SEDIMENT QUALITY ASSESSMENT SURVEY OF THE GALVESTON BAY SYSTEM

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EXECUTIVE SUMMARY

The purpose of this study was to characterize the quality of sediments at key sites in the Galveston Bay Estuary. Sediment samples were taken simultaneously for chemical and physical analyses, toxicity testing, and an assessment of benthic community structure, in order to identify areas where sediment contamination is responsible for ecosystem degradation. The chemical analyses included a suite of trace metals, polycyclic aromatic hydrocarbons (PAHs), and chlorinated hydrocarbons. Total organic carbon (TOC) and acid volatile sulfides (AVS) were also measured in order to normalize the organic and inorganic constituents, respectively. Solid-phase sediment toxicity tests with the benthic amphipod *Grandidierella japonica* were conducted. In addition, the U.S. Fish and Wildlife Service (USFWS) conducted a series of sediment porewater tests with gametes of the sea urchin *Arbacia punctulata*.

The sites selected for inclusion in this study were based upon two different sets of criteria. The original 16 sites were selected to be representative of specific areas within the Galveston Bay complex. In general, these sites were located in depositional zones away from known point source discharges. Funding provided by the the U.S. Fish and Wildlife Service (USFWS) allowed eight additional sites to be included in the survey. The criteria for selection of the eight USFWS sites was based on specific areas of concern. The USFWS sites were located adjacent to dredged material disposal islands, produced water discharges or in semi-enclosed bays adjacent to industrial or urban runoff.

Significant toxicity, as determined by the sea urchin porewater morphological development assay, was observed at 12 of the 24 sites investigated in this study which included six of the original 16 sites and six of the eight USFWS sites. The solid-phase amphipod test showed no toxicity at any of the sites. There were a number of sites with elevated levels of trace metals and petroleum hydrocarbons. Species richness and abundance data were suggestive of environmental degradation at a number of sites. The meiobenthic community appeared to be depauperate or absent from the majority of sites investigated. The chemistry, toxicity, and benthic data were ranked by station and a rank sum calculated to facilitate comparisons among the stations.

Five of the 24 sites exhibited strong evidence for contaminant-induced degradation (Burnett Bay, Cedar Bayou, South of Lake Surprise, Black Duck Bay, and F-2 separator platform) Fifteen of the 34 stations showed no evidence of contaminant-induced degradation. At eight additional sites (C-1 separator platform, Smith Point, March Point, Chocolate Bay, Eagle Point-station 2, Lost Lake Island, Atkinson Island, and the F-1 separator platform), the Sediment Quality Triad (SQT) data suggested that unmeasured chemicals or conditions were stressing the system.

This broad scale survey has identified a number of potential problem areas within the Galveston Bay system. Spearman correlation analyses suggested that decreased benthic abundance and richness, increased porewater toxicity, and increased porewater ammonia and sulfide concentrations (presumably as a result of decreased bioturbation) may be related to the presence and degree of contaminants, particularly metals. The probable source of contamination was evident at some sites and unknown at others.

From the best historical information available, it does not appear that the Galveston Bay system is, in general, more degraded than at the time of the last comprehensive survey in 1976-1977. The results of the present study suggest that there are certain areas of concern related to contaminant impacts (e.g., sites north of Morgan Point, sites adjacent to petrochemical and dredge material disposal activities). Sites sampled north of Morgan Point are undoubtedly influenced by a myriad of point and non-point source contaminant inputs. Sites adjacent to produced water separator discharges had high PAH concentrations and were highly toxic. It is apparent that the inputs from some of these sources could be reduced or eliminated by alternative regulatory and management practices including the restriction of produced water discharges into coastal estuaries and the use of dredge material disposal practices that minimize the reintroduction of sediment-associated contaminants to the bays.

INTRODUCTION

The goals of the Galveston Bay National Estuary Program (GBNEP) are to improve water quality and to enhance living resources within the Galveston Bay Estuary. In order to achieve these goals, the factors responsible for degrading the "health" of the ecosystem must be identified and prioritized. Only when the problems have been characterized, can the necessary actions be taken to enhance the vitality of the estuary.

Galveston Bay is the largest and most productive estuary in Texas. A recent report has documented declining trends in selected living resources, including species of commercial importance, in the Galveston Bay system (Walton and Green, 1992). It is suspected that the introduction of anthropogenic contaminants to the Galveston Bay Estuary is a major factor in the decline in species diversity and productivity that is apparent in certain areas of the estuary (Bechtel and Copeland, 1970; Copeland and Bechtel, 1971; Walton and Green, 1992). Other factors (e.g., changes in salinity regimes, dredging and filling activities resulting in changed circulation patterns and the accompanying alterations in sediment texture and habitat) may also be contributing to these declines.

Estuaries are the eventual repository for the contaminants that are either discharged directly into these unique environments or delivered by their life blood--the rivers and streams that feed them. The majority of the contaminants of concern entering estuaries eventually become associated with sediment particles and are deposited in the estuary. Many of the contaminants that become associated with sediments may not be bioavailable because of their chemical and physical association with particulates or organic moieties (Swartz et al., 1985). It is not possible, however, to predict which sediment samples may be toxic on the basis of analytical chemistry information alone, as has been repeatedly demonstrated (Long

and Chapman, 1985; Chapman, 1986, 1990; Chapman et al., 1987; Carr et al., 1989; Long et al., 1990).

One of the most powerful tools for determining the degree to which contaminants are responsible for the degradation of sediment quality is the Sediment Quality Triad (SQT) approach (Chapman, 1990). The SQT is an effects-based approach for evaluating and assessing pollution-induced degradation consisting of three components: sediment chemistry (a measure of contamination), sediment toxicity tests (measures of bioavailability and biological effects), and *in situ* parameters (e.g., alterations of benthic community structure). The information provided by each component is unique and complementary. All three measures are essential for determining sediment quality because no single component provides comprehensive information.

The purpose of this study was to characterize the sediments at key sites in the Galveston Bay Estuary using the SQT approach in order to identify areas where sediment contamination is responsible for ecosystem degradation. Simultaneously collected sediment samples were taken for chemical and physical analyses, toxicity testing, and an assessment of benthic community structure. The chemical analyses included a suite of trace metals, polycyclic aromatic hydrocarbons (PAHs), and pesticides. Total organic carbon (TOC), and acid volatile sulfides (AVS) were also measured in order to normalize the organic and inorganic constituents, respectively. Solid-phase sediment toxicity tests with the benthic amphipod *Grandidierella japonica* were conducted (ASTM, 1990). In addition, the USFWS conducted a series of sediment pore (interstitial) water tests (Carr et al., 1989; Carr and Chapman, 1992), which have been shown to be much more sensitive than the standard solid-phase toxicity tests (Carr and Chapman, 1992). Pore water is predicted by the equilibrium-partitioning theory to be the controlling exposure medium in the toxicity of sediments to infaunal organisms (Adams et al., 1985; Di Toro, 1988). *In situ* effects were evaluated by an examination of the benthic infaunal community structure. The results of this study provide an overview of sediment quality in Galveston Bay and allow the areas most seriously impacted by contaminants to be identified.

MATERIALS AND METHODS

Site Selection

Sampling designs and results of previous sediment survey and benthic community studies in Galveston Bay, were critically reviewed and compared with recent scientific literature to evaluate the cost effectiveness and statistical power of such designs. There are two basic approaches that could have been employed, each having their advantages and disadvantages. In order to be able to statistically compare parameters among sites, a minimum of three stations per site must be sampled in order to provide some measure of within site variability. When working within a fixed budget, and therefore a fixed number of stations, this approach limits the number of sites that can be included in the survey. The other alternative strategy was to maximize the number of sites included in the survey by eliminating replication and collecting only one composite sample per site. The disadvantage of this approach is that no

statistical comparisons among sites can be made, although the sites can be ranked for the various parameters.

We recommended to the GBNEP Scientific and Technical Advisory Committee (STAC) that a combination of these two approaches would best serve the current needs of GBNEP. We recommended establishing five benchmark sites (Table 1 and Figure 1) which would include three stations per site. These stations would be arranged in an equilateral triangle, approximately 200 m apart, similar to the strategy that is currently used in NOAA's National Status and Trends Program. These benchmark stations were selected on the basis of existing historical information, their representativeness of distinct areas within the Galveston Bay system, similarities in sediment texture, and the fact that their location was outside of any major localized contaminant sources. The data generated from these replicated benchmark sites provides a statistically sound characterization of these sites for comparisons among sites and with future studies designed for comparison purposes. This recommended approach for selecting the sites was unanimously approved by a vote of the GBNEP STAC. The remaining 11 stations were located at 11 different sites based on historical data (Table 1 and Figure 1), and were not replicated. An effort has been made to include as large an areal coverage as possible and to concentrate on depositional sediments where possible. Information generated from these 11 unreplicated sites could be used to identify sites that should be examined more closely in future studies. The USFWS also conducted a SQT survey of specific sites of concern located near known sources of contamination in conjunction with the GBNEP study (Table 1 and Figure 1). These data are included in this report and provide SQT information for an additional eight sites that would not have been included otherwise.

