‰ in Baffin Bay through November (Orlando et al., 1991).

- In Baffin Bay salinities were commonly 60 - 80 ‰ before construction of the GIWW, and salinities could decrease to as low as 2 ‰ by local thunderstorms (Gunter, 1945)

B.2.2.2. Effects of inflows

Bursts of productivity in Nueces Bay during wet periods and declines in dry years show the importance of freshwater inflow for maintaining benthic productivity (Figure III.13). The positive effect of inflow on secondary productivity has been demonstrated in past studies in Lavaca (Kalke and Montagna, 1991) and San Antonio Bays (Montagna and Yoon, 1991).

Nueces and Guadalupe Estuaries were contrasted to test the hypothesis that benthic standing crops are enhanced by freshwater inflow (Montagna and Kalke, 1992). Assuming predation pressure was similar in both estuaries, higher standing crops could be equated with higher secondary production. Guadalupe Estuary had 79 times more freshwater inflow than Nueces Estuary, and much lower salinity. Guadalupe Estuary had higher macrofaunal densities and biomass than Nueces Estuary. Density and biomass increased with decreasing salinity within Guadalupe Estuary but increased with increasing salinity in Nueces Estuary due to invasion by marine species. Macrofauna diversity increased with salinity, both within and between estuaries. Macrofaunal responses indicated that increased freshwater inflow stimulated secondary production. Macrofaunal diversity decreased with lower salinity within and between estuaries, so enhanced productivity was due to increases by freshwater and estuarine species that can tolerate low salinities. Meiofauna, however, responded differently than macrofauna. Meiofaunal densities were higher in low-inflow Nueces Estuary, and increased with increasing salinity in both estuaries. Higher macrofaunal densities were associated with lower meiofaunal densities, which could be due to either increased macrofaunal competition with or predation on meiofauna, or a lack of low-salinity tolerance by meiofauna.

Table III.11. The relationship between freshwater and marine-influenced zones in the high-inflow Guadalupe and low-inflow Nueces Estuaries. For each parameter the overall average of all sampling periods between 1987 and 1988 are given for the freshwater influenced stations and the marine-influenced stations. Adapted from Montagna and Kalke (1992).

Variable	Estuary			
	Guadalupe (1987)		Nueces (1987-8)	
	Fresh	Marine	Fresh	Marine
Salinity (ppt)	1.4	6.9	32	33
Meiofauna Density (no. $\times 10^6$ m^{-2})	0.25	1.12	1.18	3.80
Macrofauna Density (no. $\times 10^3$ m^{-2})	30.1	8.35	7.48	19.88
Biomass (g dry wt m^{-2})	5.98	3.36	4.31	4.41
Diversity (no. species)	15	26	26	59

Seagrasses are tolerant of large fluctuations in salinity. Of the most common species within the CCBNEP study area, shoal grass is the most euryhaline, followed by manatee grass and turtle grass (McMillan and Moseley, 1967). Turtle grass can tolerate salinities as low as 3.5‰ and as high as 60‰; however, plant stress is manifested through defoliation (Zieman, 1975a). The optimum salinity range for turtle grass is 24-35‰. Zieman (1975b) reported decreased productivity in turtle grass below and above 30‰. Although seagrass species are tolerant of fluctuations in salinity, there is some evidence that salinity may play a role in seed germination, and there may be ecotypic variations within a species that determine germination success (Koch and Dawes, 1991).

B.2.2. Suspended sediments loads

High suspended sediment loads can be generated by natural processes such as storms and floods as well as human activities such as shrimp trawling and dredge spoil disposal. The following is a description of turbidity and sedimentation related to natural events.

B.2.2.1. Turbidity

Wind is the primary mechanism driving turbidity characteristics and spatial structure in Corpus Christi Bay (Shideler, 1980). Under influence of prevailing southeasterly winds, characteristic of the study area, fluvial sediments delivered by Nueces River to Nueces Bay are trapped within the bay. However, during passage of winter fronts, sediments trapped within Nueces Bay are flushed to Corpus Christi Bay and thereafter dispersed southward by wind-driven waves and currents. Therefore, input of suspended sediments into Corpus Christi Bay is determined by wind speed and direction. Frequency of wind speed and direction in Corpus Christi Bay during the annual cycle is presented in Table III.12.

Table III.12. Wind speed and direction in Corpus Christi. Adapted from Naval Oceanography Command Detachment (1986) as cited in PCCA (1993). Conversion: kts = 1.852 km h^{-1}.

Month	Percent Frequency of Wind Speed and Direction in Corpus Christi																
	N		NE		E		SE		S		SW		W		NW		Mean
	%	kts	%	kts	%	kts	%	kts	%	kts	%	kts	%	kts	%	kts	kts
Jan	10	19	20	17	25	14	20	14	5	14	5	11	10	10	5	18	15
Feb	20	18	15	17	18	14	18	14	18	15	3	11	3	13	3	17	15
March	20	17	10	14	10	13	23	14	20	14	5	10	5	11	5	18	14
April	4	17	16	13	18	13	42	15	16	14	1	11	1	11	1	13	13
May	9	16	5	13	18	12	40	14	20	14	2	9	2	12	3	12	13
June	5	10	7	13	8	12	40	13	30	14	3	12	2	8	4	10	12
July	3	9	4	8	4	11	43	12	37	12	3	9	1	5	2	8	9
Aug	1	8	1	12	10	10	37	11	38	12	3	11	3	8	4	10	10
Sept	6	14	4	13	32	11	34	12	18	14	3	13	1	10	1	11	12
Oct	5	17	20	14	30	11	35	12	3	12	3	12	2	13	1	9	13
Nov	5	18	20	15	20	11	30	16	20	15	2	13	1	9	1	13	14
Dec	8	18	20	14	10	12	20	15	20	15	5	12	10	12	6	18	15
Avg.	8	15	12	14	17	12	32	14	20	14	3	11	3	10	3	13	13

Wind speed and direction also effect turbidity structure, i.e., water column transmissivity and suspended solids in Corpus Christi Bay. A detailed study of turbidity structure was performed along a salinity gradient from Nueces Bay to Aransas Pass (Shideler, 1984). Turbidity was measured as percent transmissivity in 0.25 m (12 in) of water (% T 0.25 m), while suspended sediment concentrations (mg l^{-1} or ppm) were determined from water samples. Other data collected included current direction derived from drifters, sediment grain size, temperature and salinity. Two scenarios were derived that illustrate the turbidity structure due to the two characteristic wind patterns: 1) passage of a winter front with northerly winds and, 2) prevailing south/southeasterly winds characteristic of the spring and summer in the CCBNEP study area.

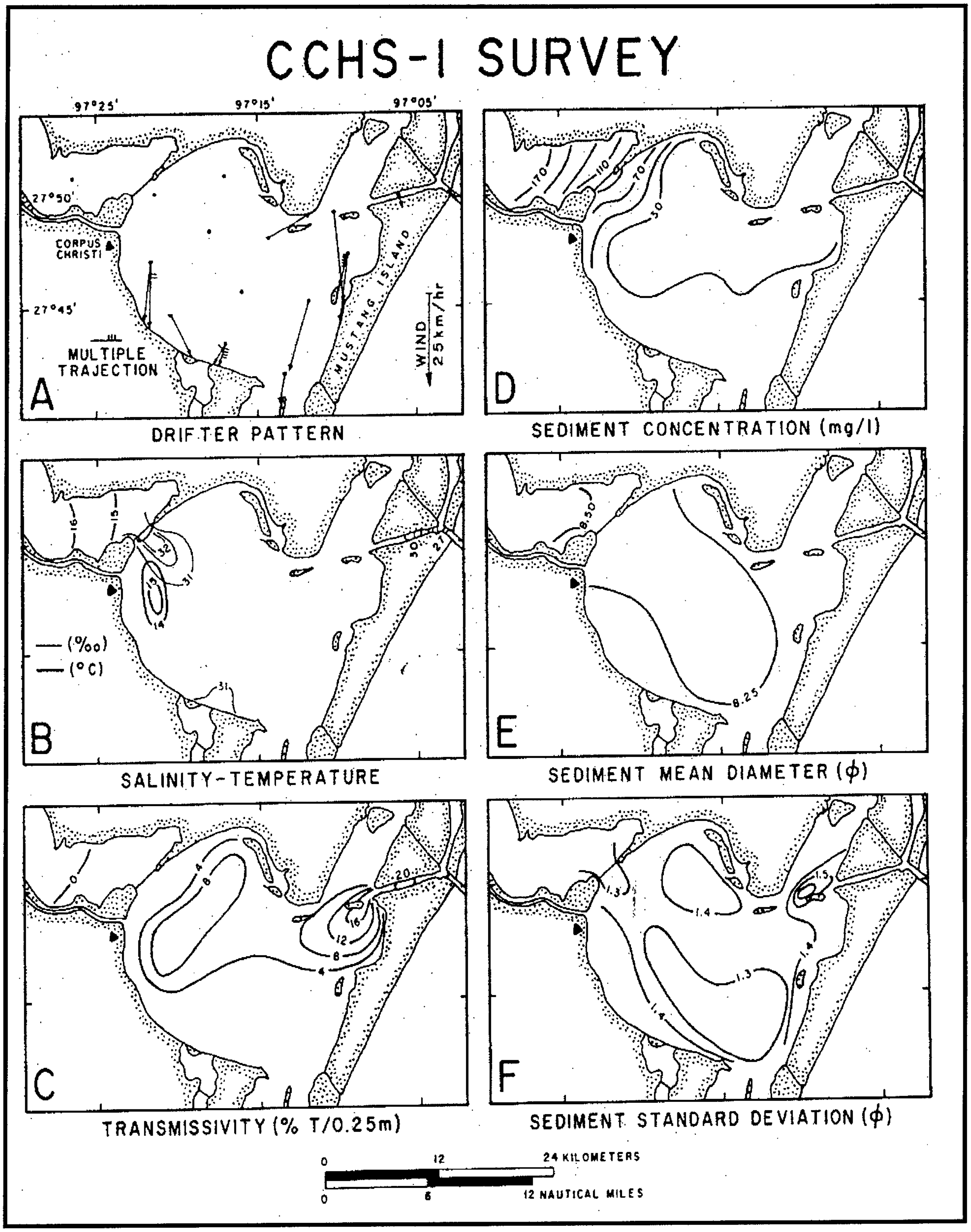

Figure III.16. Isopleths of hydrographic surface patterns in Corpus Christi Bay associated northerly winds. From Shideler (1984).

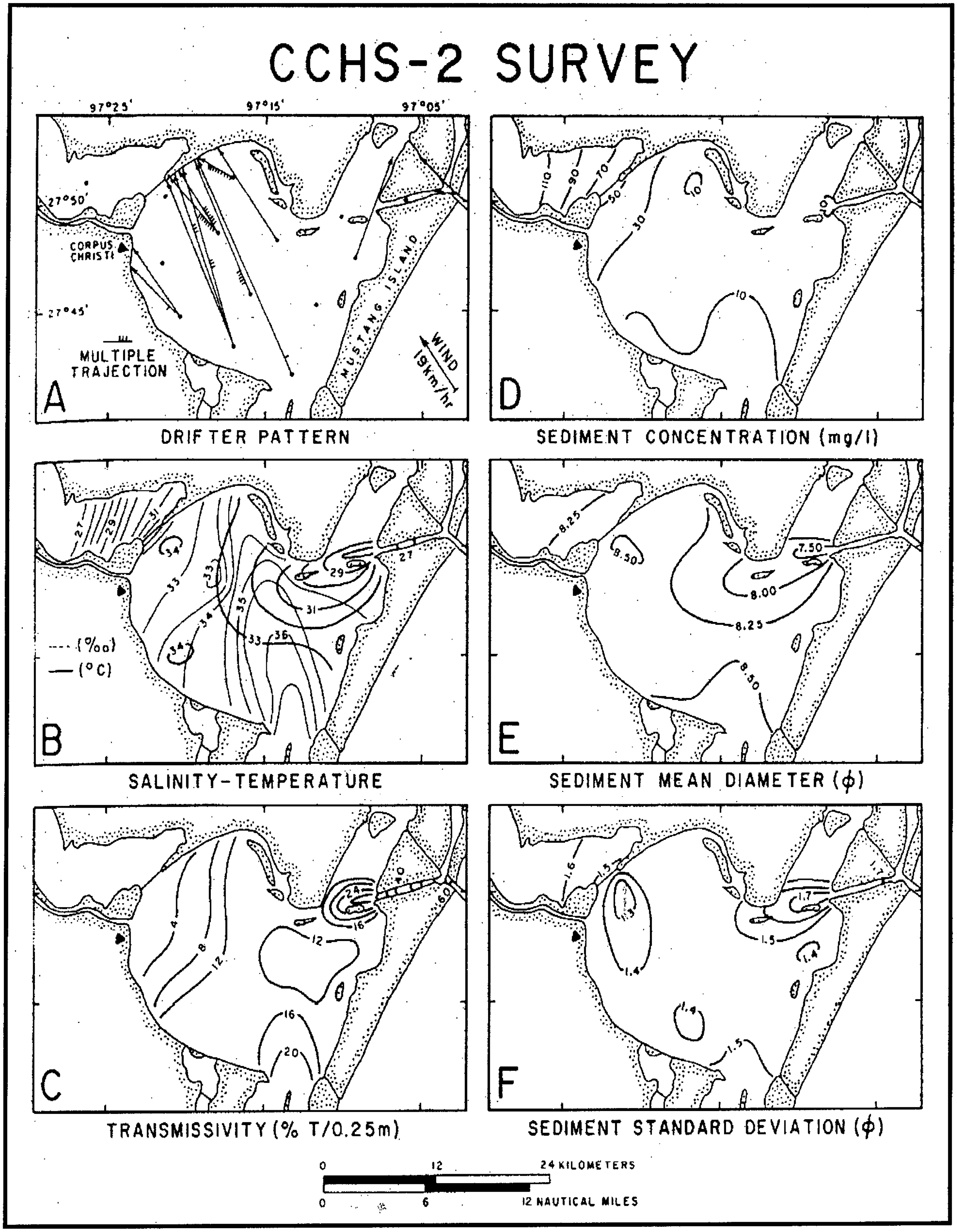

Figure III.17. Isopleths of surface hydrographic patterns in Corpus Christi Bay associate with southeasterly winds. From Shideler (1984).