Field Measurements and Sample Collection

Sediment samples were collected by researchers from the USFWS and the University of Houston-Clear Lake using USFWS research vessels. The same field measurements (water temperature, salinity, dissolved oxygen, depth, and latitude/longitude) and sample collections were made at all stations included in this study. Each station was sampled on one occasion during October 7-12, 1991. A Global Positioning System (GPS) navigation receiver (Magellan NAV 1000 Plus®) was used to determine station locations. Composite homogenized sediment samples were subsampled for the analytical and toxicity testing components of the study to maximize statistical associations.

Sediment samples (6-10 cm deep) were collected with a four-inch diameter coring device equipped with a transparent PVC barrel to enable the depth and integrity of the core to be determined before it was included in the composite sample. The transparent PVC barrels are detachable thus allowing a clean barrel to be used at each sampling station. The PVC corer is equipped with a valve that closes when the sample is withdrawn and can be opened manually to release the sample from the corer. The corer was designed and built by the USFWS and has multiple attachments which allow sampling at water depths up to 5 meters. Sediment cores (8-10) were placed in a Kynar®-lined stainless steel pan and the composite sample (~5 liters) was homogenized with a Teflon® spatula. Sediment subsamples for chemical analyses were placed in glass I-Chem containers cleaned to EPA specifications (Protocol A which includes nitric acid and methylene chloride rinses) and kept on ice until

Table 1. Sampling sites for Galveston Bay Study

Site No.	Site Description	Reason for Selection
1B[1]	Morgan Point, GERG station A, between BEG stations #353 and #351	Contaminant body burden data available from GERG study and sediment chemical data available from BEG study
2B	Jack's Pocket, Houston Light & Power (HL&P) Trinity Bay Station, (the most southeasterly station on HL&P transect G closest to the Bureau of Economic Geology (BEG) station #462)	Excellent historical data base exists for benthic community structure and sediment chemical data available from Houston Lighting & Power (1980) and Bureau of Economic Geology (1985) studies
3B	Eagle Point, GERG station B, BEG station #104	Contaminant body burden data available from GERG study and sediment chemical data available from BEG study
4B	South of Hannas Reef in East Bay, NOAA study station #17, near TWC site SMN #2439.0150, BEG station #116, and GERG station C, 500 m S.W. of tide gauge	Historical information for sediment organic and inorganic chemistry from NOAA study, benthic community structure information from TWC and BEG studies, and contaminant body burden data from GERG study
5B	West of Carancahua Reef in West Bay, BEG station #46, GERG station D	Benthic community structure data available from BEG study and contaminant body burden data from GERG study
6	Burnett Bay, near BEG station #13	Historical information on metals and benthic community structure data from BEG study available.
7	Cedar Bayou, 0.25 mi due west of BEG station #390	High sediment chromium concentrations observed in BEG study
8	Umbrella Point, BEG station #358 located near Schropp's station #8	Historical data for chemical and benthic analyses from HL&P, BEG, and Schropp (1979) Sun Oil Co. studies

Table 1. Continued

Site No.	Site Description	Reason for Selection
9	100 m due south of Fishers Reef C-2 separator platform in Trinity Bay, near BEG station #401	Benthic and chemical data available from Armstrong et al. (1979) and BEG studies
10	North of Smith Point, BEG station #265	High sediment copper and lead concentrations observed in BEG study
11	South of Lake Surprise in East Bay, 0.23 mi due west of BEG station #200	High sediment strontium concentrations observed in BEG study
12	0.5 mi north of March Point in East Bay, BEG station #208	Depositional zone representative of eastern portion of East Bay; metals and benthic community structure information available for nearby BEG stations
13	Kemah Flats, ~250 m due west of channel marker #2, BEG station #249	Adjacent to non-point source urban runoff from Clear Lake area; benthic community structure information available from BEG study
14	Southeast of Texas City, BEG station #8	Depositional zone adjacent to non-point source urban and industrial runoff; benthic community structure and metals data available for nearby BEG stations
15	Jones Bay, 0.55 mi north of BEG station #13	Historical information on metals and benthic community structure for nearby BEG stations available
16	Chocolate Bay, 0.04 mi north of BEG station #72	Historical information on metals and benthic community structure available from BEG study
1S[2]	~100 m Southwest of Alexander Island, near BEG station #7	Adjacent to a confined dredged material disposal area
2S	Black Duck Bay	Adjacent to industrial treatment lagoons

Table 1. Continued

Site No.	Site Description	Reason for Selection
3S	~200 m north of Lost Lake Island	Adjacent to a confined dredged material disposal area
4S	Southeast side of Atkinson Island, near BEG station #342	Adjacent to a semi-confined dredged material disposal area
5S	~100 m south of F-2 produced water separator platform	Adjacent to an active produced water separator platform
6S	~150 m south of F-1 produced water separator platform	Adjacent to a produced water separator platform
7S	Swan Lake	Adjacent to non-point source urban and industrial runoff
8S	Dollar Bay	Adjacent to non-point source urban and industrial runoff

[1]B=Benchmark sites with three stations per site

[2]S=U.S. Fish and Wildlife Service sites

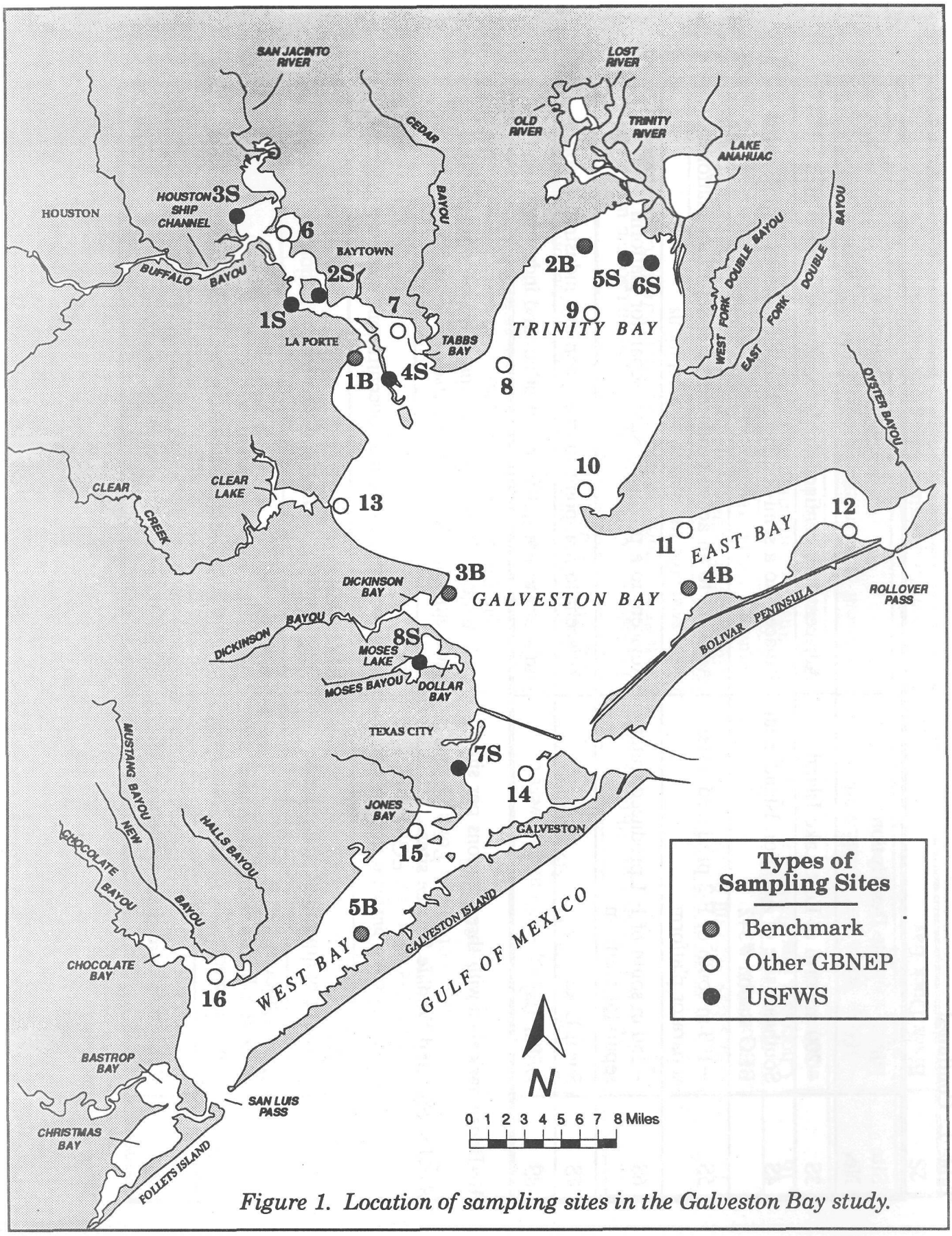

Figure 1. Location of sampling sites in the Galveston Bay study.

they were frozen. The sediment subsamples for toxicity testing were placed in acid-washed high density polyethylene containers and held on ice or refrigerated until processed. Sediment pore water was extracted from the sediment samples within five days of sample collection; the pore water was stored frozen until just prior to testing.