Case 1. Survey CCHS-1 was conducted during passage of a front with strong north wind (Figure III.16). Transmissivity was lowest within Nueces Bay (0 %T 0.25 m), and highest toward Aransas Pass (20 % T 0.25 m). Patterns in transmissivity and suspended sediments were attributed to sediment input from Nueces Bay into Corpus Christi Bay. There was also increased wave action, causing higher turbidity, along the southern margin of Corpus Christi Bay and dilution of turbid waters by Gulf of Mexico water entering Aransas Pass. These conditions occur only during periods of strong north and northeasterly winds.

Case 2. Survey CCHS-2 was completed during conditions of prevailing southeasterly wind; at a speed of 80 km h^{-1} (11.4 mph) (Figure III.17). Highest suspended sediment concentrations, 111 mg l^{-1} (ppm) were found within Nueces Bay. Turbidity structure of Corpus Christi Bay under southeasterly winds was attributed to input of less turbid waters from Upper Laguna Madre to Corpus Christi Bay. There was also higher wave action at the bay-head, confinement of turbid water within Nueces Bay and dilution of turbidity by Gulf of Mexico water entering Aransas Pass.

Seagrasses play a critical role in determining depositional and sediment characteristics of estuarine systems. Seagrasses interact with the physical environment, effectively altering local depositional and substrate characteristics of sediments. Leaves baffle wave and current action, leading to deposition of fine-grained sediment within meadows (Ginsburg and Lowenstam, 1956; Scoffin, 1970; Fonseca et al., 1982). Therefore, seagrass meadows are important in preventing and mitigating coastal erosion. In addition, root and rhizome systems stabilize sediment within seagrass beds (Ginsburg and Lowenstam, 1956). Loss of seagrass areal coverage can lead to higher overall estuarine turbidity.

B.2.2.2. Sedimentation

Sedimentation occurs when suspended material is deposited onto the bottom of an estuary. Sedimentation rates are related to many factors. Riverine input is a key sediment source in estuarine environments, and varies both seasonally and annually. This is reflected by suspended sediment concentrations in Corpus Christi Bay , which range from 11 to 52 mg l^{-1} (ppm) (Sideler and Stelting, 1981). Generally, more freshwater input equates to greater sediment deposition; however, river sediment loads also vary. Diversion channels and dams restrict flow of water through its natural course, and often leads to deposition of sediment before it reaches the estuary. Before damming of the lower Nueces River in 1930, sedimentation rate in Corpus Christi Bay was 0.48 cm y^{-1} (1.56 ft 100 y^{-1}). This rate was comparable to that of Aransas Bay, which was 0.43 cm y^{-1} (1.42 ft 100 y^{-1}) (Shepard, 1953). Until that time, Nueces River was the primary source of sediment for Nueces and Corpus Christi Bays. Since damming, this sediment source is no longer thought to be important (Shepard, 1953).

Once sediment reaches the estuary, dispersal is wind driven (Sideler and Stelting, 1981). Resuspension of sediment can also be caused by erosion of shoreline and channel margins. Such erosion may be caused by boat and ship wakes, heavy flooding, or from wind-generated waves (Onuf, 1994). Erosion is a source of net sediment accumulation in the estuary, but it (as well as riverine input) may be offset by subsidence, which is discussed in section B.2.3. Factors responsible for erosion are also responsible for resuspension of sediment. Sediment resuspension in shallow waters can be initiated at current speeds as low as 10 cm s^{-1} (4 in s^{-1}) (de Jonge and van den Bergs, 1987). Although resuspended sediment is omitted from calculation of sedimentation rates, it presents the same problem for benthic organisms in settlement of suspended material and increased turbidity. Turbidity as a disturbance is discussed in section B.2.2.1.

Little information exists regarding how estuarine organisms are affected by sediment load in the water column (Sherk and Cronin, 1970). The primary means by which silt alters benthic habitat is by blanketing the sediment surface (Ellis, 1936). At high sedimentation rates, animals not able to avoid burial may be smothered. For example, silt layers between 1 - 25 mm (0.04 - 1 in) thick killed 90% of freshwater mussels on the bottom (Ellis, 1936). Influence of sediment is species specific, related to an organism's size and feeding type, and sedimentation rate. Some animals, such as eastern oyster, *Crassostrea virginica*, are remarkably silt tolerant provided they are not completely buried (Sherk and Cronin, 1970).

Organism feeding mode is one key process affected by particulates in water. Many benthic species are known to be facultative suspension feeders (Taghon et al., 1980). Facultative suspension feeders are deposit feeders that switch to suspension feeder based on current speed or particulate loadings. However, some organisms cannot tolerate suspended particulates; this is known as trophic amensalim (Rhoads, 1974). For example, there is usually a very low density of infaunal suspension or filter feeders in environments with high rates of silt deposition. In contrast, depositing feeding organisms generally have high densities in depositional environments with high suspended sediment concentration (Brehmer, 1965; Rhoads, 1973; 1974). Some species are better able to cope with higher rates of sedimentation, thus changes in sedimentation rate may shift the balance of competition. In contrast, wind-driven turbulence can resuspend microphytobenthos, fecal material, and detritus, which are then available for suspension feeders and can induce high feeding rates (Taghon et al., 1980). However, it has been found sediment resuspension does not alleviate limitations to mussel growth arising from seston depletion (Fréchette and Grant, 1991).

B.2.3. Subsidence

Land surface subsidence is lowering of sediment surface relative to, and accounting for, changes in eustatic sea level. Subsidence can be caused by a number of factors including: 1) natural processes such as compaction and faulting, 2) withdrawal of shallow subsurface fluids, 3) shallow sulfur and salt mining, and 4) possibly extraction of deep oil and gas reserves (Lofgren, 1977, Paine, 1993). Withdrawal of groundwater, and associated compaction of water bearing sediments, is associated with highest rates of subsidence (Paine, 1993). However, lower subsidence rates in regions with little ground water withdrawal, may be related to extraction of oil and gas resources. Subsidence rates vary temporally and spatially, probably due to differential rates of subsurface fluid withdrawal and natural causes of subsidence (compaction and faulting) (Swanson and Thurlow, 1973). It may also be related to seasonal variation in barometric pressure, wind, and river discharges.

It could be argued that subsidence in Texas is not a natural phenomenon, but one caused by human activities associated with water use, and oil and gas extraction. Long term subsidence rates (on a geologic scale of 10^5 years) in Copano Bay have been estimated at < 0.05 mm y^{-1} (0.002 in y^{-1}). In 1982, the Texas Department of Water Resources determined subsidence in Region 4 (Refugio, Aransas, San Patricio, Nueces, Kleberg, and Jim Wells Counties) was typically < 15 cm (0.6 in) for two different 33 year periods (1918-1951 and 1942-1975). This translates to approximately 0.45 mm y^{-1} (0.02 in y^{-1}) subsidence, almost 10 times greater than the more long term estimate derived from Copano Bay. Two areas in Region 4, one of which is in Corpus Christi, were found to have subsidence rates > 15 cm (0.6 in). However, more recent estimates of subsidence rates vary spatially, ranging from 1 - 22 mm y^{-1} (0.04 - 0.9 in y^{-1}) along the entire Texas coast (Paine, 1993). The greatest amount of land surface subsidence in Corpus Christi, Texas was 1.61 m (5.3 ft) between 1942-1975 for a rate of 4.33 mm y^{-1} (0.17 in y^{-1}). This subsidence occurred in the western part of the city where ground water extraction was not large enough to be the primary cause (Ratzlaff, 1982). Because the subsided region corresponds with the Saxtet Oil and Gas Field; subsidence was caused by withdrawal of oil, gas, and related fluids (Ratzlaff, 1982). Conversely, subsidence in the region of Texas between Sinton and Harlingen has been relatively stable (Paine, 1993). In this region, rates of surface motion tend to be 2 - 4 mm y^{-1} (0.08 - 0.16 in y^{-1}) upward. However, between Sinton and Port Aransas, rates of surface movement varied between 5 mm y^{-1} (0.2 in y^{-1}) downward to 10 mm y^{-1} (0.4 in y^{-1}) upward.

There is no evidence of significant subsidence in estuarine areas of the CCBNEP study area. Natural subsidence occurs at rates < 0.05 mm y^{-1} (0.002 in y^{-1}) and may be offset by influx of sediment with freshwater inflow. However, with increased water diversion, less sediment is deposited in estuaries to counter balance subsidence. Intertidal habitats are at risk because of subsidence. If elevation of sediment surface changes with respect to sea level, zonation of intertidal benthic habitats will change. Areal extent of emergent marshes may increase or decrease, as well as areal extent of submerged seagrass meadows. Unfortunately, it is not possible to predict influence of subsidence and deposition processes that vary spatially and temporally by substantial amounts.

B.2.4. Hypoxia

Hypoxia (low oxygen conditions) is defined as dissolved oxygen (DO) levels < 2 mg l^{-1} (ppm). Hypoxia is considered a disturbance because few animals can withstand low oxygen levels for extended periods. Animals able to survive such harsh conditions tend to be opportunistic species such as the polychaete *Streblospio benedicti*. However, even this species cannot survive for long in low oxygen environments. In laboratory experiments, it was found that *S. benedicti* can survive for more than two weeks at dissolved oxygen levels of 14.5 % and 7% of air saturation at 26 C (79 F) (Llansó, 1991). Under anoxic (zero DO) conditions *S. benedicti* died within 55 hours.

One surprising finding from studies described in section B.2. is that each summer, hypoxia events were noted in bottom water at one site in a portion of Corpus Christi Bay near Shamrock Island (Figure III.18). This site is Station D in Montagna and Kalke (1992). Except for 1993, when salinity levels were relatively low and the water column well-mixed, hypoxia occurred every year at this station. Hypoxia occurs only in summer, when temperatures and salinities are high (Figure III.18). Several processes are thought to be related to onset of hypoxia including water column stratification and organic matter decomposition (Officer et al., 1984; Pokryfki and Randall, 1987). The most likely explanation for hypoxia in Corpus Christi Bay is water column stratification. This is surprising because the estuary is shallow, windy, and thought to be well-mixed.

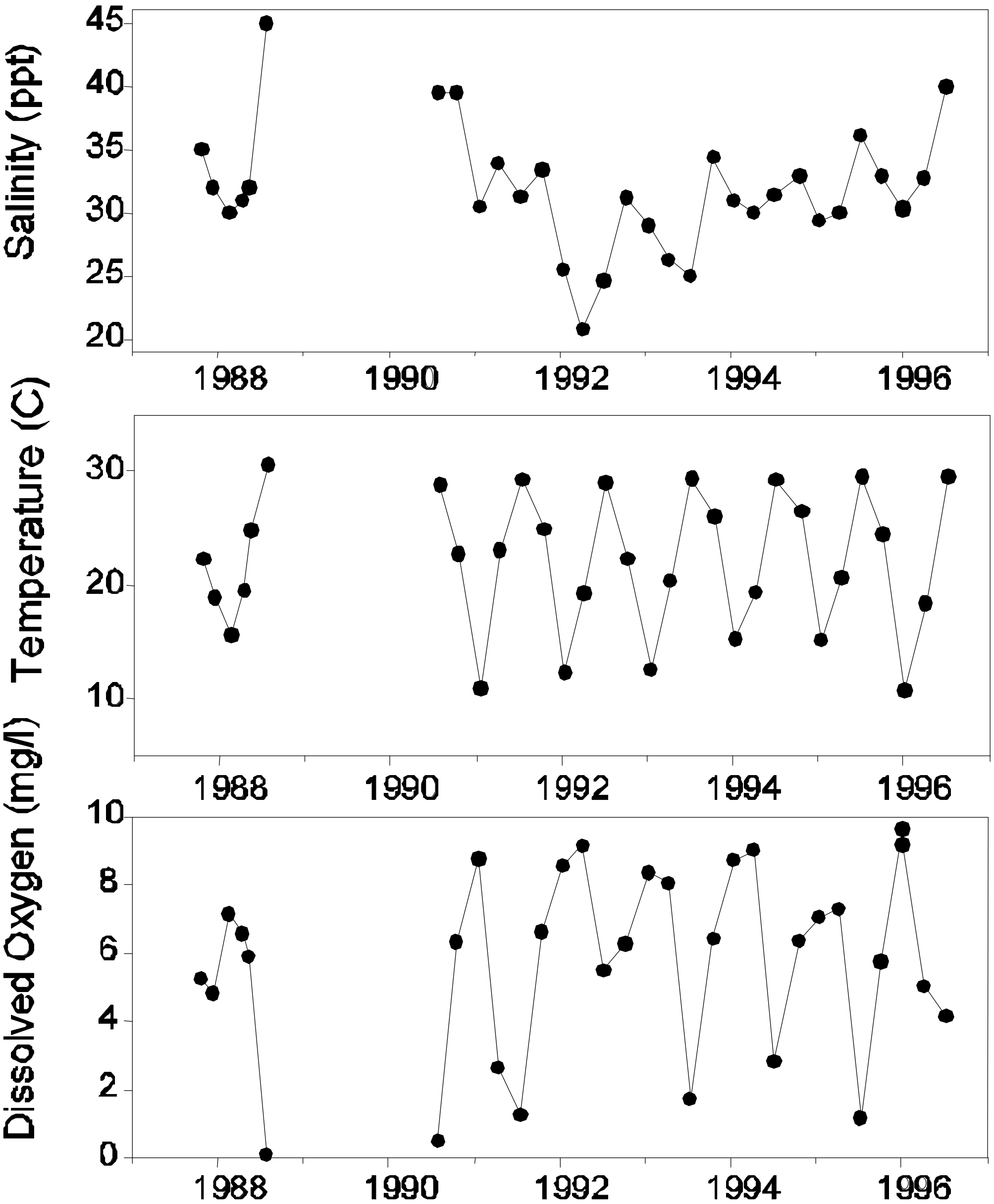

Figure III.18. Long-term hydrographic conditions in Corpus Christi Bay, Station D, near Shamrock Island. From Montagna, unpublished data.

Ritter and Montagna (1997) sampled the area in Corpus Christi Bay near Shamrock Island every three weeks between 3 May and 26 August, 1994, to determine temporal extent of hypoxia. As in previous studies, hypoxia was associated with water column stratification. Hypoxia was present between 14 June and 5 July. The hypoxic area extended south for about 2 miles, but not in any other direction. Sampling was also performed through the month of July, 1996, to determine spatial extent of the hypoxic area, hydrographic characteristics associated with hypoxia, and benthic effects. Ten stations were sampled, five in the hypoxic area and five outside the hypoxic area (Table III.13).

Table III.13. Effects of benthic hypoxia in Corpus Christi Bay in July 1996. Average for five stations in each area.