Separate samples were collected simultaneously for the benthic community structure analyses because these samples must be collected undisturbed. These samples were collected with a two-inch diameter coring device with five replicates per station. Sediment cores were individually examined for the presence of a thin layer of light brown sediment on the surface, evidence that the sample had been collected evenly and without disturbing the sediment. Only undisturbed samples were used. A two-inch corer was used for all benthic community sampling in the Houston Lighting and Power Co. Trinity Bay study (HL&P, 1980), and the results from replicate two-inch cores compares well with other larger benthic samplers (Baker, et al., 1977). Each replicate was placed in a polyethylene container and the sample fixed with buffered formalin. At the laboratory, these samples were sieved and processed.

Sediment Chemistry

The sediment chemistry and grain size analyses were conducted by the Texas A&M University Geochemical and Environmental Research Group (GERG) through the contract laboratory program with the USFWS Patuxent Analytical Control Facility (PACF). The sediment samples were stored frozen until they were shipped on dry ice by overnight express mail to GERG.

For the organic analyses, the sediment samples were freeze-dried and extracted in a Soxhlet extraction apparatus. The freeze-dried sediment samples were homogenized and a 10-gram sample was weighed into an extraction thimble. Surrogate standards and methylene chloride were added and the samples extracted for 12 hours. The extracts were treated with copper to remove sulfur and were purified by silica/alumina column chromatography (MacLeod et al., 1985; Brooks et al., 1989) to isolate the aromatic/pesticide/PCB fraction. The quantitative analyses were performed by capillary gas chromatography (CGC) with electron capture detector for pesticides and PCBs, and a mass spectrometer detector in the SIM mode for aromatic hydrocarbons (Wade et al., 1988).

For the trace metal scan samples, the sediments were prepared using a strong acid leach (2 g dry weight in boiling aqua regia for 2 hours). Aluminum, boron, beryllium, chromium, copper, iron, magnesium, manganese, nickel, strontium, vanadium, and zinc were determined using an Applied Research Laboratories SpectraSpan VI direct coupled argon plasma (DCP) emission spectroscopy. Arsenic, cadmium, lead, and selenium were determined by graphite furnace atomic absorption spectrophotometry (AAS) using a Perkin Elmer 3030. Samples were digested separately for mercury analysis using EPA method 245.5 (sulfuric/nitric acid digestion) and analyzed by cold vapor AAS. Acid volatile sulfides (AVS) and simultaneously extracted metals (SEM) analysis for Cu, Ni, and Zn were analyzed in accordance with the draft EPA method (1991) using graphite furnace AAS. Total organic carbon (TOC) was measured using a Coulometer TOC analyzer and sediment grain size analyses were also performed.

Sediment Toxicity Testing

Solid-phase Tests

The toxicity of the sediments from all stations in this study was determined using a static 10-day solid-phase test (ASTM, 1990) with the corophiid amphipod *Grandidierella japonica*. Sediment samples were held on ice or refrigerated during transport and storage. The test was started within two weeks of sample collection. The test was conducted under static conditions in environmental chambers at 20°C, 30±1 $^{o}/_{oo}$ salinity and a 24-hour light cycle to encourage maximum contact time with the sediments. One liter I-Chem® glass jars with Teflon®-lined lids were used for exposure chambers. Gentle aeration was supplied to each exposure chamber to ensure that dissolved oxygen levels remained above 90% saturation. Water quality measurements (temperature, dissolved oxygen, salinity, pH, ammonia) were made on each exposure chamber at the beginning and termination of the 10-day exposure.

G. japonica were cultured in-house for a minimum of three generations. The animals were retrieved from the culture aquaria with a fine mesh net on the day of the start of the test. Only non-gravid animals larger than 2.5 mm were included. The animals were randomly transferred one-by-one to 20 ml vials and the vials randomly selected for addition to the test chambers. There were four replicates per treatment and five animals per replicate. A reference sediment (Mustang Island dune sand which is 100% fine sand and is used to culture the amphipods) was also tested in conjunction with the test sediments.

Porewater Tests

In addition to the solid-phase tests that were performed as part of the GBNEP contract, the Fish and Wildlife Service funded and conducted (i.e., no EPA funds were used) an additional series of sediment toxicity tests using the porewater approach. The toxicity of sediment pore water and positive control samples (sodium dodecyl sulfate) was determined using two different tests with the sea urchin *Arbacia punctulata* (Weber et al., 1988; Oshida et al., 1981). These types of tests have been used to assess the toxicity of marine and estuarine sediments previously (Carr et al., 1989; Long et al., 1990; Carr and Chapman, 1992). The toxicity tests conducted in this study were the fertilization test and the morphological development assay. These tests utilize different end points, thereby providing different information regarding the mode of toxicity of the sediment-associated contaminants.

The sediment samples were held on ice or refrigerated (4°C) until the pore water was extracted. All pore water was extracted within five days after sample collection. The pore water was extracted using a pressurized squeeze extraction device with polyester filters (8 μm pore size). This filter has been found to be the optimum filter material and pore size for extracting pore water (Carr and Chapman, in review). The pore water was frozen immediately after extraction and stored frozen until the day before the start of the toxicity test, at which time the samples were thawed in a tepid water bath. Water quality measurements (dissolved oxygen, pH, hydrogen sulfide, temperature and ammonia) were then made and the salinity of the samples adjusted to 30±1 $^{o}/_{oo}$, if necessary, using milli-Q deionized water or hypersaline brine. The samples were then centrifuged in polycarbonate

bottles, the supernatant decanted and stored refrigerated overnight; samples were returned to 20°C before the start of the tests.

A reference pore water sample collected from Redfish Bay, which was handled identically to the Galveston Bay samples, was included with each series of toxicity tests as a negative control. This site is far removed from any known sources of contamination and has been used previously as a reference site (Carr and Chapman, in review). In addition, a dilution series test with sodium dodecyl sulfate (SDS) was also conducted with each fertilization test series as a positive control and the EC_{50} determined in order to maintain a record of gamete viability and test acceptability. Millipore filtered (0.45 μm) seawater and reconstituted brine (30°/$_{oo}$ salinity) controls were included with every series of tests.

Preliminary studies have indicated that freezing/thawing has no effect on the toxicity of porewater samples (Carr et al., 1989; Carr and Chapman, in review). The USFWS conducted (i.e., no EPA funds were used) a series of comparison studies to evaluate further the effects of freezing and thawing with both squeeze extracted (PVC extractor with polyester filters as described above) and centrifuged samples. Aliquots of pore water obtained from sediments processed within 24 hours of collection by the two extraction methods (using subsamples from the same sediments) were frozen in a conventional freezer overnight while the remainder of the samples were held refrigerated. On the following day the frozen aliquots were thawed in a refrigerator and then all the treatments were tested together using the two sea urchin assays described previously.

Statistical comparisons among treatments were made using analysis of variance (ANOVA) and Ryan's Q test (Day and Quinn, 1989) on arc sine square root transformed data with the aid of SAS (SAS, 1985). Dunnett's one-tailed t-test, which controls the experimentwise error rate for multiple comparisons, was used to make pair-wise t-test comparisons with the arc sine square root transformed data. The trimmed Spearman-Karber method with Abbott's correction was used to calculate EC_{50} values for reference toxicant and dilution series tests (Hamilton et al. 1977).

Benthic Community Structure Analysis

The benthic community structure analyses were conducted by the University of Houston-Clear Lake (UHCL) under the direction of Dr. Cynthia L. Howard. Dr. James Baker of ENSR Corp., was the taxonomic consultant for polychaetes, oligochaetes, and crustaceans. The purposes of this component of the study were (1) to process, sort, identify and enumerate benthic macroinfauna in core sediment samples; (2) to provide phylogenetic listings of species, replicate count data, total numbers of individuals, major taxa and species, species diversity and related parameters for each station analyzed; and (3) to maintain an archive of species identified during the project.

Each sediment sample was washed through a No. 60 (0.25 mm) mesh sieve. All material remaining on the screen was washed into an appropriately sized plastic wide-mouth container, preserved with 10% buffered formalin, and stained with a mixture of 1 mg/L Eosin B and Sudan IV. To begin the benthic analysis, samples were rinsed out of the plastic containers into a #200 mesh sieve to remove the formalin. Material in the sieve was washed

back into the plastic container and represerved with ethanol. Each organism was identified to species or the lowest taxon possible (depending on the condition of the organism), counted, then gently removed from the tray and placed into a 1-dram vial containing ethanol. Any organisms that were not identified by the sorters were placed into 1-dram vials containing ethanol, labelled as described above and identified by the Task Leader or consultant. The following analyses of the macrobenthic data were completed: phylogenetic listing of species, replicate count data, species totals, total taxa, mean number of taxa per replicate, total number of individuals, species diversity, species richness, species evenness, and total for major species groups.

RESULTS

Field Measurements and Sample Collection

The site/station coordinates and water quality parameters at the time of sample collection are given in Appendix 1. The surface water salinity ranged from a low of 4 $^{o}/_{oo}$ at station 2B-3 to a high of 23 $^{o}/_{oo}$ at the three benchmark stations at site 5B. Dissolved oxygen concentrations were at normal levels at all sites. All stations were sampled within a six day period beginning October 7, 1991.

Chemical and Physical Analyses

Sediment moisture content, total organic carbon (TOC), and grain size are shown in Appendix 2. Although we attempted to select depositional sites for this study, there was considerable variability in sediment texture. TOC values ranged from a low of 0.20 % for station 3B-1 2.1 % for station 5B-2. The high TOC values found at site 5B are likely related to the abundance of organisms found at this site.

The concentration of chlorinated hydrocarbons were below detection limits (0.01 μg/g) for nearly all samples except for a few stations with detectable concentrations of PCBs (Appendix 3). The concentration of the pesticides, aldrin, dieldrin, endrin, and mirex were below detection limits for all stations (data not shown). Polycyclic aromatic hydrocarbons (PAHs) were elevated at a number of sites (i.e., exceeding background levels for other sites in this study for specific PAHs by a factor of 5 or more, see Appendix 4). Quality control data for the chlorinated hydrocarbons and PAHs are given in Appendices 5 and 6, respectively. The three sites adjacent to produced water separator platforms (sites 9, 5S and 6S) all had elevated PAH concentrations. The lower molecular weight and most readily degraded PAHs, that are indicative of recent petroleum contamination (e.g., naphthalenes and phenanthrenes), were also found at elevated levels at stations 1B-3, 2B-3, 3B-1, 3B-2, 3B-3, 6, 1S, and 3S. Site 3B is near the location of the *Apex* barge spill which occurred on July 28, 1990. The higher molecular weight PAHs (4 rings and above), which are indicative of chronic petroleum inputs, were most elevated at site 9 (the C-2 separator platform that has discharged produced water for 20^{+} years but was apparently inactive at the time the samples were taken) and site 5S (the F-2 separator platform which was actively discharging produced water (112 $^{o}/_{oo}$ salinity) at the time the sediment sample was taken).

The sediment metal concentrations for the metal scan are summarized in Appendix 7. The metal scan data were normalized to a reference element, aluminum, for each metal and compared with the expected ratios for these metals in estuaries following the procedures of Schropp et al. (1990). As stated in Shropp et al. (1990) "Aluminum is used as the reference element because (1) it is the second most abundant metal in the earth's crust, (2) proportions of metal to aluminum are relatively constant in the crust, and (3) aluminum concentrations are not likely to be significantly affected by anthropogenic aluminum sources". Based on data from uncontaminated estuaries from Florida, Georgia and South Carolina, regression equations and 95% confidence intervals have been calculated. Metal to aluminum ratios that are higher than the 95% confidence interval are indicative of anthropogenic contributions. A number of stations were found to have levels of zinc in excess of naturally occurring levels (i.e., the metal to aluminum ratios were higher than the 95% confidence interval values which have been determined for uncontaminated sediments)(Table 2). Some possible sources of zinc include sacrificial anodes, zinc paint on ships, fishing weights, street runoff, and pipe dope in drilling oil and gas wells contains zinc and lead. Elevated zinc levels were found at or near some of these same sites in the BEG (1985) study (McNiff et al., in press). The concentration of copper, chromium, and lead were also found to be elevated at site 7S.

Acid volatile sulfides (AVS) are now thought to be the primary ligands controlling the bioavailability of divalent metals in most sediments (Di Toro et al., 1990). Ratios of metal to AVS exceeding one may be indicative of potential toxicity. The sediment AVS concentrations ranged from 0.649 to 29.9 μmoles/g (Appendix 8). The associated simultaneously extracted metal (SEM) values for copper, nickel, and zinc, which were extracted during the same digestion procedure used for the AVS determination are also given (Appendix 8). Quality control data for both the trace metal scans and SEM analyses are given in Appendix 9. SEM concentrations are thought to provide a better (more conservative) estimate of the metals associated with the AVS pool. The bulk sediment zinc/AVS ratio was greater than one for all three stations at site 4B and at site 14 (Table 2). The bulk sediment metal/AVS ratio was well below one for all other metals and stations investigated in this study. Likewise the SEM/AVS ratios were well below one for all stations.

Sediment Toxicity Testing

Solid-phase Tests

No significant differences in toxicity were observed among any of the stations in the 10-day static amphipod test with *Grandidierella japonica* using Ryan's Q test for multiple comparisons and Dunnett's t-test in which all treatments were compared with the reference (Appendix 10). The sensitivity of this test may have been reduced because fewer animals per treatment (5 rather than 10 to 20) were used due to a shortage of animals available from the laboratory culture. This shortage of animals resulted in larger animals than usual being included in the test (no gravid animals were used) which we believe was responsible for the higher than normal mortality in the reference treatment due to natural attrition, which also may have decreased the sensitivity of the test. The water quality measurements were all within acceptable ranges during the test (Appendices 11-13).

Table 2. Stations at which metals were found to be in excess of naturally occurring levels, and their associated acid volatile sulfide (AVS) normalized bulk sediment values.

Station #	Site Description	Metal	Bulk Sed. Conc.[1] (μg/g)	Bulk Sed. AVS Conc.[2] (μmoles/g)	AVS Norm.[3]	Amount Above 95% Interval (μg/g)
1B-1	Morgan Point	Zn	42.2	3.76	0.17	1.36
4B-1	Hannas Reef	Zn	70.2	0.86	1.25	NC[4]
4B-2		Zn	125	1.56	1.23	46.03
4B-3		Zn	70.2	0.65	1.65	NC[4]
6 **	Burnett Bay	Zn	158	18.4	0.13	70.59
7 *	Cedar Bayou	Zn	74.8	17.3	0.07	2.48
11 **	South of Lake Surprise	Zn	81.7	7.42	0.17	2.54
13	Kemah Flats	Zn	67.5	8.47	0.12	0.67
14	Texas City	Zn	65.5	0.65	1.54	8.08
15	Jones Bay	Zn	62.5	8.33	0.11	5.06
1S *	Alexander Island	Zn	55.9	8.15	0.10	2.33
2S **	Black Duck Bay	Zn	197	33.2	0.09	116.17
7S	Swan Lake	Cr	81.5	13.0	0.12	5.23
		Cu	37.7		0.05	12.13
		Pb	147		0.05	126.74
		Zn	165		0.19	114.05

[1] Metal concentration in bulk sediment on dry wt. basis as measured by DCP analysis.
[2] AVS concentration in bulk sediment on dry wt. basis.
[3] Normalization of bulk sediment concentration on a μmoles/g basis.
[4] No excess calculated for treatment using metal/aluminum ratio analysis.
* Toxic in sea urchin porewater morphological development test.
** Toxic in both porewater assays.

Porewater Tests

The porewater toxicity tests with the sea urchin *Arbacia punctulata* were conducted independently by the USFWS (i.e., no EPA funds were expended). In contrast to the results from the amphipod solid-phase test, highly significant toxicity was observed at a number of stations for both the fertilization and morphological development assays (Appendix 14). The morphological development assay proved to be the more sensitive of the two porewater tests with statistically and biologically significant toxicity observed at 12 of the 34 stations investigated while only six stations exhibited statistically and biologically significant toxicity in the fertilization test (Figure 2). The Environmental Protection Agency and the Army Corps of Engineers (Holland, 1990; EPA/COE, 1991) have recommended that a second criteria of biological significance (defined as a response of $\leq$ 80% of the reference value) be used in sediment toxicity test analyses as the statistical power of tests with high levels of precision (e.g., the sea urchin fertilization and morphological development assays) may produce "hits" merely due to test precision. All of the stations adjacent to dredge material disposal islands (1S, 3S and 4S) and produced water separators (5S, 6S and 9) were significantly toxic in the sea urchin morphological development assay. In addition, the stations at Burnett Bay (6), Black Duck Bay (2S), Cedar Bayou (7), Smith Point (10), south of Lake Surprise (11), and Chocolate Bay (16) were also significantly toxic (Figure 2). None of the stations at the five benchmark stations exhibited significant toxicity for either of the porewater tests.

The low percent fertilization for the reconstituted brine sample (1-REC) was due to contaminated deionized water (we suspect that the chloramine concentrations were higher than usual in the feed water to the deionizer) that was used to dilute the hypersaline brine solution. The contaminated deionized water was not added to any samples other than the hypersaline brine solution because all porewater samples were < 30 $^{o}/_{oo}$ salinity. The hypersaline brine solution was added to all of the samples, in varying amounts, to adjust the salinity of the pore water to 30 $^{o}/_{oo}$ prior to the start of the test. The bioavailability of some metals (e.g., cadmium and silver) is expected to be reduced by increasing the salinity due to complexation with chlorides (Engel et al., 1981). Increasing salinity is not expected to appreciably affect the bioavailability of non-polar organic compounds. The results of porewater toxicity tests in which the salinity has been increased (in order to conduct the sea urchin assay) may actually underestimate the toxicity of the samples.