Area	Biomass ($g\ m^{-2}$)	Density ($n\ m^{-2}$)	Diversity (no. species)	Diversity (Hill's N1)	D. O. ($mg\ l^{-1}$)
Control	4.376	11,875	13	7.0	4.4
Hypoxia	0.343	2,553	1.5	4.2	2.1

There was twice as much oxygen in the control than in the hypoxic areas sampled (Table III.12). This had a major effect on productivity in those sediments, as indicated by a biomass standing stock that was 14 times greater in the normoxic (> 3.0 mg l^{-1}) (ppm) sediments. There was a concomitant five-fold decrease in density and species number in the hypoxic sediments.

Hypoxia was due to water column stratification and resulting large differences in surface and bottom water salinity (Figure III.19). Temperature was constant throughout the water column at both stations, so differences in water masses were driven by influx of salty water of same temperature. The halocline (where salinity changes abruptly) was at 2 m below the surface at both stations (Figure III.19). Salinity was similar at the surface and constant throughout surface water mass at both stations. In the lower half of the water column, salinity decreased by about 2 ppt in the normoxic station, but increased by about 5 ppt at the hypoxic station. Oxygen concentrations decreased continuously at the normoxic station from 5.6 mg l^{-1} to 3.6 mg l^{-1}. In contrast, at the hypoxic station, oxygen was constant at 5.3 mg l^{-1} above the halocline, but decreased to 1.9 mg l^{-1} below the halocline. Differences in water mass structure indicate a layer of hypersaline bottom water was causing stratification, and oxygen was depleted from bottom water mass, which was not mixing with the surface.

Areal extent of the hypoxic zone was relatively small. The area with hypoxia roughly formed a triangle with three points connecting Oso Bay inlet to Corpus Christi Bay, Shamrock Island, and the southeastern-most point of Corpus Christi Bay. There was a gradient of increasing salinity from Oso Bay to Corpus Christi Ship Channel and Port Aransas. The distribution of hypersaline bottom water indicates it could be derived from hypersaline Laguna Madre from the ICWW or pumped into Oso Bay by the CPL Barney Davis power plant. Another possibility is simply the combination of evaporation and lack of water movement in the area, and sinking of dense saltier water to the bottom.

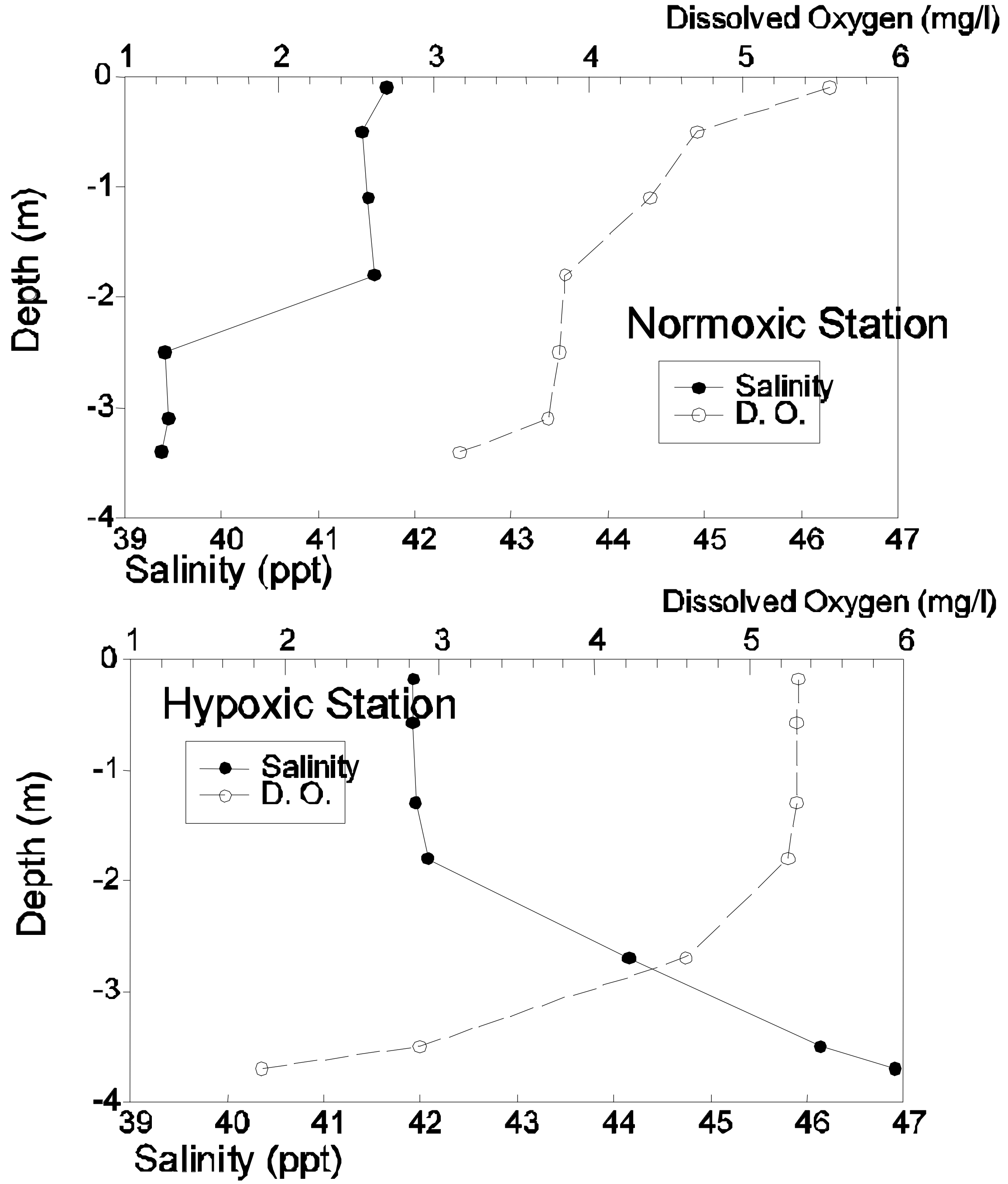

Figure III.19. Stratification at normoxic and hypoxic stations in Corpus Christi Bay. From Ritter and Montagna (submitted msc.).

The hypoxic area is also characterized by the lowest current flow vectors in all of Corpus Christi Bay (personal communication: Cheryl Brown, Texas A&M University; Junji Matsomoto, Texas Water Development Board). The hypoxia demonstrates the flaw in the dogma that open bays in the CCBNEP study area are well-mixed because of high winds and shallow depths. Hypoxic events in Corpus Christi Bay appeared to be due to altered circulation patterns in the bay and were limited to a short duration during summers. Hypoxic events also have biological consequences because benthic biomass, abundance, and diversity is degraded (Table III.13).

B.2.5. Brown tide

The effect of brown tide is most obvious in Upper Laguna Madre and Baffin Bay. Prior to the brown tide bloom in 1990, benthic biomass, abundance, and diversity were relatively high in Upper Laguna Madre, but have decreased since then (Figure III.14). This is especially true for biomass, which has declined almost four-fold. This represents an enormous loss of secondary production in the ecosystem even though primary production increase 10 fold. In Baffin Bay, biomass declined to near zero levels for the first two years following the brown tide bloom, but has had seasonal recovery since 1992. However, highest biomass values were recorded prior to brown tide.

Benthic infaunal abundance values in Baffin Bay have not recovered to pre-brown tide levels as of 1997. In contrast, abundance appeared to be incrementally recovering every year in Laguna Madre, but it had not fully recovered in 1995. Biodiversity is still low in Laguna Madre, but has recovered in Baffin Bay. Diversity in Baffin Bay is often near 1, because the system is dominated by one opportunist species, the polychaete *Streblospio benedicti*. The bivalve, *Mulinia lateralis*, can apparently eat and assimilate brown tide, but populations have been virtually absent since the brown tide bloom (Montagna et al., 1993).

B.2.6. Bioturbation

Bioturbation is alteration of substrate through burrowing and feeding activities of benthic organisms such as fish, crabs, polychaetes, and molluscs (Lalli and Parsons, 1993). In many cases, animal activity can continually rework sediment and disrupt progression of community succession by affecting presence and recruitment of other species or trophic groups. Bioturbation affects benthic communities primarily by altering sediment stability and influencing settlement and recruitment of organisms.

Bioturbated sediment is typically less stable and more easily resuspended than sediment that has not been altered by burrowing organisms (Rhoads, 1973; 1974). Presence of polychaete tubes can lead up to a 46% decrease of critical erosion velocity around tubes (Luckenbach, 1986). Other factors that may work in conjunction with bioturbation to affect sediment stability include: topography, sediment size, composition of sediment, and sediment binding (Luckenbach, 1986). Bioturbation of the top few centimeters of sediment by deposit feeders creates a soft layer rich in fecal material that is easily resuspended by low velocity currents (Rhoads and Young, 1970). Instability of the fecal layer created by deposit feeders may influence presence of other trophic groups (Rhoads and Young, 1970). For example, resuspended fecal material and sediment may clog filtering appendages of suspension feeders, bury juveniles and recently settled larvae, discourage settlement of suspension feeding larvae, and inhibit attachment of sessile epifauna (Rhoads and Young, 1970).

Sediments around burrows and polychaete tubes tend to have a greater density of animals (DePatra and Levin, 1989; Luckenbach, 1987). For example, *Streblospio benedicti* and *Nereis succinea* are common around *Diopatra* spp. tubes, but fewer in sediments lacking such tubes (Luckenbach, 1987). However, both species have been found to reduce survival rate of *Mulinia lateralis*, which is inhibited from recruiting around *Diopatra* spp. tubes due to high post-settlement mortality. Fiddler crab burrows may influence hydrography such that meiofauna are more easily deposited or trapped in burrows (DePatra and Levin, 1989). In addition, some species (e.g., harpacticoid copepod *Zausodes* spp.), may be attracted to fecal mounds (e.g., those made by an Enteropneust) to avoid predation or to seek food or a preferred sediment type (Varon and Thistle, 1988).

Though very small, meiofauna are capable of significant sediment reworking. Macrofauna tracks, trails, and burrows disappeared after macrofauna (but not meiofauna) removal when left in an undisturbed aquarium (Cullen, 1973). After only two days, evidence of previous macrofauna activity began to blur. All macrofaunal evidence disappeared within two weeks and was attributed to meiofaunal activity. Several animals are thought to be responsible. Ostracods (about 0.5 mm long) burrow up to 4 mm down and are capable of moving sediment several times their own size with jerky and vigorous motions. Copepods and malacostracans exhibited similar behavior but their ability to move sediment varied with size and vigor. Nematodes create a fine network of burrows that are thought to be reinforced with mucous. These burrows may persist a few hours until disturbed by other biogenic activity. Meiofauna activity is also thought to play an important role in development of an oxidized layer in the upper 0.5 - 1.5 cm (0.2 - 0.6 in) of sediment

Head-down deposit feeders, characteristic of highly bioturbated areas, are not commonly found in the CCBNEP study area (Montagna and Kalke, 1992). However, several other bioturbators occur in the region. Examples include crabs, stomatopods, enteropneusts, skates, rays, and ophiuroids. Presence of ophiuroids in the estuary is unique to the CCBNEP area. Stone crabs (*Menippe adina*) are known to excavate burrows in mud, and hide in crevices of oyster reefs and jetties (Britton and Morton, 1989). Juvenile stone crabs use crevices in mud banks as refuges (Britton and Morton, 1989). Stomatopoda (mantis shrimp) are specialized predators for benthic macrofauna (Ruppert and Barnes, 1994). Many stomatopods live in burrows excavated during feeding activities that may be as long as 4 meters vertically (Ruppert and Barnes, 1994). Ophiuroida (basket, serpent, and brittle stars) are the most mobile of the echinoderms and may grow as large as 12 cm (4.7 in) in diameter (Ruppert and Barnes, 1994). Ophiuroids of the family Amphiuridae dig channels through the sediment using undulatory motion of arms and digging with tube feet. (Ruppert and Barnes, 1994). Numerous demersal fish are known to create sediment disturbances via swimming, feeding and resting activities. Examples include stingrays (Dasyatidae) and skates (Rajidae) (Britton and Morton, 1989).

Bioturbation is a natural ecological process. Although it may affect distribution of some species, it should not be considered a disturbance *per se*. However, when considering how human activities may influence benthic communities, bioturbation should be taken into account.

Table of Contents

C. Key Resource Management Concerns

Key resource management concerns of the CCBNEP-Scientific Technical Advisory Committee (STAC) were identified by interviewing members and ex-officio members of that committee. A total of 32 members responded to questions while five declined to be interviewed. Questions asked during the interview are included in Appendix A and correspond with the following subsections.

C.1. Resource issues

Figure III.20 provides a diagrammatic representation of the number of interviewees expressing concern, or lack of concern, about resource management issues. It is important to note that a single interviewee could express concerns about a number of different issues. Thus, total number of responses exceeds the number of people interviewed.

The top four resource management issues based on this survey are: dredging, pollution, freshwater inflow, and trawling. Eleven or more interviewees expressed concerns regarding these key issues. Although, dredging was ranked first in order of concern, only 40% of people interviewed expressed concern about effects of dredging activities (e.g., excavation and deposition). One person expressed the opinion that dredging was not an issue in the CCBNEP study area.

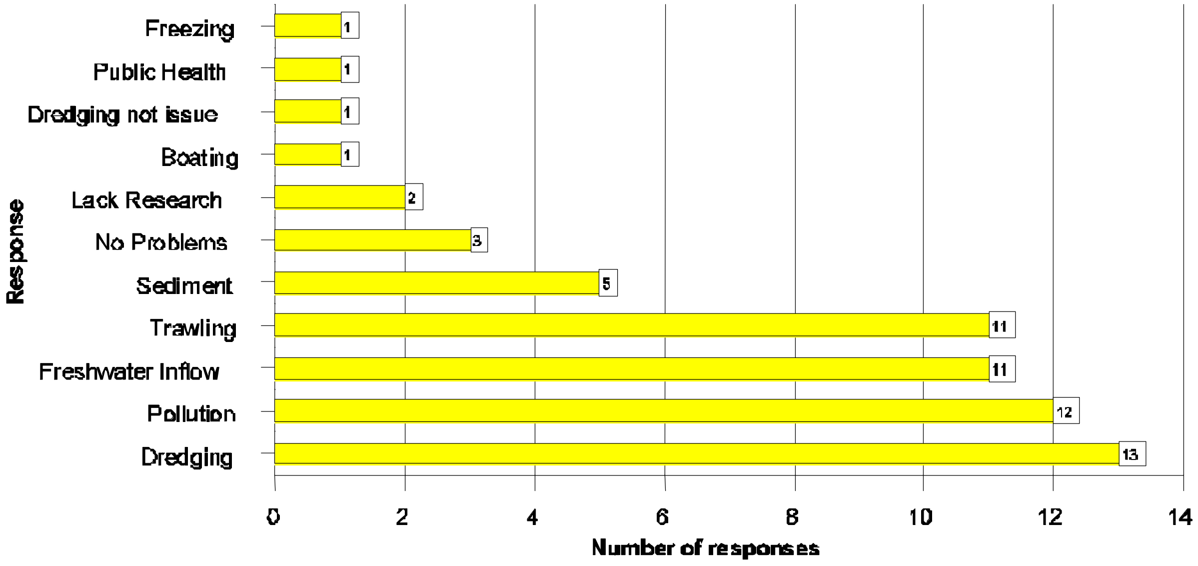

Figure III.20. Number of people interviewed who expressed concern, or lack of concern, about a resource management issue.