The reference toxicant (SDS) EC_{50} values were 2.27 and 7.32 mg/L for the first and second porewater fertilization test series and >5 and >20 mg/L for the morphological development tests. These values are within the normal range of EC_{50}s observed for these tests and verify the acceptability of the sea urchin gametes used in the porewater tests.

Water quality data for the fertilization and morphological development tests are shown in Appendices 15 and 16, respectively. The hydrogen sulfide concentration was at or below the detection limit (0.01 μg/L) for all but one sample (1B-3, sperm cell test). The unionized ammonia nitrogen concentrations ranged from 2.4 to 360.3 μg/L, which is below the no-observed-effect concentration (NOEC) for these sea urchin assays (USFWS, unpublished data).

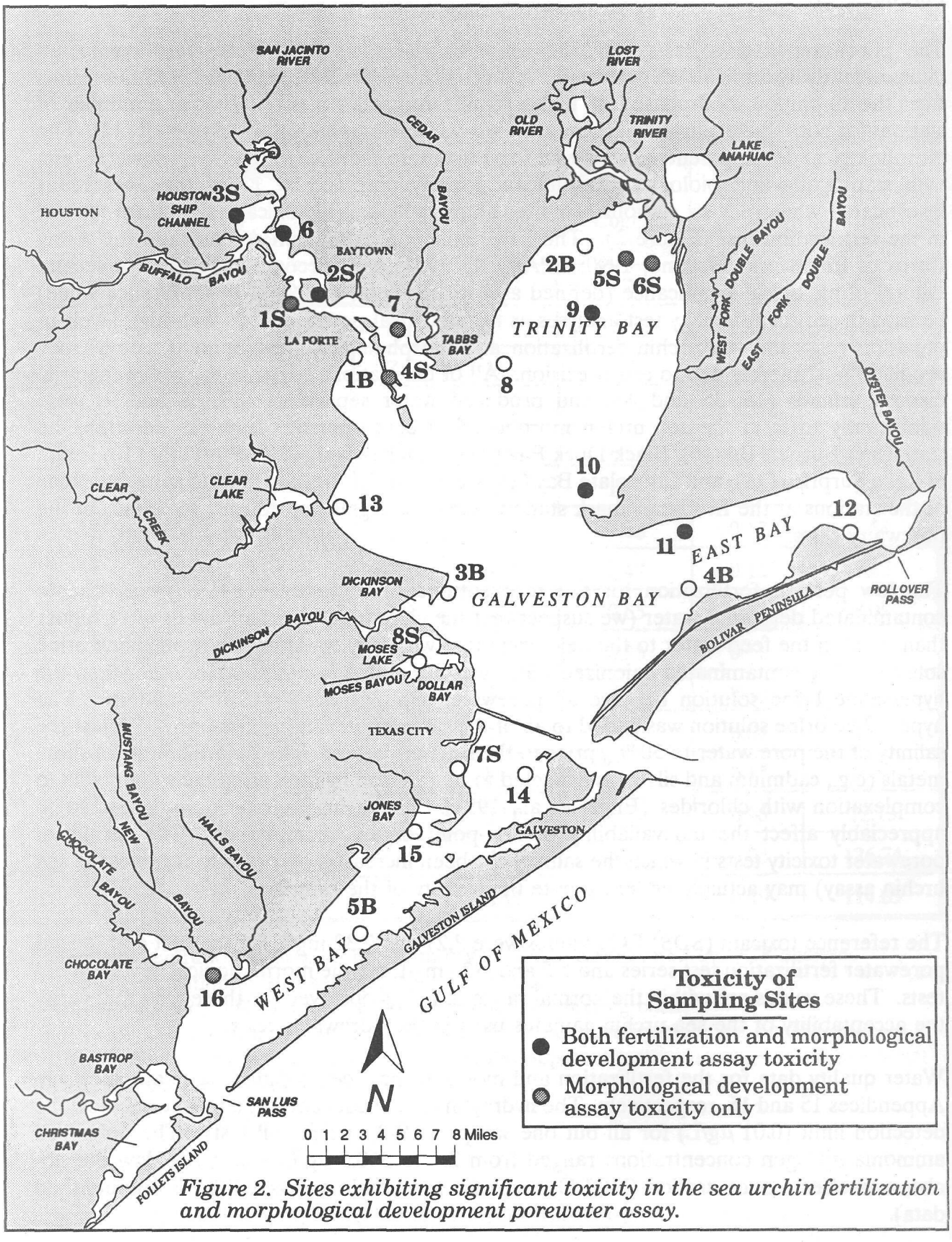

Figure 2. Sites exhibiting significant toxicity in the sea urchin fertilization and morphological development porewater assay.

Porewater Storage and Extraction Method Comparison Studies

The effect of freezing/thawing on the toxicity of pore water from nine different sites with varying degrees and types of contamination was assessed for both centrifuged (30 min at 4,500 rpm) and squeezed samples (Appendix 17). For three of the nine centrifuged samples, freezing/thawing produced a statistically significant increase in the toxicity of the sample. In only one of the nine squeezed porewater samples was a statistically significant increase observed (this was the same sample (#9) which showed the greatest increase in toxicity between the fresh vs. frozen centrifuged sample). There was a good correlation between fresh centrifuged vs. fresh squeezed samples with only sample #1 showing a statistically significant difference in toxicity. Using the biological significance criteria ($\leq$ 80% of the reference response) described in the previous section, only the centrifuged fresh vs. frozen samples #8 and #9 were significantly different.

We suspect that the increase in toxicity observed after freezing/thawing, which was most apparent with the centrifuged sample, is due to the considerably higher amount of particulate material remaining in the centrifuged supernatant as compared with the squeezed sample (Carr and Chapman, in review). This fine suspended material cannot be reduced substantially by increasing the duration of the centrifugation. We hypothesize that this is due to the electrostatic repulsion of similarly charged colloidal particles (Everett, 1988). When the supernatant (or pore water obtained from squeezing or suction) is centrifuged, the amount of suspended material retained on a 0.45μm filter was negligible for samples with a wide variety of sediment textures (Carr and Chapman, in review). When pore water containing particle-sorbed contaminants is frozen, apparently some contaminants can be released back into solution upon thawing, which would explain the increased toxicity observed predominantly in some of the centrifuged porewater samples. Based on the results of these storage and extraction experiments, we feel that the porewater storage and extraction methods used in the sediment quality survey are valid and unlikely to have introduced any significant artifactual responses.

Benthic Community Structure Analysis

A phylogenetic listing of all the macrobenthic invertebrates identified in this study are shown in Appendix 18. The "unknown" polychaetes were incomplete specimens with cephalic regions intact. These speciments for which positive identifications were not possible were included in the abundance calculations but not the species richness and diversity indices. Species richness and abundance were highest at site 5B, near Carancahua Reef in West Bay (Appendix 19). Annelids (polychaetes and oligochaetes) were the predominant organisms at most sites (Appendix 20). The most depauperate stations were in Burnett Bay (6), Alexander Island (1S), and Black Duck Bay (2S) (Figures 3 and 4). Amphipods were only observed at nine of the 34 stations and were only present at low abundance. Meiobenthic species (animals < 0.5 mm) were absent or few in numbers and diversity at all but a few sites (e.g., Caranchaua Reef). Station rankings for richness, diversity, evenness, abundance, and for the major species groups are shown in Appendices 21 and 22. A complete phylogenetic listing of abundance for each station is provided in Appendix 23.