About 38% of interviewees expressed concerns about pollution in the CCBNEP study area. Concerns were expressed regarding discharge of contaminants and domestic wastes into bays, non-point source pollution (e.g., fertilizers and insecticides), and brine discharges. One person expressed the concern that there is not enough water quality testing.

Freshwater inflow and trawling each were expressed as resource concerns by 31% of people interviewed. Concerns regarding freshwater inflow arise from effects of fluctuating salinity on benthic animals which is also related to rainfall, evaporation, run-off, and mixing with ocean water. Trawling concerns arise from physical disturbance it creates as well as the removal of organisms thought to be important links in the food web.

About 14% of the people interviewed expressed concerns related to sediment (e.g., suspended sediment, turbidity, and siltation). Other concerns include: lack of research or data (6%), effects of recreational boating on benthos (3%), public health (3%), and effects of natural freezes (3%). It is interesting to note that three respondents (9%) said they had no concerns or their concerns had already been addressed.

C.2. Social and economic concerns

Responses to the question on social and economic concerns were not clear enough to yield demonstrable results. Several interviewees said they did not know enough about social and economic issues to comment. Others replied with management recommendations or concerns about how human activities affect benthos. However, some responses were particularly enlightening with regard to economic conflicts that arise through competing uses of coastal resources. Rresources of the CCBNEP study area are economically valuable from a number of perspectives. Conflicts may arise due to competing pressures often placed on these resources. The following are examples of potential conflicts in the CCBNEP study area.

- Competing uses of freshwater include human consumption (including watering lawns), industrial uses, and recreational and commercial fishery productivity (which is affected by freshwater releases to the bay).

- Maintenance of the GIWW often leads to disposal of dredge material in the open bay, which may be detrimental to benthic resources (e.g., seagrasses, benthic animals) and affect commercial and recreational fisheries. However, dredge material can be used to enhance the environment by creation of new wetlands and rookery islands.

- Release of municipal and industrial effluent into bays creates conflict between needs of industries and municipalities to dispose of unwanted effluent and needs of the tourism industry to have an aesthetically pleasing environment. Discharge of effluent may also conflict with needs of recreational and commercial fishing industries if effluent contaminates and degrades the environment.

- Shoreline modification may bring private property rights and human economic needs to develop property to its highest and best use into conflict with needs of the fishing industry to have wetlands and seagrass beds as nursery areas and critical habitats for commercially and recreationally important finfish and shellfish resources.

- Economic need to produce, transport and refine petroleum may lead to conflicts with tourism and fishing industries if contaminated produced water is discharged into bays or if oil or other petroleum product is spilled.

For more information regarding social and economic status of the CCBNEP study area, refer to the human resources report prepared by Jones et al. (1997). For a more general background in social and economic issues related to coastal management, the reader is referred to Edwards (1987) and Ditton et al. (1977). Another resource is Schmidheiny (1992), which provides a global business perspective on development and environment by focusing on sustainable development.

C.3. Management recommendations

Specific management recommendations were elicited from people interviewed (Figure III.21). Respondents could offer more than one recommendation. Thus, total number of responses exceeds the number of respondents.

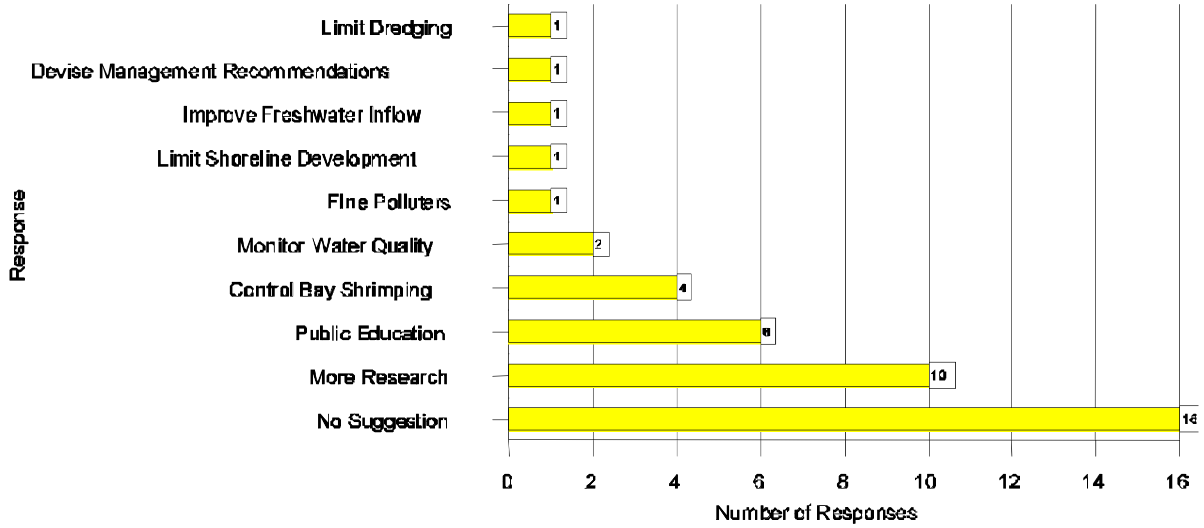

Figure III.21. Number of people interviewed who provided a particular resource recommendation.

Suggestions with greatest number of responses were: more research (31%), more public education (19%), and reduced (or controlled) bay shrimping (13%). About 6% of people interviewed recommended continued monitoring of water quality. Only 3% (1 respondent for each response) suggested imposing fines on polluters, limiting shoreline development, improving freshwater inflow, limiting dredging, devising management guidelines for turbidity and salinity, and allocating more research funding. It is important to note that 50% of people interviewed made no suggestions for management actions.

Numerous research recommendations were made. In general, these suggestions included greater funding for long-term studies, collection of consistent data, and faster, more efficient, collection of information. Other suggestions include gauging freshwater inflow into Baffin Bay, determining effects of trawling, assessing societal costs of environmental degradation, and investigating indirect economic effects of resource management activities.

Most respondents expressed a need for more research (Fig. III.19). The following questions represent the highest priority research issues with respect to management of resources in the CCBNEP study area.

- How does removal of organisms due to trawling affect food webs?
- What factors are responsible for maintaining brown tide?
- How much freshwater is required to maintain sustainable yields of natural resources?
- What are quantitative benthic effects of the input of ground water from septic tanks?
- How does brown tide influence community dynamics?

Table of Contents

IV. DISCUSSION

One of the most challenging undertakings in applied ecology is to detect effects of anthropogenic disturbances. It is difficult because natural variation is usually high in almost all variables. Many times mean values are not as interesting as variance of variables (Green and Montagna, 1996). This is because the nature of disturbance is to change the range of responses, therefore increasing variance of a response. When statistically analyzing disturbances, there are two potential problems: failing to detect differences that exist, and overlooking differences that occur. Failure to detect a difference that exist, called the Type II error, is often due to a lack of power in a study design. When this occurs, natural variability is so large that means for a disturbed site and an undisturbed site are not statistically significantly different. Failure to detect real differences occurs only when means, and not variances, are compared. The goal of the following discussion is to synthesize information on relative contributions of different disturbances, to compare disturbance effects in different habitats, and to examine effects of anthropogenic disturbances on vegetated and unvegetated habitats of estuaries of the CCBNEP study area.

A. Relative Contribution of Anthropogenic and Natural Disturbances

Natural disturbances identified and reported include: storms, floods, droughts, sediment resuspension, sediment deposition, and bioturbation. Anthropogenic disturbances identified and reported on include: shrimp trawling, dredging, commercial marine operations, marine construction, and the chemical industry. Some disturbances do not fit easily into the two categories, but have characteristics of both natural and anthropogenic disturbances.

Three disturbances that affect benthos are difficult to classify as either natural or anthropogenic disturbances: reduced freshwater inflows, altered circulation patterns, and brown tide bloom. For each of these disturbances, there is a strong natural component, but it is highly likely human activities exacerbate the degree of disturbance. The CCBNEP study area is in a semi-arid climate zone, but man has further reduced natural inflow rates. The region has a natural microtidal range with high residence times, yet channelization has further restricted, or redirected, current flow. Algal blooms often result from eutrophication due to man’s activities, but it is unknown if this is the case for brown or red tides. Even when it is clear which disturbances are natural and which are due to anthropogenic influence, it is not easy to determine relative contribution of each.

Freshwater inflows are subject to extreme year-to-year natural variability in the CCBNEP study area. The southern part of the CCBNEP study area is typically water deficient and often hypersaline due to climatology. During droughts, hypersalinity is common throughout the area. These observations lead to speculation that bays in the area contain organisms and communities adapted to high and variable salinity. A consequence of this hypothesis is the suggestion that inflow reduction is unimportant, because organisms are already adapted to drought disturbance, which is natural. However, we have been unable to uncover any studies that support this hypothesis. Furthermore, ecological theory (Section I.C.) and our own research (Section III.B.2) do not support the hypothesis that there is a “desert estuarine” community. Instead, theory and data indicates communities respond to disturbance on any scale one event at a time. Because theory (Figure I.3) and empirical data (Table III.11; Figures III.11-12) indicate reduced freshwater inflow leads to community change and reduced productivity, long-term reduced inflows due to reservoir construction are most likely an anthropogenic disturbance. Because estuaries of the CCBNEP study area are already stressed by naturally low rates of inflow, the estuaries should be considered especially sensitive to anthropogenic reductions of inflow rates. The correct analogy to deserts is: arid environments are fragile and can take long time periods to recover from disturbance.

A secondary inflow issue is the dam itself. Dams reduce sediment loadings to bays that are undergoing subsidence due to water and hydrocarbon extraction. This results in vegetated habitat loss due to sediment starvation and deepening of bay bottoms. Dams also alter freshwater habitats critical to some species.

Hypoxia in Corpus Christi Bay results from water column stratification, which leads to oxygen depletion in bottom water (Figure III.18). Oxygen depletion is a natural result of high temperatures, productivity, and hence high respiration rates of benthos. However, it is not certain stratification is natural or caused by anthropogenically altered circulation patterns. Hypoxia is confined to a portion (perhaps 25% or less) of the southeast part of the bay. Hypersaline bottom water appears to come from Oso Bay, and appears to be well-mixed in northern and western parts of Corpus Christi Bay. Isolation of hypoxic water indicates altered circulation probably plays an important role in both introduction of hypersaline water and confinement of water to that part of the bay. Because of the uncertainty, this is an information gap that must be be filled with physical hydrographic studies. Ironically, hypoxia may be related to low inflows as well, because hypoxia may not appear during years with heavy rainfall and low salinity (Figure III.18). Because of interaction between altered inflows and altered circulation, hypoxia is also probably exacerbated by anthropogenic disturbance.

Both reduced inflow and hypoxia lead to reductions of benthic productivity, so they are clearly serious disturbances. Low inflow rates are exacerbated by reduced releases and hypoxia is exacerbated by both low inflow and altered circulation. Therefore, a preliminary conclusion is both reduced inflow and hypoxia are mainly anthropogenic disturbances. In this regard, these two disturbances are similar to shrimp trawling, dredging and marine operations and construction.

In many places around the world, nuisance and harmful phytoplankton blooms are traceable to increased anthropogenic inputs of various compounds into coastal waters (Smayda and Shimizu, 1993). In some cases biocides cause a breakdown of grazer control allowing phytoplankton biomass to increase. In other cases, inputs of limiting nutrients cause increased phytoplankton growth, a process called nutrification, or eutrophication. However, there is no data to suggest appearance of brown tide correlates with increased nutrient inputs of terrestrial origin. Studies to determine relative importance of new (terrestrial) nutrients and old (recycled from bay sediments) nutrients in maintaining the persistence of the brown tide are currently underway. The brown tide bloom began with simultaneous occurrence of three events: grazer control disruption (probably due to hypersalinity), a nutrient pulse (due to a freeze-induced fish kill), and an unstable ecosystem (also probably due to hypersalinity) that was conducive to a “weed species” bloom (Buskey et al., 1997).

Although it appears that brown tide is a natural event, it is interesting to note low inflow rates and high salinity may have played a role in at least two of the three possible causes of the bloom. Once again, we see that rather than being separate, natural and anthropogenic influences are synergistic.

Shrimp trawling leads to three issues: physical disturbance of sea floor, higher turbidity, and bycatch. It appears about 80% (of weight) of shrimp trawl harvest in the CCBNEP study area is bycatch (Section III.A.1.1.7). Although direct loss of benthic species is minimal, there may be ecological effects due to alteration of predator densities, and alteration of the nitrogen balance in the ecosystem (Figure III.6). Predator or prey removal is a very serious scientific issue. Although we do not fully understand effects of the trophic cascade caused by predator or prey removal, it is clear that community structure in the bay is being altered. Bycatch affects nutrient recycling as well, because organisms removed no longer contribute to processing of material and flow of energy in the ecosystem. It has been suggested a 50% reduction of bycatch in the Gulf of Mexico would lead to substantial increases in shrimp populations (Browder, 1991). Finally, the only issue for which impacts have not been clearly defined is effect of sediment disturbance by trawling, even though the issue was first recognized as early as 1376 (Graham, 1955). However, based on studies outside the CCBNEP study area, we can expect this disturbance will lead to communities with fewer molluscs (Messieh et al., 1991), and reduced benthic biodiversity in general (Dayton et al., 1995).

Dredging and dredge-spoil disposal is another large-scale anthropogenic disturbance. Main effects on benthos are removal, smothering, and turbidity related (Section III.A.3). Apparently, benthos can recolonize dredge spoil rapidly (Rhoades et al., 1978). Turbidity has negative effects on suspension and filter-feeding benthos (Section III.A.5), but main effects appear to be on vegetated habitats (Table III.1). Seagrasses may be buried and die and increased turbidity decreases light, and thus decreases photosynthesis. Seagrass habitats up to 1.2 km away from dredging can be affected (Onuf, 1994). Elevation change of the sediment surface is also a more serious disturbance for vegetated than non-vegetated habitats.

Commercial and recreational boat traffic have different impacts, depending on vessel draft. Commercial traffic may increase turbidity and erosion, and decrease biodiversity of infaunal species along shoals near shipping lanes (Flint and Younk, 1983). Propeller scars from recreational boating over seagrass beds appear to be a problem where depths average < 1 m (3 ft). Long-term damage can occur due to propeller scarring (Phillips, 1960; Zieman, 1976; Durako et al., 1992). Extent of propeller damage in the CCBNEP study area appears to be greater than originally anticipated (Volume II). Losses of seagrass bed habitat will have concomitant effects on benthos, fish and wildlife that utilize the habitat.

In general, soft-bottom benthic communities in the CCBNEP study area appear to be in a state of constant disturbance. Evidence for this conclusion is: 1) communities are dominated by polychaetes and molluscs are rare, 2) dominant species with life history characteristics typical of pioneering, not climax species, and 3) diversity is generally low. Two natural processes are most likely responsible for this condition: low freshwater inflow and high turbidity. Low freshwater inflow rates combine with high evaporation rates to yield relatively high salinities, even hypersalinity. High turbidities are found in the area because of extensive fetch across large, shallow bays, which allows high average wind speeds to produce high rates of sediment resuspension. High turbidity limits seagrass distribution and negatively affects molluscs. Natural high turbidity is exacerbated by anthropogenic activities (e.g., dredging, trawling, and ship traffic). Anthropogenic disturbances of reduced inflows, dredging, and shrimp trawling exacerbate all natural stresses to the ecosystem. Effects of anthropogenic disturbances are multiplicative and synergistic with naturally occurring disturbances.

Multiple stressors affect the ecosystem. It is not possible, with the current state of knowledge, to partition the percent contribution of each individual stressor to the total stress on the environment. One problem is that most of the disturbances occur in different benthic habitats, and affect different ecological processes. In some cases, disturbances affect energy flow, and in others, it affects community structure. Therefore, trying to compare disturbances is like comparing apples and oranges. It is possible to contrast multiple stressors, but it must be done within the context of a single study and with a single currency. One possible approach is to create an ecosystem model. A good example of this approach is the study of shrimp bycatch effects (Browder, 1991). It is not possible to distinguish how much diversity loss or productivity decline is due to any individual stress alone. Instead, all studies report on cumulative effects of both natural and anthropogenic disturbances, whether or not this is explicitly stated by the authors.

B. Comparison of Anthropogenic and Natural Disturbances in Different Areas

In some instances, information exists that allows direct comparison of anthropogenic and natural disturbances. In these cases, the same ecological processes are affected and responses are measured in the same units. However, even in these cases, effects of various anthropogenic disturbances are restricted to areas where human activities occur. For any disturbance, the main cause and effect is site specific.

Dredging removes substrate, increases turbidity, and reduces light that seagrasses need for growth. Although benthic organisms are removed by dredging and smothered by dredge spoil, soft-bottom benthic communities recover within a year. Therefore, it appears dredging disturbs seagrasses more than soft-bottom benthos. Based on habitat distribution, dredging is potentially more of a problem in Upper Laguna Madre than in Corpus Christi Bay or other bays in the CCBNEP study area.

Shrimp trawling creates a constant mechanical disturbance to the bottom. Removal of non-target species from the ecosystem can alter food web structure and function. Shrimping is more or less restricted to open bays. Therefore, shrimp trawling affects soft bottom benthos more than vegetated habitats, so it is an issue in open bays where most shrimping occurs (e.g., Aransas Bay and Corpus Christi Bay).

Freshwater inflow reduces salinity and transports nutrients and organic matter to the estuary. Filter-feeding and suspension-feeding organisms are dependent on inflows. Therefore, reduced freshwater inflows primarily disturb secondary bays, where estuarine or brackish communities should be dominating (e.g., Nueces Bay or Copano Bay). Oyster harvest in South Texas is a fraction of harvest in more northern bays, because of naturally low inflow rates. However, as inflow rates are reduced, existing populations are further stressed. Tertiary bays of Baffin Bay are also negatively affected by reduced inflow, but this is natural. Hypersalinity exists in Baffin Bay and Upper Laguna Madre most of the time. It also exists in Nueces, Corpus Christi, Copano and Aransas Bays during droughts. Hypersalinity is a severe disturbance that affects the entire CCBNEP study area and results in reduced benthic biodiversity and productivity. Only Upper Laguna Madre appears to be resilient to hypersalinity resulting in the highest finfish landings for all bays in Texas. The relatively high benthic biodiversity and productivity there is due to the presence of extensive seagrass habitat. Low freshwater inflow rates are natural in the study area, but man’s activities has further reduced these inflows, and restricted circulation. The net effect is hypersalinity, even in marginally dry years.

Brown tide has altered food web structure and reduced light to the bottom. However, brown tide is found mostly in Baffin Bay and Upper Laguna Madre. Therefore, it is mainly a problem in Upper Laguna Madre because of seagrasses. Decline in areal extent of seagrass habitat in Upper Laguna Madre has already been detected. If this continues, conversion of seagrass to unvegetated bare bottom is likely and Upper Laguna Madre will resemble Baffin Bay. Baffin Bay has the lowest benthic biodiversity and productivity of any bay in the CCBNEP study area, primarily due to low freshwater inflow.

Hypoxia appears to be restricted to the southeastern portion of Corpus Christi Bay in summer (mid June to early August). Hypoxia affects benthos more than nekton, and seagrasses are not found in this area. Therefore, hypoxia is mainly a problem to benthos of the identified area.

Hydrocarbon exploration and production occurs almost everywhere in the CCBNEP study area. However, produced water discharges are concentrated in Nueces Bay. This is especially unfortunate because Nueces Bay is a valuable nursery habitat for many important species, which is already stressed with low freshwater inflow. Pipeline construction across marshes and seagrass beds are another special concern because these are valuable habitats. These effects appear to be constrained to Nueces Bay and Upper Laguna Madre. The Corpus Christi Ship Channel is a site of special concern because of the potential for transportation accidents. It is well recognized that spills in coastal estuarine habitats have high potential for ecological damage and long-term effects. Fortunately major accidents are rare and most incidences occur in the Corpus Christi Ship Channel, which is a man-made enclosed area where spills can be spotted and controlled rapidly. The high incidence of spills and kills in Corpus Christi Bay illustrate this potential.

Overall, it appears that estuaries in the CCBNEP study area are in a naturally stressful environment due to the semi-arid climate and low microtidal range. These two natural stressors insure that any of man’s activities that add to stress in the ecosystem have severe impacts. Stressors are also synergistic. The net effect of low freshwater inflow rates and restricted circulation is greater than the sum of its parts.

Table of Contents

V. DATA AND INFORMATION GAPS

Understanding environmental issues requires technical information. Environmental issues are complex because of the variety of information needed, natural variability, and interactions among multiple stressors. To understand complex environmental problems one needs information on the environment, species in question, and how to separate and independently assess each stressor. This is almost impossible to do with just field studies or monitoring alone. Experimental studies are also required. In general, there is a need for explicit, quantitative understanding of the effects of most disturbances.

Population levels of desirable species fluctuate. To assess the importance of disturbance in causing fluctuations, it is not possible to rely on just the number of organisms censussed. The population density indicates the net number of organisms. Individuals are gained and lost from a population due to many different processes. Information about population dynamics, e.g., age specific birth rates, age-specific survival rates (which includes natural death, predation, and losses due to other environmental issues), immigration and emigration rates, and population size are also needed. Experimental exposures to disturbances are also needed to prove that results from field monitoring studies are more than correlative. Finally, homogeneity does not exist in any population in spatially heterogeneous environments. In the CCBNEP study area we typically lack detailed data on life history, food web structure and function, and benthic-pelagic coupling for many important species. This kind of data goes beyond the scope of routine monitoring studies.

The above example demonstrates that environmental problems are enormously complex, don’t lend themselves to simple reductionist experimentation, and exist within the context of natural background variability in space and time. Anthropogenic disturbances occur concurrently with other anthropogenic and natural disturbances. We have virtually no information from elsewhere, as well as from within the CCBNEP study area, on the mechanisms, interactions, or synergistic responses among multiple stressors. Studies are often performed on a single issue and assume other effects can be ignored. There is a need for environmental studies at the multidisciplinary systems level.

Notwithstanding the need for systems level research and information on interactions among multiple stressors, there are two kinds of specific data gaps: issues data gaps and site data gaps. First, conclusions on many of the issues reviewed in the present study were based on data from outside the CCBNEP area. Specific studies in the CCBNEP study area are needed on the following:

- Effects of freshwater inflow on productivity.
- By-catch and mechanical disturbance effects of shrimp trawling on bay benthos and food webs.
- Estuarine physical circulation patterns to understand causes and maintenance of hypoxia. In particular, all current physical models in use in the CCBNEP study area are two-dimensional or 1.5-dimensional and assume a well mixed water column, so they cannot predict stratification and hypoxia that occurs.

Second, data in the CCBNEP study comes from just a few restricted sites. We have some data from Upper Laguna Madre and Baffin, Corpus Christi and Nueces Bays.

- Almost no data exists on benthos and benthic habitats north of Aransas Pass. This includes Copano, Aransas, and St. Charles Bays. This data gap is especially important because the area has different inflow conditions than bays south of Aransas Pass and different uses in the watersheds.
- We also lack data on Redfish Bay, which is the lagoonal linkage between Corpus Christi and Aransas Bays. The role of lagoonal bays, which connect primary bays, in maintaining productivity in the CCBNEP study area is unknown.

Table of Contents

VI. CONCLUSION

Benthic habitats of the CCBNEP study area, including both vegetated and unvegetated bottoms, are diverse. Vegetated bottoms provide critical habitat for numerous fish and wildlife with commercial and recreational value. Primary stresses to vegetated habitats are from both natural disturbances (e.g., brown tide) and anthropogenic disturbances (e.g., dredging and propeller scarring). However, these stresses are also synergistic. For instance, reduced light due to man's activities is especially acute during current low light conditions created by brown tide. Unvegetated habitats include bay bottoms and oyster reefs. Oyster reefs are relatively small and cover small areas of the CCBNEP study area relative to the northern estuaries of Texas. Bay bottoms are mostly soft-bottom benthic habitat consisting of mud, sand, and mud-sand sediments. Bay bottoms are the most important fishery habitat, providing a bounty of shrimp each year, and providing the food chain that supports fish species important to commercial and recreational anglers. Primary stresses to bay bottoms are from naturally low inflow/evaporation ratios and concomitant high salinities, and from anthropogenic disturbances (e.g., spills, hypoxia, freshwater diversion, and shrimp trawling). Synergistic interactions between three natural features of the CCBNEP study area (low inflow rates, low tidal energies, and high turbidity) and three anthropogenic disturbances (reduced inflows, altered circulation, and shrimp trawling) appear to be responsible for generally low productivity and low biodiversity found in open bays. This manifests itself in an environment characterized by hypersalinity, seasonal hypoxia, high sediment turnover rates, and high turbidity levels. The biotic result is a benthic community dominated by polychaetes, depauperate in molluscs and large benthos that live in deeper sediments, and monospecific nuisance algal blooms. It is highly likely that there are synergistic relationships among the multiple stressors. Sufficient information does not exist to determine the quantitative benefit if any, or all, anthropogenic disturbances could be reduced or eliminated. It is also possible that reduction in any one of them would yield a disproportionate benefit to the ecosystem. Although data is insufficient to be certain which disturbance that might be, results of this literature review indicate inflow diversion is the largest problem, because it has synergistic effects on almost all other natural and anthropogenic disturbances. However,widespread catastrophic impacts are likely to occur during a major oil transportation accident.

Table of Contents

VII. REFERENCES

Alongi , D. M. and P. Christoffersen. 1992. Benthic infauna and organism-sediment relations in a shallow, tropical coastal area: influence of out-welled mangrove detritus and physical disturbance. Marine Ecology Progress Series 81:229-245.

Anderson, D .J. 1986. Ecological succession. *in*: Community ecology: pattern and process. Kikkawa, B. J. and D. J. Anderson (eds.). Boston: Blackwell Scientific Publications. pp. 269-285.

Amos, A. F., S. C. Rabalais, and R. S. Scalan. 1983. Oil-filled Callianassa burrows on a Texas barrier-island beach. Journal of Sedimentary Petrology 53:411-416.

Armstrong, H. W., K. Fucik, J. W. Anderson and J. M. Neff. 1979. Effects of oilfield brine effluent on sediments and benthic organisms in Trinity Bay, Texas. Marine Environmental Reseach 2:55-69.

Andrew, N. L. and J. G. Peppernell. 1992. The bycatch of shrimp trawl fisheries. Oceanography and Marine Biology Annual Review 30:527-565.

Anonymous. 1995. Shrimp fishery bycatch reduction. Gulf and South Atlantic Fisheries Development Foundation. Final Report to NOAA pursuant to award NA47FM0131.

Atkinson, D. B. 1984. Discarding of small redfish in the shrimp fishery Port au Choix, Newfoundland, 1976-80. Journal Northwest Atlantic Fisheries. Society 5:99-102.

Baird, C., J. Marshall, D. Ockerman, and T. Dybala. 1996. Characterization of nonpoint sources and loadings to the Corpus Christi Bay National Estuary Program study area. Technical Report No. 5 of the Corpus Christi Bay National Estuary Program. Texas Natural Resources Conservation Commission, Austin, Texas. 226 p.

Boesch, D. F. and N. N. Rabalais (eds.). 1987. Long-term environmental effects of offshore oil and gas development. Elsevier Applied Science, London. 696 pp.

Bomar, G. W. 1983. Texas weather. University of Texas Press, Austin. 265 pp.

Boyd, N. M., B. Fuls and L. W. McEachron. 1994. Trends in relative abundance and size of selected finfishes and shellfishes along the Texas coast: November, 1975 - December, 1992. Management Data Series, No. 112.

Brehmer, M. L. 1965. Turbidity and siltation as forms of pollution. Journal of Soil and Water Conservation 20:132-133.

Brenchley, G. A. 1981. Disturbance and community structure: an experimental study of bioturbation in marine soft-bottom environments. Journal of Marine Research 39:767-790.