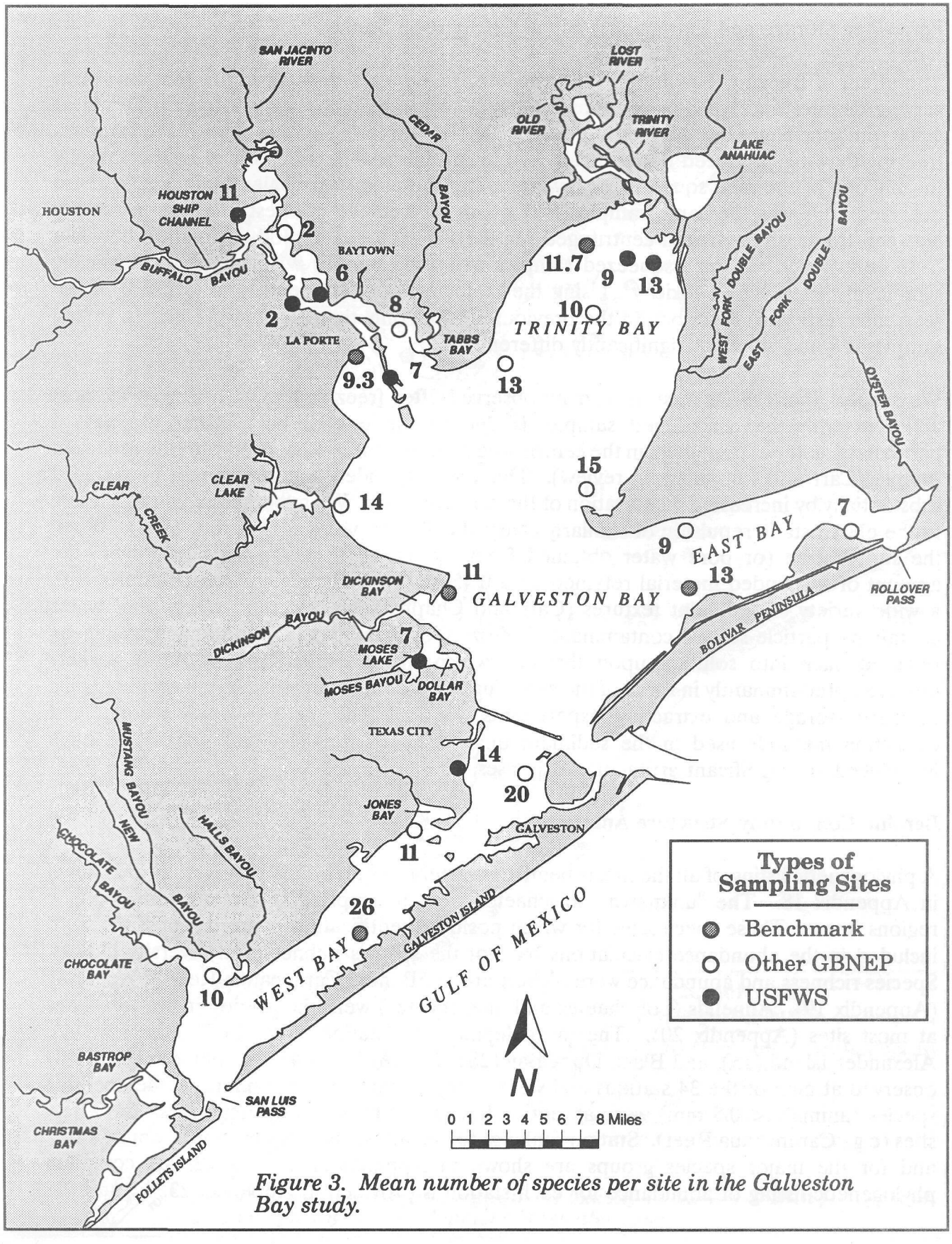

Figure 3. Mean number of species per site in the Galveston Bay study.

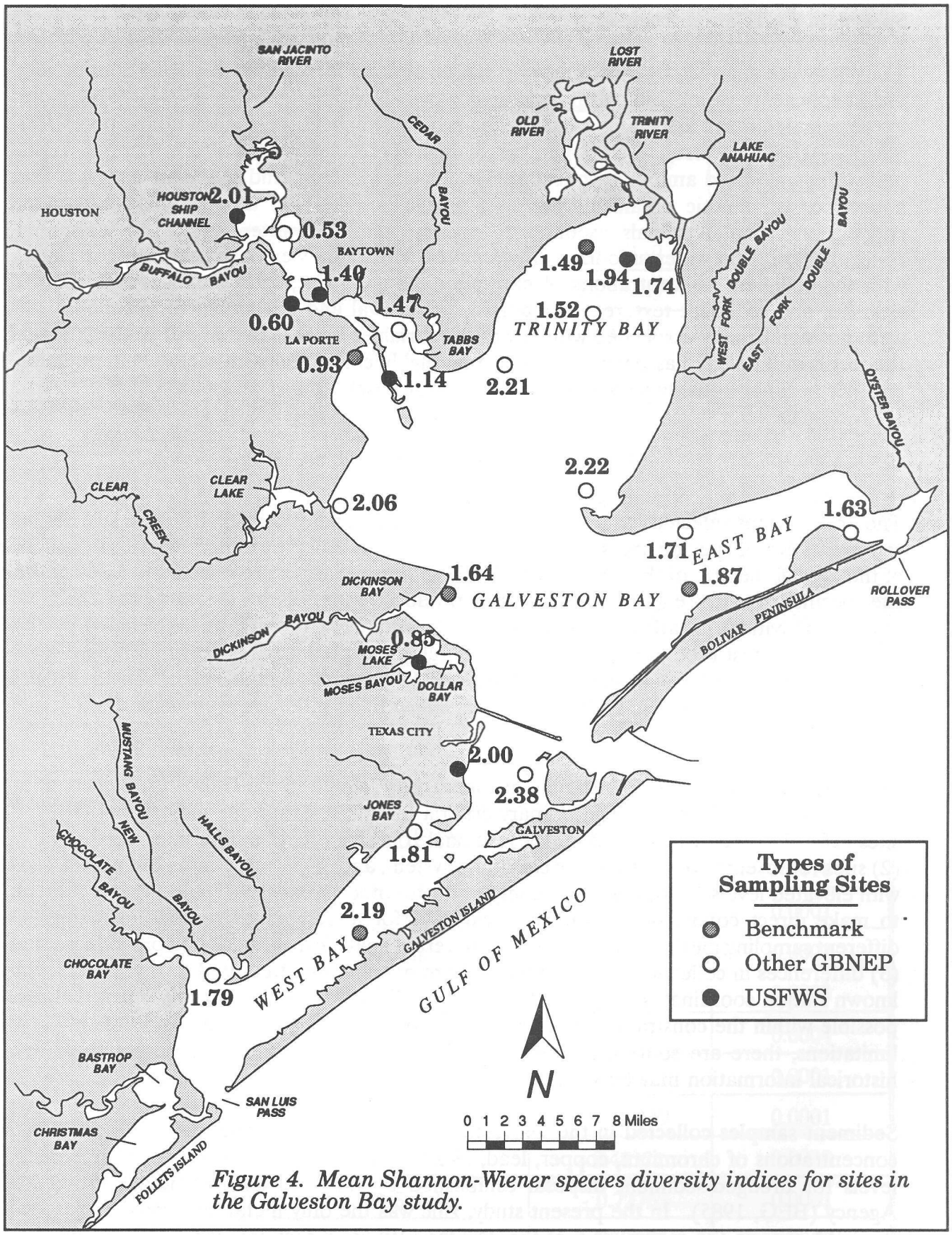

Figure 4. Mean Shannon-Wiener species diversity indices for sites in the Galveston Bay study.

Spearman Correlation Analysis

The data were analyzed by Spearman correlation analysis for significant associations among the chemical, physical, biological, and toxicological parameters (Table 3). The more interesting significant associations include the relationship between the sea urchin porewater morphological test results and a number of high molecular weight PAHs, copper, AVS, porewater unionized ammonia concentration, species richness, and sediment texture. The abundance of benthic organisms was also inversely correlated with arsenic, chromium, copper, zinc, total metals, sediment texture, and porewater unionized ammonia concentration. Porewater unionized ammonia concentrations were also positively correlated with the sediment concentrations of chromium, copper, zinc, total metals, AVS, species richness, and sediment texture. The amphipod survival and sea urchin fertilization assay were not significantly associated with any of the variables investigated in this study although the fertilization assay has been observed to be highly correlated with metal contamination in other sediment quality surveys (Carr et al., in review).

DISCUSSION

The original sites selected for this study (1B through 5B and 6 through 16) were selected as representative of specific areas within the Galveston Bay complex to provide an assessment of the overall "health" of the system. The USFWS sites (1S-8S) were selected to assess more site specific concerns (e.g., dredged material, produced water discharges, and petrochemical industry activities). Statistically and biologically significant toxicity, as determined by the most sensitive test used in this study (the sea urchin morphological development porewater assay), was observed at 6 of the 8 USFWS sites (1S through 6S) and at 6 of the 16 original sites (6, 7, 9, 10, 11, 16). No toxicity was observed with the solid-phase amphipod test at any of the sites.

The purpose of reviewing all available historical data related to the Galveston Bay complex was to aid in site selection. The primary criteria for site selection for the original GBNEP sites (1B-5B and 6-16) included (1) depositional zones to reflect a time integrated sample, (2) sites representative of specific areas of the system, and (3) in some cases but not all, sites with elevated levels of contaminants compared with other areas (see Table 1). It is difficult to make direct comparisons with historical data for a number of reasons including (1) different sampling methods and designs, (2) different analytical methods and procedures, and (3) differences in collection site location. Where possible we attempted to revisit sites with known station coordinates (e.g., stations in the BEG (1985) study), but this was not always possible within the constraints of other site selection criteria. With an awareness of these limitations, there are some instances where comparisons between the present study and historical information may be valid.