Britton, J. C. and B. Morton. 1989. Shore ecology of the Gulf of Mexico. University of Texas Press, Austin, TX. 387 pp.

Broks, K. M. 1996. Evaluating the environmental risks associated with the use of chromated copper arsenate-treated wood products in aquatic environments. Estuaries 19:296-305.

Browder, J. 1991. Panel 1. Effects of shrimp trawl bycatch on finish populations and ecosystems. *in*: Proceedings of the Shrimp Trawl Workshop. Center for Marine Conservation, Washington D.C. pp. 5-7.

Brown, L. F., Jr., R. A. Morton, J. H. McGowen, C. W. Kreitler and W. L. Fisher. 1974. Natural hazards of the Texas coastal zone. University of Texas Bureau of Economic Geology. Austin, Tx. 13 pp.

Brown, L. F., Jr. 1977. Environmental geologic atlas of the Texas coastal zone: Brownsville-Harlingen area. University of Texas Bureau of Economic Geology. Austin, Texas. 131 pp.

Buskey, E. J., Montagna, P. A., Amos, A. F., and Whitledge, T. E. 1997. Disruption of grazer populations as a contributing factor to the initiation of the Texas brown tide algal bloom. Limnology and Oceanography 42:1215-1222.

Caddy, J. F. 1973. Underwater observations on tracks of dredges and trawls and some effects of dredging on a scallop ground. Journal of the Fisheries Research Board of Canada 30:173-180.

Campbell, P. 1996. Personal Communication.

Campbell, R. P. and P. C. Choucair. 1995. Characterization of finfish bycatch of private-boat recreational anglers in Texas marine waters. Final report to NOAA National Marine Fisheries Service, Saltonstall-Kennedy Grant Program.

Carr, R. S., D. C. Chapman, C. L. Howard and j. M. Biedenbach. 1996. Sediment quality triad assessment survey of the Galveston, Bay, Texas system. Ecotoxicology 5:341-364.

Caudle, C. S. 1995. An impact assessment of produced water discharges to Nueces Bay - August 1993. Texas Natural Resource Conservation Commission, Austin. 30 pp.

Churchill, J. H. 1989. The effect of commercial trawling on sediment resuspension and transport over the Middle Atlantic Bight continental shelf. Continental Shelf Research 9:841-864.

Clark, D. and T. Miller-Way. 1992. An environmental assessment of the effects of open-water disposal of maintenance dredged material on benthic resources in Mobile Bay, Alabama. Report number: Miscellaneous Paper D-92-1, US Army Engineer Waterways Experiment Station, Vicksburg, MS.

Clements, F. E. 1916. Plant Succession: an analysis of the development of vegetation. Carnagie Institute of Washington: Washington. 512 pp.

Clements, F. E. 1936. Nature and structure of the climax. Journal of Ecology 28:254-422.

Cody, T., P. C. Hammerschmidt, G. C. Matlock, C. E. Bryan, J. E. Clark, and R. P. Campbell. 1989. Texas Shrimp Fishery Management Plan. Fishery Management Plan Series, No. 2. Texas Parks and Wildlife Department, Coastal Fisheries Branch. Austin, Texas. 57 p.

Commito, J. A. 1982. Importance of predation by infaunal polychaetes in controlling the structure of a soft-bottom community in Maine, USA. Marine Biology 68:77-81.

Conley, M. F. 1996. Effect of a persistent brown tide on macroinfaunal communities in Baffin Bay and Laguna Madre, Texas. Masters Thesis, University of Texas at Austin, Department of Marine Science.

Conley, M. F., P. A. Montagna, R. D. Kalke, D. A. Stockwell. Relationship between macroinfaunal diversity and community stability, and a disturbance caused by a persistent brown tide bloom. In preparation.

Connell, J. H. and R. O. Slatyer. 1977. Mechanisms of succession in natural communities and their role in community stability and organization. The American Naturalist 111:1119-1144.

Cowles, H. C. 1899. The ecological relations of the vegetation on the sand dunes of Lake Michigan. Botanical Gazette 27:95-117, 167-202, 281-308, 361-391.

Cowles, H. C. 1901. The physiographic ecology of Chicago and vicinity; a study of the origin, development, and classification of plant societies. Botanical Gazette 31(2):73-182.

Crowe, A. L. and C. E. Bryan. 1986. A description of the Texas shrimp fleet, 1979 - 1983. Management Data Series, No. 101. Texas Parks and Wildlife Department, Coastal Fisheries Division. Austin, Texas. 11 p.

Crowe, A. L. and C. E. Bryan. 1987. Trends in composition of the Texas commercial shrimp fleet. Management Data Series, No. 121. Texas Parks and Wildlife Department, Coastal Fisheries Division. Austin, Texas. 11 p.

Cullen, D. J. 1973. Bioturbation of superficial marine sediments by interstitial meiobenthos. Nature 242:323-324.

Cyrus, D. P. and S. J. M. Blaber. 1992. Turbidity and salinity in a tropical northern Australian estuary and their influence on fish distribution. Estuarine, Coastal and Shelf Science 35: 545-563.

Dankers, N. and D. R. Zuidema. 1995. The role of the mussel (*Mytilus edulis* L.) and mussel culture in the Dutch Waden Sea. Estuaries 18:71-80.

Dauer, D. M. 1993. Biological criteria, environmental health and estuarine macrobenthic community structure. Marine Pollution Bulletin 26:249-257.

Dayton, P. K., S. F. Thrush, M. T. Agardy, and R. J. Hofman. 1995. Environmental effects of marine fishing. Aquatic Conservation: Marine and Freshwater Ecosystems. 5:205-232.

De Jonge, V. N. and J. Van den Bergs. 1987. Experiments on the resuspension of estuarine sediments containing benthic diatoms. Estuarine, Coastal, and Shelf Science 24:725-740.

Department of the Army, Corps of Engineers. 1995. Waterborne commerce of the United States. Part 2 - Waterways and harbors Gulf Coast, Mississippi River System and Antilles, for calendar year 1994. Water Resources Support Center WRSC-WCUS-94-2.

DePatra, K. D. and L. A. Levin. 1989. Evidence of the passive deposition of meiofauna into fiddler crab burrows. Journal of Experimental Marine Biology and Ecology 125:173-192.

Ditton, R. B. 1977. Coastal Resources Management. Lexington, MA:Lexington Books. 196 pp.

D'Unger, C., D. Chapman and R. S. Carr. 1996. Discharge of oilfield-produced water in Nueces Bay, Texas: case study. Environmental Management 20:143-150.

Dunton, K.H. 1994. Seasonal growth and biomass of the subtropical seagrass *Halodule wrightii* in relation to continuous measurements of underwater irradiance. Marine Biology 120:479-489.

Durako, M. J., M. O. Hall, F. Sargent, and S. Peck. 1992. Propeller scars in seagrass beds: an assessment and experimental study of recolonization in Weedon Island State Preserve, Florida. In: Proceedings of the 19th Annual Conference of Wetland Restoration and Creation, Weeb, F. (ed.), Hillsborough Community College, Tampa, Fl. pp. 42-53

Edwards, S. F. 1987. An introduction to coastal zone economics: concepts, methods, and case studies. New York, NY:Taylor and Francis. 134 pp.

Eleuterius, L.N. 1987. Seagrass ecology along the coasts of Alabama, Louisiana, and Mississippi. In: Proceedings of the symposium on subtropical-tropical seagrasses of the Southeastern United States, Durako, M. J., Phillips, R. C., and Lewis, R. R., III, (eds.), Florida Department of Natural Resources. St. Petersburg, Florida, pp. 11-24.

Ellis, M. J. 1986. The Hurricane Almanac. Caribbean Blue, Inc. Portland, Texas. 168 pp.

Ellis, M. M. 1936. Erosion silt as a factor in aquatic environments. Ecology 17:29-42.

Engler, R. M. 1990. Managing dredged materials. Oceanus 33:63-69.

Fegley, S. R. 1988. A comparison of meiofaunal settlement onto the sediment surface and recolonization of defaunated sandy sediment. Journal of Experimental Biology and Ecology 123:97-113.

Fesenmaier, D. R., S. Um, W. S. Roehl, A. S. Mills, T. Ozuna, Jr., L. L. Jones and R. Guajarado Q. 1987. Regional and statewide economic impacts of sportfishing, other recreational activities, and commercial fishing associated with bays and estuaries of the Texas Gulf Coast. TAES, Texas A&M Univ. College Station, Texas, 55 pp.

Fisher, W. L. 1973. Environmental carrying capacity: a geologic basis for management in the Texas coastal zone. Abstracts with Programs:621.

Flint, R. W. and R. D. Kalke. 1986. Biological enhancement of estuarine benthic community structure. Marine Ecology Progress Series 31:23-33.

Flint, R. W. and J. A. Younk. 1983. Estuarine benthos: long-term community structure variations, Corpus Christi Bay, Texas. Estuaries 6:126-141.

Floderus, S., and L. Pihl. 1990. Resuspension in the Kattegat: Impact of variation in wind climate and fishery. Estuarine, Coastal and Shelf Science 31: 487-498.

Fonseca, M. S., Fisher, J. S., Zieman, J. C. and Thayer, G. W. 1982. Influence of the seagrass, *Zostera marina* L., on current flow. Estuarine, Coastal and Shelf Science 15:351-364.

Food and Agriculture Organization. 1994. A global assessment of fisheries bycatch and discards. U.N. FAO Fish. Tech. Paper No. 339.

Forbes, T. L., V. E. Forbes, and M. H. Depledge. 1994. Individual physiological responses to environmental hypoxia and organic enrichment: implications for early soft-bottom community succession. Journal of Marine Research 52:1081-1100.

Fuls, B. 1995. Assessment of composition and magnitude of bycatch associated with the commercial shrimp trawling industry in the central lower Texas coastal bays during the spring and fall Texas commercial bay-shrimp open seasons. Final report for the Saltonstall-Kennedy Grant Program. Texas Parks and Wildlife Department, Rockport, Texas.

Gallagher, E. D., P. A. Jumars, and D. D. Trueblood. 1983. Facilitation of soft-bottom benthic succession by tube builders. Ecology 64:1200-1216.

Ginsburg, R. N. and Lowenstam, H. A. 1956. The influence of marine bottom communities on the depositional environment of sediments. Journal of Geology. 66. 310-318.

Graham, M. 1955. Effect of trawling on animals of the sea bed. Deep-Sea Research (Papers in Marine Biology and Oceanography) Suppl. to Vol. 3: 1-6.

Grassle, J. F. and L. S. Morse-Porteous. 1987. Macrofaunal colonization of disturbed deep-sea environments and the structure of deep-sea benthic communities. Deep Sea Research 34:1911-1950.

Green, R. H. and P. Montagna. 1996. Implications for monitoring: study designs and interpretation of results. Canadian Journal of Fisheries and Aquatic Sciences 53:2629-2636.

Gunter, G. 1945. Studies on marine fishes of Texas. Publications of the Institute of Marine Science 1:1-190.

Hall, S. J., D. Raffaelli, and S. F. Thrush. 1994. Patchiness and disturbance in shallow water benthic assemblages. In: Aquatic ecology: scale, pattern, and process, Hildrew, A. G., R. D. Raffaelli and P.S. Giller (eds.), Blackwell Scientific Publications, Oxford, pp. 333-375.

Hayes, J. W., and M. J. Rutledge. 1991. Relationship between turbidity and fish diets in Lakes Waahi and Whangape, New Zealand. New Zealand Journal of Marine and Freshwater Research 25: 297-304.

Henry, W. K., D. M. Driscoll and J. P. McCormack. 1975. Hurricanes of the Texas Coast. Center for Applied Geosciences, Texas A&M Univ. College Station, Texas. 48 pp.

Hoar, P., J. Hoey, J. Nance and C. Nelson. 1992. A research plan addressing finfish bycatch in the Gulf of Mexico and South Atlantic shrimp fisheries. Gulf and South Atlantic Fishery Development Foundation, Tampa, Florida.

Holmes, C. W., C. J. McLerran, and E. A. Slade. 1973. Biological activity and channelization as factors in the distribution and sedimentation of heavy metals in two marine estuarine systems. Abstracts with Programs - Geological Society of America 5:673.

Horn, H. S. 1974. The ecology of secondary succession. Annual Review of Ecology and Systematics 5:25-37.

Hudson, A. V. and R. W. Furness. 1988. Utilization of discarded fish by scavenging seabirds behind whitefish trawlers in Shetland. Journal of Zoology 215:151-166.

Iversen, E. S., D. M. Allen, and J. B. Higman. 1993. Shrimp capture and culture fisheries of the United States. Halsted Press. New York. 247 p.

James, W. P., S. Giesler, R. DeOtte and M. Inoue. 1977. Environmental considerations relating to operation and maintenance of the Texas Gulf Intercoastal Waterway. Texas A&M University Sea Grant Publ., TAMU-SG-78-204, College Station, Texas.

Jokiel, P. L. 1989. Effects of marine mining dredge spoils on eggs and larvae of a commercially important species of fish, the Mahimahi (*Coryphaena hippurus*). Marine Mining 8: 303-315.

Jones, J. B. 1992. Environmental impact of trawling on the seabed: a review. New Zealand Journal of Marine and Freshwater Research 26: 59-67.

Jones, L. L. 1997. Economic impacts on regional and State economies of human uses of the coastal resources of the Corpus Christi Bay National Estuary Program Study Area. Technical Report No. 16 of the Corpus Christi Bay National Estuary Program. Texas Natural Resources Conservation Commission, Austin, Texas. 62 p.

Kaiser, M. J., and B. E. Spencer. 1994. Fish scavenging behaviour in recently trawled areas. Marine Ecology Progress Series 112: 41-49.

Kaiser, M. J., and B. E. Spencer. 1996. The effects of beam-trawl disturbance on infaunal communities in different habitats. Journal of Animal Ecology 65: 348-358.

Kalke, R. and P. A. Montagna. 1991. The effect of freshwater inflow on macrobenthos in the Lavaca River delta and upper Lavaca Bay, Texas. Contributions in Marine Science 32:49-77.

Kelly, J. R., T. W. Duke, M. A. Harwell and C. C. Harwell. 1987. An ecosystem perspective on potential impacts of drilling fluid discharges on seagrasses. Environmental Management 11:537-562.

Kennelly, S. J. 1992. The effect of shrimp trawling bycatch other commercial and recreational fisheries - an Australian perspective. In: International Conference on Shrimp Bycatch. Southeastern Fisheries Association, Tallahassee, Florida, pp. 97-114.

Kennelly, S. J. 1995. The issue of bycatch in Australia’s demersal trawl fishery. Reviews in Fish Biology and Fisheries. 5:213-234.

Kennicutt, M. C., III, Boothe, P. N., Wade, T. L., Sweet, S. T., Rezak, R., Kelly, F. J., Brooks, J. M., Presley, B. J. and Wiesenburg, D. A. 1996. Geochemical patterns in sediments near offshore production platforms. Canadian Journal of Fisheries and Aquatic Sciences 53:2554-2566.

Knott, D. M., E. L. Wenner and P. H. Wendt. 1997. Effects of pipeline construction on th evegetation and macrofauna of two South Carolina, USA salt marshes. Wetlands 17:65-81.

Krost, P., M. Bernhard, F. Werner, and W. Hukriede. 1990. Otter trawl tracks in Kiel Bay (Western Baltic) mapped by side-scan sonar. Meeresforsch 32: 344-353.

Koch, E. W. and C. J. Dawes. 1991. Influence of salinity and temperature on the germination of *Ruppia maritima* L. From the North Atlantic and Gulf of Mexico. Aq. Bot. 40:387-391.

Lalli, C. M. and T. R. Parsons. 1993. Biological Oceanography: an introduction. Butterworth Heinemann, Oxford, 199pp.

Llansó, R. J. 1991. Tolerance of low dissolved oxygen and hydrogen sulfide by the polychaete *Streblospio benedicti* (Webster). Journal of Experimental Marine biology and Ecology 153:165-178.

Leslie, J. K., and C. A. Timmins. 1992. Distribution and abundance of larval fish in Hamilton Harbour, a severely degraded embayment of Lake Ontario. Journal of Great Lakes Research 18: 700-708.

Levin, L. A. and C. R. Smith. 1984. Response of background fauna to disturbance and enrichment in the deep sea: a sediment tray experiment. Deep Sea Research 31:1277-1285.

Levings, S. C. and S. D. Garrity. 1994. Effects of oil spills on fringing red mangroves (*Rhizophora mangle*): losses of mobile species associated with submerged prop roots. Bulletin of Marine Science 54:782-794.

Lewis, F. G. and A. W. Stoner. 1983. Distribution of macrofauna within seagrass beds: an explanation for patterns of abundance. Bulletin of Marine Science 33:296-304.

Lofgren, B. E. 1977. Background studies for appraising subsidence in the Texas Gulf coast region. U.S.G.S. Open-file Report 77-412. U.S. Department of Interior, Geological Survey.

Longley, W. L., ed. 1994. Freshwater inflows to Texas bays and estuaries: ecological relationships and methods for determination of needs. Texas Water Development Board and Texas Parks and Wildlife Department, Austin, Texas.

Lubchenco, J. and B. A. Menge. 1978. Community development and persistence in a low rocky intertidal zone. Ecological Monographs 59:67-94.

Luckenbach, M. W. 1986. Sediment stability around animal tubes: the roles of hydrodynamic processes and biotic activity. Limnology and Oceanography 31:779-787.

Luckenbach, M. 1987. Effects of adult infauna on new recruits: implications for the role of biogenic refuges. Journal of Experimental Marine Biology and Ecology 105:197-206.

Main, J. and G. I. Sangster. 1981. A study of sand clouds produced by trawl boards and their possible effects on fish capture. Scottish Fisheries Research Report No. 20. Dept of Agriculture and Fisheries for Scotland. Aberdeen, Scotland. 19 pp.

Mannino, A. and P. A. Montagna. 1996. Fine-scale spatial variation of sediment composition and salinity in Nueces Bay of South Texas. Texas Journal of Science 48:35-47.

Mannino, A. and P. A. Montagna. 1997. Small-scale spatial variation of macrobenthic community structure. Estuaries 20:159-173.

Martin, C. M., Montagna, P. A. 1995. Environmental assessment of La Quinta Channel, Corpus Christi Bay, Texas. Texas Journal of Science 47:203-222.

Maurer, D., R. T. Keck, J. C. Tinsman, W. A. Leathem, C. Wethe, C. Lord, and T. M. Church. 1986. Vertical migration and mortality of marine benthos in dredged material: a synthesis. International revue gestam Hydrobiologia 71:49-63.

Matlock, G. C. 1982. Bycatch of southern flounder and gulf flounder by commercial shrimp trawlers in Texas bays. Texas Parks and Wildlife Department, Coastal Fisheries Branch, Management Data Series, No. 31,

McMillan, C. and Moseley, F. N. 1967. Salinity tolerances of five aquatic spermatophytes of Redfish Bay, Texas. Ecology. 48:503-506.

Messieh, S. N., T. W. Rowell, D. L. Peer and P. J. Cranford. 1991. The effects of trawling, dredging and ocean dumping on the eastern Canadian continental shelf seabed. Continental Shelf Research 11: 1237-1263.

Mills, E. L. 1969. The community concept in marine zoology, with comments on continua and instability in some marine communities: a review. Journal Fisheries Research Board of Canada 26(6):1415-1428.

Moffett, A. W. 1970. The shrimp fishery in Texas. Texas Parks and Wildlife Department. Austin, Texas. 38 p.

Montagna, P. A. 1993. Comparison of ecosystem structure and function of created and natural seagrass habitats in Laguna Madre, Texas. Annual Report to U.S. Environmental Protection Agency, Region 6. Marine Science Institute, The University of Texas, Port Aransas, TX, 8 pp.

Montagna, P. A. and D. E. Harper, Jr. 1996. Benthic infaunal long-term response to offshore production platforms. Canadian Journal of Fisheries and Aquatic Sciences 53:2567-2588.

Montagna, P. A. and R. D. Kalke. 1992. The effect of freshwater inflow on meiofaunal and macrofaunal populations in the Guadalupe and Nueces estuaries, Texas. Estuaries 15:307-326.

Montagna, P. A. and R. D. Kalke. 1995. Ecology of infaunal Mollusca in south Texas estuaries. American Malacological Bulletin 11:163-175.

Montagna, P. A., and Li, J. 1996. Modeling and monitoring long-term change in macrobenthos in Texas estuaries. Final Report to the Texas Water Development Board. University of Texas at Austin, Marine Science Institute, Technical Report No. TR/96-001, Port Aransas, Texas, 149 pp.

Montagna, P.A., J. Li and G. T. Street. 1996. A conceptual model of the Corpus Christi Bay National Estuary Program study area. Technical Report No. 8 of the Corpus Christi Bay National Estuary Program. Texas Natural Resources Conservation Commission, Austin, Texas.

Montagna, P. A., D. A. Stockwell and R. D. Kalke. 1993. Dwarf surfclam *Mulinia lateralis* (Say, 1822) populations and feeding during the Texas brown tide event. Journal of Shellfish Research 12:433-442.

Montagna, P. A. and W. B. Yoon. 1991. The effect of freshwater inflow on meiofaunal consumption of sediment bacteria and microphytobenthos in San Antonio Bay, Texas, U.S.A. Estuarine and Coastal Shelf Science 33:529-547.

Moran, P. J. and T. R. Grant. 1989a. The effects of industrial pollution on the development and succession of marine fouling communities. I. Analysis of species richness and frequency data. Marine Ecology 10:231-246.

Moran, P. J. and T. R. Grant. 1989b. The effects of industrial pollution on the development and succession of marine fouling communities. II. Multivariate analysis of succession. Marine Ecology 10:247-261.

Nance, J. M. 1991. Effects of oil/gas field produced water on the macrobenthic community in a small gradient estuary. Hydrobiologia 220:189-204.

National Marine Fisheries Service. 1990. Evaluation of bycatch reduction devices (BRDs). NMFS Pascagoula/Galveston Labs.

NMFS. 1995. Cooperative research program addressing finfish bycatch in the Gulf of Mexico and South Atlantic shrimp fisheries: A report to congress. National Marine Fisheries Service. Southeast Regional Office, St. Petersberg, Florida, 55 pp.

Nessmith, C. 1980. Natural and induced revegetation processes in an artificially disturbed seagrass meadow in Texas. Master’s Thesis. The University of Texas at Austin.

Nichols, S. 1991. Panel 1. Effects of shrimp trawl bycatch on finish populations and ecosystems. In: Proceedings of the Shrimp Trawl Workshop, Center for Marine Conservation, Washington, D.C., pp. 5-7.

Nichols, S., A. Shah, G. Pellegrin, Jr. and K. Mullin. 1990. Updated estimates of annual shrimp fleet bycatch in the offshore waters of the Gulf of Mexico. NMFS, Southeast Fisheries Center, Pascagoula Lab.

Odum, E. P. 1969. The strategy of ecosystem development. Science 164:262-270.

Odum, H. T. 1963. Productivity measurements in Texas turtle grass and the effects of dredging an intercoastal channel. Publications of the Institute of Marine Science 9:48-58.

Officer, C. B., R. B. Biggs, J. L. Taft, L. E. Cronin, M. A. Tyler and W. R. Boynton. 1984. Chesapeake Bay Anoxia: origin, development, and significance. Science 223:22-27.

Onuf, C. P. 1994. Seagrasses, dredging and light in Laguna Madre, Texas, U.S.A. Estuarine, Coastal, and Shelf Science 39:75-91.

Oppenheimer, C.H. 1963. Effects of hurricane Carla on the ecology of Redfish Bay, Texas. Bulletin of Marine Science of the Gulf and Caribbean 13: 59-72.

Orlando, S.P. et al. 1991. Analysis of Salinity Structure and Stability for Texas Estuaries. Strategic Assessment Branch, NOS/NOAA. Rockville, M. 97 pp.

Orth, R. J. 1971. The effect of turtle grass, *Thalassia testudinum*, on the benthic infauna community structure in Bermuda. Bermuda Biological Station Special Report No. 9: 18-37.

Orth, R. J. 1977. The importance of sediment stability in seagrass communities. In: Ecology of marine benthos, B.C. Coull (ed.), Univ. of South Carolina Press, Columbia, pp. 281-300.

Palmer, M. A., P. A. Montagna, R. B. Spies and D. Hardin. 1988. Meiofauna dispersal near natural petroleum seeps in the Santa Barbara Channel: a recolonization experiment. Oil & Chemical Pollution 4:179-189.

Paine, J. G. 1993. Subsidence of the Texas coast: inferences from historical and late Pleistocene sea levels. Tectonophysics 222:445-458.

Parker, R. H. 1959. Macro-invertebrate assemblages of central Texas coastal bays and Laguna Madre. Bulletin of the American Association of Petroleum Geologists 43:2100-2166.

PCCA 1993. Phase II: Safeharbor environmental impact statement. Port of Corpus Christi Authority. p. 2.4-19.

Pearson, T. H. and R. Rosenberg. 1978. Macrobenthic succession in relation to organic enrichment and pollution of the marine environment. Oceanography and Marine Biology Annual Review 16: 229-311.

Phillips, R. C. 1960. Observations of the ecology and distribution of Florida seagrasses. Professional Papers Serial for the Florida Board of conservation 2:1-72.

Pinto, B., D. Pelligrini, M. Gabellini, and A. Ausili. 1995. Harbor and coastal sediment chemistry and toxicity: a preliminary assessment of dredging activities. Journal of Aquatic Ecosystem Health 4:249-255.

Pokryfki, L. and R. E. Randall. 1987. Nearshore hypoxia in the bottom water of the Northwestern Gulf of Mexico from 1981 to 1984. Marine Environmental Research 22:75-90.

Pomeroy, L. R. and W. J. Wiebe. 1988. Energetics of microbial food webs. Hydrobiologia 159:7-18.

Ponwith, B. and Q. R. Dokken. 1996. Fisheries resources. In: Tunnell, J. W., Jr., Q. R. Dokken, E. H. Smith and K. Withers, Current Status and Historical Trends of the Estuarine Living Resources within the Corpus Christi Bay National Estuary Program Study Area, Vol. 1. Technical Report No. 6A of the Corpus Christi Bay National Estuary Program. Texas Natural Resources Conservation Commission, Austin, Texas. Pp. 484-582.

Poore, G. C. B. 1982. Benthic communities of the Gippsland Lakes, Victoria. Australian Journal of Marine and Freshwater Research 33:901-915.

Quammen, M. L. and C. P. Onuf. 1993. Laguna Madre: seagrass changes continue decades after salinity reduction. Estuaries 16: 302-310.

Ratzlaff, K. 1982. Land-surface subsidence in the Texas coastal region. Report 272. Texas Department of Water Resources, Austin, Texas.

Rayburn, R. 1992. Shrimp bycatch from the perspective of the state's role as fishery trustee. International Conference on Shrimp ByCatch. National Oceanic and Atmospheric/National Marine Fisheries Service, MARFIN Grant NA-90AA-H-MF745. 364 p.

Reise, K. 1978. Experiments on epibenthic predation in the Wadden Sea. Helgolander Wiss. Meeresunters. 31:55-101.

Rhoads, D. C. 1973. The influence of deposit-feeding benthos on water turbidity and nutrient recycling. American Journal of Science 272:1-22.

Rhoads, D. C. 1974. Organism-sediment relations on the muddy sea floor. Oceanography and Marine Biology Annual Review 12:263-300.

Rhoads, D. C., P. L. McCall and J. Y. Yingst. 1978. Disturbance and production on the estuarine seafloor. American Scientist 66: 577-586.

Rhoads, D. C. and D. K. Young. 1970. The influence of deposit-feeding organisms on sediment stability and community trophic structure. Journal of Marine Research 28:150-178.

Ritter, M. C. and P. A. Montagna. 1997. Seasonal hypoxia and benthic response in a Texas bay. Estuaries (submitted msc.).

Robinson, L., P. Campbell and L. Butler. 1994. Trends in Texas Commercial Fishery Landings, 1972-1993. Management Data Series, No. 111.

Robinson, L., P. Campbell and L. Butler. 1995. Trends in Texas commercial fishery landings, 1972-1994. Management Data Series, No. 117. Texas Parks and Wildlife Department, Coastal Fisheries Division, Austin, Texas. 133 p.

Rothlisberg, P. 1992. The fate of material discarded from shrimp trawlers. in: International Conference on Shrimp Bycatch, Southeastern Fisheries Association, Tallahassee, Florida, pp 115-124.

Rykiel, E. J., Jr. 1985. Towards a definition of ecological disturbance. Australian Journal of Ecology 10:361-365.

Santos, S. L. and J. L. Simon. 1974. Distribution and abundance of the polychaetous annelids in a south Florida estuary. Bulletin of Marine Science 24: 669-689.