Sediment samples collected in the Galveston Bay system during the mid-1970s contained concentrations of chromium, copper, lead, nickel, and zinc exceeding proposed screening levels for dredged-sediment disposal established by the U.S. Environmental Protection Agency (BEG, 1985). In the present study, zinc was the only metal observed to exceeded the EPA criteria (75 mg/kg) at 6 of the 24 sites. Where direct comparisons with the

Table 3. Spearman correlation coefficients and associated probability values for highly significant associations ($p \leq 0.005$) among the physical, chemical, biological, and toxicological data from the Galveston Bay SQT study. A negative correlation coefficient indicates an inverse relationship between the variables.

Variables		Spearman Correlation Coefficient	Unadjusted Probability Value
Sea Urchin % Normal Development vs.	Fluoranthene	- 0.47110	0.0049
	Benzo(a)anthracene	- 0.52463	0.0014
	Chrysene	- 0.47748	0.0043
	Perylene	- 0.50844	0.0021
	Benzo(b)fluoranthene	- 0.47219	0.0048
	Acid Volatile Sulfide	- 0.59884	0.0002
	Copper	- 0.47841	0.0042
	Unionized Ammonia	- 0.65303	0.0001
	Species Richness	0.50544	0.0023
	% Sand	0.47857	0.0042
Benthic Abundance vs.	Arsenic	- 0.51005	0.0021
	Chromium	- 0.58698	0.0003
	Copper	- 0.50608	0.0023
	Zinc	- 0.58673	0.0003
	Total Metals	- 0.54099	0.0010
	Unionized Ammonia	- 0.51536	0.0018
	% Sand	0.64675	0.0001
Unionized Ammonia vs.	Chromium	0.59193	0.0002
	Copper	0.57981	0.0003
	Zinc	0.57121	0.0004
	Total Metals	0.60764	0.0001
	Acid Volatile Sulfide	0.67239	0.0001
	Species Richness	- 0.48321	0.0038
	% Sand	- 0.72763	0.0001
Species Richness vs.	Acid Volatile Sulfide	- 0.50119	0.0025

historical metals data are warranted, the present levels, in general, appear to be similar or less than the concentrations observed in the BEG (1985) study. The only historical PAH data available for comparison are from the study by Armstrong et al. (1979) in the vicinity of the C-2 separator platform in Trinity Bay. From the best estimate of total PAHs in the sediment near the separator platform, it appears that sediment PAHs are as high or higher now than in 1974 (Armstrong et al., 1979). Comparisons between the historical and present benthic community data are complicated by differences in sampling design and techniques. The meiobenthic community was absent or depauperate at all but a few sites (e.g., Carancahua Reef) in the present study but these smaller species were not included in the BEG (1985) study. There are no historical sediment toxicity data available for comparison.

Sites adjacent to the active produced water separators had high PAH concentrations and were highly toxic. A correlation between toxicity and proximity to a major produced water discharge in Tabbs Bay has recently been reported (Roach et al., 1992). It appears that the impact of chronic produced water discharges in shallow estuaries, such as Trinity Bay, is still evident for some time after the discharge is discontinued with a pronounced accumulation of the more refractory high molecular weight PAHs remaining.

The sediment chemistry data were compared with sediment quality "guidelines" that have been developed by Long and Morgan (1990) and MacDonald (1992). These guidelines were developed by using data available from previous studies in which biological effects were determined in conjunction with chemical measurements. Long and Morgan (1990) used the available screened data to calculate ER-L values (equivalent to the lower 10 percentile), which indicated the low end of the range of concentrations in which biological effects were observed or predicted. They also calculated ER-M values (equivalent to the 50 percentile point in the available screened data), above which effects were frequently or always observed or predicted among most species. MacDonald (1992) used a similar approach (termed the weight-of-evidence approach) in which the no-observed-effect limit (NOEL) and probable effects level (PEL) concentration, which is comparable to the ER-M value of Long and Morgan (1990), were calculated using an expanded data base. Another sediment quality guideline that has been used is the apparent effects threshold (AET) approach (Tetra Tech, 1986). This approach is also based on relationships between measured concentrations of a contaminant and observed biological effects, mainly on benthic organisms. The AET value is defined as the concentration of a contaminant in sediment above which significant ($\alpha \leq 0.05$) biological effects are always observed. All of these procedures utilize unnormalized bulk sediment chemistry data.

There were a number of sites in which the NOEL and ER-L values were exceeded for a variety of different contaminants (Table 4). Sites 5S and 9, near the F-2 and C-1 separator platforms, respectively, had sediment concentrations above the NOEL or ER-L values for a number of PAHs. Total PCB concentrations were above the NOEL for Black Duck Bay (2S). A number of sites had metal concentrations above the NOEL or ER-L. Burnett Bay (6) and Swan Lake (7S) exceeded the NOEL or ER-L for chromium, lead, and zinc with the lead level exceeding the ER-M at site 7S. Swan Lake received runoff from Tex-Tin superfund site and other industrial facilities prior to the construction of Texas City Hurricane Flood Protection Levee.

Table 4. No-observed-effect level (NOEL), probable effects level (PEL), effects range low and median (ER-L and ER-M, respectively), and the apparent effects threshold (AET) values for key contaminants and stations exceeding those values.

Contaminant	NOEL	PEL	ER-L	ER-M	AET	Stations exceeding NOEL or ER-L[1]
PCBs (μg/kg)	24	260	50	400	50	2S (50) Black Duck Bay
Polycyclic Aromatic Hydrocarbons (μg/kg)						
Acenaphthene	22	450	150	650	56	5S (70) F-2 separator platform
Acenaphthylene	< 35	500	-	-	44	7S (51) Swan Lake
Anthracene	85	740	960	960	13,000	-
Fluorene	18	460	35	640	3600	5S (54) F-2 separator platform
Naphthalene	130	1100	340	2100	160	-
Phenanthrene	140	1200	225	1380	170	9 (250) C-2 separator platform
Σ LMW PAHs[2]	250	2400	-	-	2100	5S (252) F-2 separator platform 9 (278) C-2 separator platform
Benz(a)-anthracene	160	1300	230	1600	5100	9 (341) C-2 separator platform
Benzo(a)pyrene	230	1700	400	2500	-	-
Chrysene	220	1700	400	2800	190	9 (288) C-2 separator platform
Dibenzo(a,h)-anthracene	31	320	60	260	1200	9 (32) C-2 separator platform
Fluoranthene	380	3200	600	3600	390	9 (706) C-2 separator platform
Pyrene	290	1900	350	2200	16,000	9 (586) C-2 separator platform
Σ HMW PAHs[3]	870	8500	-	-	-	9 (2152) C-2 separator platform
Trace Elements (mg/kg)						
As	8	64	33	85	64	7S (9) Swan Lake
Cd	1	7.5	5	9	7.5	-

Table 4. Continued

Contaminant	NOEL	PEL	ER-L	ER-M	AET	Stations exceeding NOEL or ER-L[1]
Cr	33	240	80	145	2600	6 (63) Burnett Bay 7 (40) Cedar Bayou 11 (37) South of Lake Surprise 2S (41) Black Duck Bay 5S (40) F-2 separator platform 7S (82) Swan Lake
Cu	28	170	70	390	310	-
Pb	21	160	35	110	150	4B-2 (26) Hannas Reef 5B-3 (22) Carancahua Reef 6 (35) Burnett Bay 2S (23) Black Duck Bay 7S (147) Swan Lake
Hg	0.1	1.4	0.15	1.3	1.3	6 (0.3) Burnett Bay 2S (.13) Black Duck Bay 3S (.14) Lost Lake Island
Ni	-	-	30	50	> 140	-
Zn	68	300	120	270	340	4B-1 (70), 4B-2 (125), 4B-3 (70) Hannas Reef 6 (158) Burnett Bay 7 (75) Cedar Bayou 9 (81) C-2 separator platform 11 (82) South of Lake Surprise 5S (78) F-2 separator platform 7S (165) Swan Lake

[1] Measured concentration in parentheses. Units for contaminants: PCBs and PAHs (μg/kg), trace elements (mg/kg)

[2] Sum of the following low molecular weight PAHs; acenaphthene, acenaphthylene, anthracene, fluorene, 2-methylnaphthalene, naphthalene and phenanthrene. These are the PAHs selected by Long and Morgan (1990) for this summary parameter.

[3] Sum of the following high molecular weight PAHs; benz(a)anthracene, benzo(a)pyrene, chrysene, dibenzo(a,h,)anthracene, fluoranthene and pyrene. These are the PAHs selected by Long and Morgan (1990) for this summary parameter.

The sediment chemistry, toxicity, and benthic richness and abundance data can be used to rank the stations for a particular parameter and calculate a rank sum for each station (Table 5). The PAH concentrations were TOC normalized as this provides a better estimate of bioavailability than unnormalized values for hydrophobic compounds. The sum of the metals of toxicological significance were used to estimate the potential impact of trace metal contamination. The metal concentrations were not normalized because factors other than AVS are known to significantly affect the bioavailability of certain metals. The Shannon-Wiener species diversity index was used in the ranking procedure because this index incorporates both measures of richness and abundance. Only the sea urchin morphological development porewater toxicity test data was used in the ranking procedure because this was the most sensitive test and therefore provided the most information concerning the relative toxicity among the stations. As with any ranking procedure, there is an element of subjectivity. We have not used a weighting factor for the different parameter but by separating the metals and PAHs into two separate categories, the chemical data is essentially being weighted twice as high as the other two components of the sediment quality triad. Bearing in mind these subjective qualifications, the ranking procedure provides a valid means of making comparisons among the stations.