Santos, S. L. and J. L. Simon. 1980. Response of soft-bottom benthos to annual catastrophic disturbance in a south Florida estuary. Marine Ecology Progress Series 3:347-355.

Sargent, F. J., T. J. Leary, D. W. Crewz, and C. R. Kruer. 1995. Scarring of Florida’s seagrasses: assessment and management options. FMRI Tech. Rep. TR-. Florida Marine Research Institute, St. Petersburg, Florida. 37 pp. plus appendices.

Savino, J. F., M. A. Blouin, B. M. Davis, P. L. Hudson, T. N. Todd and G. W. Fleischer. 1994. Effects of pulsed turbidity and vessel traffic on lake herring eggs and larvae. Journal of Great Lakes Research 20: 366-376.

Schmidheiny, S. 1992. Changing course. Cambridge MA:The MIT Press. 374 pp.

Schubel, J. R., H. H. Clarke, R. E. Wilson, W. M. Wise, M. G. Heaton and M. G. Gross. 1978. Field investigations of nature, degree and extent of turbidity generated by open-water pipeline disposal operations. Technical Report D-78-30, U. S. Army Engineer Waterways Experiment Station, CE, Vicksburg, Mississippi, 245 pp. plus appendices.

Schwartz, F. J., G. Safrit, J. Purifoy and R. Butterfield. 1993. A 10-month survey of fishes and invertebrates inhabiting a 30-yr old barrier island dredged canal system at Pine Knoll Shores, North Carolina. The Journal of the Elisha Mitchell Scientific Society 109:99-114.

Schwinghamer, P., J. Y. Guigné and W. C. Siu. 1996. Quantifying the impact of trawling on benthic habitat structure using high resolution acoustics and chaos theory. Canadian Journal of Fish and Aquaculture Science 53:288-296.

Scoffin, T. P. 1970. The trapping and binding of subtidal carbonate sediments by marine vegetation in Bimini Lagoon, Bahamas. J. Sed. Pet. 40(1-2). 241-273.

Shepard, F. P. 1953. Sedimentation rates in Texas estuaries and lagoons. Bulletin of the American Association of Petroleum Geologists 37:1919-1934.

Sherk, J. A., Jr. and L. E. Cronin. 1970. The effects of suspended and deposited sediments on estuarine organisms, an annotated bibliography of selected references. University of Maryland, N.R.I. Ref. No. 70-19, 561pp.

Sherman, K. M. and B. C. Coull. 1980. The response of meiofauna to sediment disturbance. Journal of Experimental Biology and Ecology 46:59-71.

Sheridan P. F., J. A. Browder and J. E. Powers. 1984. Ecological interactions between penaeid shrimp and bottomfish assemblages. In: Penaeid Shrimps - their Biology and Management, Gulland, J. A. and B. J. Rothschild, Fishing News Books, Farnham, England pp 235-253.

Shideler, G. L 1980. Reconnaissance observations of some factors influencing the turbidity structure of a restricted estuary: Corpus Christi Bay, Texas. Texas Journal of Science 32:60-71.

Shideler, G. L. 1984. Suspended sediment responses in a wind-dominated estuary of the Texas Gulf coast. Journal of Sedimentary Petrology 54: 731-745.

Sideler, G. L. and C. E. Stelting. 1981. Sedimentary processes in restricted Gulf coast estuarine system: Corpus Christi Bay, Texas. AAPG Bulletin 65:992.

Simboura, N., A. Zenetos, P. Panayotidis and A. Makra. 1995. Changes in benthic community structure along an environmental pollution gradient. Marine Pollution Bulletin 30: 470-474.

Smayda, T J. and Y. Shimuzu (eds.). 1993. Toxic phytoplankton blooms in the sea. Esevier Press, Amsterdam, 952 p.

Smith, V. J., R. J. Swindlehurst, P. A. Johnston and A. D. Vethaak. 1995. Disturbance of host defense capability in the common shrimp, *Crangon crangon*, by exposure to harbor dredge spoils. Aquatic Toxicology 32:43-58.

Soule, D. F. 1988. Marine organisms as indicators: reality or wishful thinking? In: Marine Organisms as Indicators, Soule, D. F. and G. S. Kleppel (eds.),Springer-Verlag, New York, pp. 1-11.

Sousa, W. P. and J. H. Connell. 1992. Grazing and succession in marine algae. In: Plant animal interactions in the marine benthos, D. M. John, S. J. Hawkins, and J. H. Price (eds.), Systematics Association Special Volume No. 46, Clarendon Press, Oxford. pp. 447-491.

Stephens, P. M. 1987. Response of excised gill tissue from the New Zealand scallop, *Pectin novaezelandiae*, to suspend silt. New Zealand Journal of Marine and Freshwater Research 21:605-614.

Stoner, A. W. 1980. Abundance, reproductive seasonality and habitat preferences of amphipod crustaceans in seagrass meadows of Apalachee Bay, Florida. Contributions in Marine Science 23:63-77.

Summerson, H. C. and C. H. Peterson. 1984. Role of predation in organizing benthic communities of a temperate-zone seagrass bed. Marine Ecology Progress Series 15:63-77.

Sutherland, J. P. and R. H. Karlson. 1977. Development and stability of the fouling community at Beaufort, North Carolina. Ecological Monographs 47:425-446.

Swanson, R. L. and C. I. Thurlow. 1973. Recent subsidence rates along the Texas and Louisiana coasts as determined from tide measurements. Journal of Geophysical Research 78:2665-2671.

Taghon, G. L., Nowell, A. R. M. and P. A. Jumars. 1980. Induction of suspension feeding in spionid polychaetes by high particulate fluxes. Science 210:562-564.

Texas Department of Water Resources. 1982. The influence of freshwater inflows upon the major bays and estuaries of the Texas Gulf coast. Vol. 8. Executive Summary (Second Ed.) Texas Department of Water Resources, Austin, Texas.

TGFOC 1923. Annual report of the State of Texas Game, F TGFOC 1923. Annual report of the State of Texas Game, fish and Oyster Commission ish and Oyster Commission for the fiscal year 1922-1923. Austin, Texas.

Texas Maritime Museum. Public display. Rockport, Texas.

Texas Parks Wildlife Department. 1995a. Texas Commercial Fishing Guide 1995-1996. Texas Parks and Wildlife Department, Austin, Texas. 23 p.

Texas Parks Wildlife Department. 1995b. The Texas Shrimp Fishery. A report to the Governor and the 74th Legislature. Texas Parks and Wildlife Department, Austin, Texas. 22 p.

Thayer, G. W., S. M. Adams, and M. W. La Croix. 1975. Structural and functional aspects of a recently established *Zostera marina* community. Estuarine Research 1:518-540.

Thomas, L. P., Moore, D. R., and Work, R. C. 1961. Effects of Hurricane Donna on the turtle grass beds of Biscayne Bay, Florida. Bulletin of Marine Science of the Gulf and Caribbean 11. 191-197.

Thrush, S. F., J. E. Hewitt, V. J. Cummings and P. K. Dayton. 1995. The impact of habitat disturbance by scallop dredging on marine benthic communities: what can be predicted from the results of experiments? Marine Ecology Progress Series 129: 141-150.

Tunnell, J. W., Jr., Q. R. Dokken, E. H. Smith and K. Withers. 1996. Current Status and Historical Trends of the Estuarine Living Resources within the Corpus Christi Bay National Estuary Program Study Area, Vol. 1 of 4. Technical Report No. 6A of the Corpus Christi Bay National Estuary Program. Texas Natural Resources Conservation Commission, Austin, Texas. 543 p.

Turner, S. J., S. F. Thrush, V. J. Cummings, J. E. Hewitt, M. R. Wilkinson, R. B. Williamson and D. J. Lee. 1997. Changes in epifaunal assemblages in response to marina operations and boating activities. Marine Environmental Research 43:181-199.

U.S. Army Corps of Engineers. 1968. Report on Hurricane "Beulah", 8-21 September, 1967. U. S. Army Corps of Engineers, Galveston Dist. 26 pp.

U.S. Army Corps of Engineers. 1971. Report on Hurricane "Celia", 30 July-5 August, 1970. U. S. Army Corps of Engineers, Galveston Dist. 13 pp.

U. S. Army Corps of Engineers. 1972. Report on Hurricane "Fern", September 7-13, 1971. U. S. Army Corps of Engineers, Galveston Dist. 11 pp. April 25, 1997

U.S. Army Corps of Engineers. 1979. Waterborne Commerce of the United States: Part 2 - Waterways and Harbors, Gulf Coast, Mississippi River System and Antilles. Department of Army, Corps of Engineers, Water Resources Support Center. WRSC-WCUS-79-2.

U.S. Army Corps of Engineers. 1981. Report on Hurricane "Allen", 3-10 August, 1980. U. S. Army Corps of Engineers, Galveston Dist. 28 pp.

U.S. Army Corps of Engineers. 1994. Waterborne Commerce of the United States: Part 2 - Waterways and Harbors, Gulf Coast, Mississippi River System and Antilles. Department of Army, Corps of Engineers, Water Resources Support Center. WRSC-WCUS-94-2.

Van Dolah, R. F., P. H. Wendt and M. V. Levisen. 1991. A study of the effects of shrimp trawling on benthic communities in two South Carolina sounds. Fisheries Research 12: 139-156.

Van Dolah, R. F., P. H. Wendt and N. Nicholson. 1987. Effects of a research trawl on a hard-bottom assemblage of sponges and corals. Fisheries Research 5: 39-54.

van Tussenbroek, B. 1994. The impact of Hurricane Gilbert on the vegetation development of *Thalassia testudinum* in Puerto Morelos Coral Reef Lagoon, Mexico: a retrospective study. Botanica Marina 37:421-428.

Varon, R. and D. Thistle. 1988. Response of a harpacticoid copepod to a small-scale natural disturbance. Journal of Experimental Marine Biology and Ecology 118:245-256.

Virnstein, R. W., P. S. Mikkelson, K. D. Cairns, and M. A. Capone. 1983. Seagrass beds vs. Sand bottoms: the trophic importance of their associated benthic invertebrates. Florida Science 46:363-381.

Walker, K. R. and L. P. Alberstadt. 1975. Ecological succession as an aspect of structure in fossil communities. Paleobiology 1: 238-257.

Ward, G. H. 1980. Hydrography and circulation processes of Gulf estuaries. In: Estuarine and Wetland Processes with emphasis on modeling, Hamilton, P. and Macdonals, K. B. (eds.), Plenum Press, New York, pp. 183-215.

Ward, G.H. 1997. Processes and trends of circulation within the Corpus Christi Bay National Estuary Program Study Area. Technical Report No. 21 of the Corpus Christi Bay National Estuary Program. Texas Natural Resources Conservation Commission, Austin, Texas. 286 p. plus appendices.

Warren, T. A., L. M. Green and K. W. Spiller. 1994. Trends in finfish landings of sport-boat anglers in Texas Marine Waters, May, 1974-May 1992. Management Data Series, No. 109.

Warwick, R. M. and K. R. Clarke. 1994. Relearning the ABC: taxonomic changes and abundance/biomass relationships in disturbed benthic communities. Marine Biology 118: 739-774.

Weber, D. E., D. A. Flemer and C. M. Bundrick. 1992. Comparison of the effects of drilling fluid on macrobenthic invertebrates associated with the seagrass, _*Thalassia testudinum*, in the laboratory and field. Estuarine Coastal and Shelf Science 35:315-330.

Weis, J. S. and P. Weis. 1996. The effects of using wood treated with chromated copper arsenate in shallow water environments: a review. Estuaries19:306-310.

Wilson, M. A. 1987. Ecological dynamics on pebbles, cobbles, and boulders. Palaios 2: 594-599.

Windom, H. L. 1972. Environmental aspects of dredging in estuaries. Journal of Waterways, Harbors and Coastal Engineering Division, ASCE 98(WW4):475-487.

Windom, H. L. 1975. Water-quality aspects of dredging and dredge-spoil disposal in estuarine environments. Pages 559-571 in L. E. Cronin, ed. Estuarine Research. Vol. II. New York:Academic Press.

Wirth, E. F., G. I. Scott, M. H. Fulton, R. F. VanDolah, P. P. Maier, N. Hadley, J. W. Daugomah, and P. B. Key. 1996. In situ monitoring of dredged material spoil sites using the oyster *Crassostrea virginica*. Archives of Environmental Contamination and Toxicology 30:340-348.

Wood, T. M., R. L. Lehman and J. Bonner. 1997. Ecological impacts of bioremediation field study experiments. Pages 119-124 in: In situ and on-site bioremediation: Volume 4. Battelle Press, Columbus.

Young, D. K. and M. W. Young. 1982. Macrobenthic invertebrates in bare sand and seagrass (*Thalassia testudinum*) at Carrie Bow Cay, Belize. In: The Atlantic Barrier Reef Ecosystem at Carrie Bow Cay, Belize, 1. Structure and communities, pp. 115-126.

Zajac, R. N. and R. B. Whitlatch. 1982a. Responses of estuarine infauna to disturbance. I. Spatial and temporal variation of initial recolonization. Marine Ecology Progress Series 10: 1-14.

Zajac, R. N. and R. B. Whitlatch. 1982b. Responses of estuarine infauna to disturbance. II. Spatial and temporal variation of succession. Marine Ecology Progress Series 10: 15-27.

Zieman, J. C. 1975a. Tropical seagrass ecosystems and pollution. In: Tropical marine pollution, E. J.Ferguson Wood and R. E. Johannes (eds.), Elsevier Sci. Publ. Co., Amsterdam, pp. 63-74.

Zieman, J. C. 1975b. Quantitative and dynamic aspects of the ecology of turtle grass, *Thalassia testudinum*. In: Estuarine Research, V. 1 Chemistry, biology, and the estuarine system, L. Eugene Cronin (ed.), Academic Press, Inc. New York, pp. 541-562.

Zieman, J. C. 1976. The ecological effects of physical damage from motor boats on turtle grass beds in southern Florida. Aquatic Botany. 2:127-139.

VIII. APPENDIX A

Questions for Telephone Interviews

Characterization of Anthropogenic and Natural Disturbance on Bay Bottom Habitats in the CCBNEP Study Area

STAC/Exofficio Member:______________ Phone:
Person Interviewed:____________________ Phone:

Organization:

Does your organization maintain any public databases that would catalogue these resources?

What do you see as the key resource management concerns regarding benthic resources of the CCBNEP (Corpus Christi Bay National Estuary Program) study area?

What social and economic concerns are related to the management of these resources?

What management recommendations would you suggest to address these concerns?