The station with the lowest rank sum was the C-2 separator platform (9) which was highly toxic, had the third highest concentration of TOC normalized PAHs, relatively high metal concentrations and a relatively low species diversity index. The station with the second lowest rank sum was Burnett Bay (6) which was highly toxic, had the lowest species diversity index, and relatively high levels of metals. The other active produced water separator platform station (5S) ranked fifth overall. The stations at Black Duck Bay (2S) and Alexander (dredge material disposal) Island (1S), ranked fourth and third, respectively, overall.

The benchmark stations, where three stations were sampled per site, in general, had relatively high rank sums. Morgan Point (1B) had the lowest rank score of the benchmark stations due primarily to the low Shannon-Wiener diversity index but was followed closely by Eagle Point (near the site of a recent oil spill), which had relatively high PAH concentrations. The variability among stations at a particular site for most parameters was low with one notable exception being the relatively high PAH concentrations observed at station 1B-3 and 2B-3, as compared with the other stations at those sites.

Chapman (1990) has provided some guidance for interpreting the results the three components of the SQT collectively. It was necessary to make some subjective decisions concerning the chemical and benthic data interpretation. Significant toxicity was indicated by a statistically and biologically significant reduction, as compared with the reference samples, for any of the three tests conducted. Stations in which the NOEL of MacDonald (1992) were exceeded (see Table 4) were considered to have elevated levels of contaminants. For the benthic data, stations which had fewer than 10 species and less than 100 total individuals in the five replicate samples combined (see Appendix 19) were considered to be significantly altered. Using these criteria, there are five stations (Burnett Bay, Cedar Bayou,

Table 5. Station ranks and rank sum of benthic invertebrate species diversity (Shannon-Wiener), toxicity (sea urchin morphological development), TOC normalized PAHs and bulk sediment metal concentrations for Galveston Bay Study. The higher the ranking the less degraded the station relative to the other stations.

Station	Site Description	Ranks				Rank Sum	Relative Ranking
		Benthic Diversity	Toxicity	PAHs	Metals[1]		
1B-1	Morgan Point	5	21	10	27	63	13
1B-2		8	26	11	29	74	22
1B-3		3	24	6	24	57	10
2B-1	Jack's Pocket	9	26	12	34	81	27
2B-2		15	26	15	28	84	28
2B-3		13	21	2	31	67	17
3B-1	Eagle Point	22	24	1	33	80	25
3B-2		7	20	9	30	66	16
3B-3		18	16	5	32	71	21
4B-1	Hannas Reef	23	31	27	9	90	30
4B-2		21	31	28	4	84	28
4B-3		24	33	31	10	98	31
5B-1	Carancahua Reef	33	29	32	26	120	34
5B-2		21	33	34	23	111	32
5B-3		32	29	33	20	114	33
6	Burnett Bay	1	1	24	3	29	2
7	Cedar Bayou	11	1	29	7	48	7
8	Umbrella Point	30	15	22	12	79	23
9	C2 separator platform	12	1	3	6	22	1
10	Smith Point	31	1	16	17	65	15
11	South of Lake Surprise	16	1	30	5	52	8
12	March Point	14	14	23	16	67	17

Table 5. Continued

Station	Site Description	Ranks				Rank Sum	Relative Ranking
		Benthic Diversity	Toxicity	PAHs	Metals[1]		
13	Kemah Flats	29	13	26	11	79	23
14	Texas City	34	18	14	14	80	25
15	Jones Bay	20	18	14	15	67	17
16	Chocolate Bay	19	12	18	19	68	20
1S	Alexander Island	2	1	8	22	33	3
2S	Black Duck Bay	10	1	25	2	38	4
3S	Lost Lake Island	28	1	19	13	61	12
4S	Atkinson Island	6	1	17	21	45	6
5S	F2 separator platform	25	1	4	8	38	4
6S	F1 separator platform	17	1	21	25	64	14
7S	Swan Lake	27	21	7	1	56	9
8S	Dollar Bay	4	17	20	17	58	11

[1] Metals of toxicological significance (As, Cd, Cr, Cu, Hg, Ni, Pb, and Zn)

South of Lake Surprise, Black Duck Bay, and F-2 separator platform) with strong evidence of contaminant-induced degradation (Table 6). Fifteen of the 34 stations showed no evidence of contaminant-induced degradation based on these criteria. At eight additional sites (C-1 separator platform, Smith Point, March Point, Chocolate Bay, Eagle Point-station 2, Lost Lake Island, Atkinson Island, and the F-1 separator platform), the SQT data suggested that unmeasured chemicals or conditions were stressing the system.

Chemical and physical disturbances which disrupt natural benthic communities can result in sediments becoming anoxic and an increase in the porewater ammonia concentrations due to a lack of bioturbation. Although the concentration of hydrogen sulfide was below the detection limit of 0.01 mg/L for all but one of the porewater samples, there was a high degree of association between the sea urchin embryological development assay and AVS content of the sediments and the porewater ammonia concentrations. The total metals concentrations was highly positively associated with percent silt and porewater ammonia concentrations and negatively associated with benthic abundance. These result suggest that silty, contaminant-enriched sediments tend to have elevated AVS levels and porewater ammonia levels and low benthic abundance, which may be the cause or the result of the other factors. It is very difficult to determine cause and effects relationships between sediment toxicity or benthic community indices and other specific parameters because many of the parameters (e.g., sediment texture, contaminant concentrations, porewater ammonia and sulfide concentrations) tend to covary.

CONCLUSIONS

From the historical information that is available and warrants direct comparison with the present study, it appears that contaminant concentrations in the Galveston Bay system, in general, have not increased since the mid-1970s and, for some metals, may have declined. There are certain areas, however, that appear to have been impacted by anthropogenic influences. In general, the benthic communities at the sites included in this study, with the possible exception of the Carancahua Reef site, were comprised of contaminant-resistant, euryhaline, opportunistic species which are indicative of a stressed environment. The dominant species (e.g., *Mediomastus, Capitella,* and *Steblospio*) are early colonizer species that often thrive in habitats that are chronically stressed by chemical or physical insults (Pearson and Rosenberg, 1976). The meiobenthic community was absent or depauperate at all but a few sites (e.g., Carancahua Reef) which may indicate chemical or physical impacts or decreased nutrients entering the system. While the dramatic salinity fluctuations which accompany the nutrient nourishing flow of freshwater into estuaries undoubtedly is in part responsible for the limited species diversity observed in the Galveston Bay system, other insults related to contaminants and physical disturbances associated with altered circulation, and sediment resuspension or nutrient declines, may contribute additional stresses to this naturally stressful environment.

Five of the 24 sites investigated exhibited strong evidence of contaminant-induced degradation (Burnett Bay, Cedar Bayou, South of Lake Surprise, Black Duck Bay, and F-2 separator platform). Fifteen of the 34 stations showed no evidence of contaminant-induced degradation. At eight additional sites (C-1 separator platform, Smith Point, March Point,

Table 6. **Summary of Sediment Quality Triad data.[1] A plus for chemistry indicates a concentration of a contaminant exceeds the NOEC (Table 4). A plus for toxicity indicates a significant decrease in normal development in the sea urchin porewater morphological development test (Figure 2). A plus for benthos indicates species richness < 10 and total abundance < 100 (Appendix 19).**

Chemistry	Toxicity	Benthos	Sites (Station Number)	Possible Conclusions
+	+	+	Burnett Bay (6), Cedar Bayou (7), South of Lake Surprise (11), Black Duck Bay (2S), F-2 Separator Platform (5S)	Evidence of contaminant-induced degradation
-	-	-	Morgan Point (1B-1,1B-2, 1B-3), Jacks Pocket (2B-1, 2B-2, 2B-3), Eagle Point (3B-1, 3B-3), Carancahua Reef (5B-1, 5B-2), Umbrella Point (8), Kemah Flats (13), Texas City (14), Jones Bay (15), Dollar Bay (8S)	No evidence of contaminant-induced degradation
+	-	-	Hannas Reef (4B-1, 4B-2, 4B-3), Carancahua Reef (5B-3), Swan Lake (7S)	Contaminants are not bioavailable
-	+	-	Smith Point (10), Chocolate Bay (16), Atkinson Island (4S), F-1 Separator Platform (6S)	Unmeasured chemicals or conditions exist with the potential to cause degradation
-	-	+	Eagle Point (3B-2), March Point (12)	Benthic response not due to contaminants
+	+	-	C-2 Separator Platform (9), Lost Lake Island (3S)	Contaminants may be stressing the system
-	+	+	Alexander Island (1S)	Unmeasured contaminants are causing degradation
+	-	+	None	Contaminants are not bioavailable or benthic response not due to chemistry

[1] Adapted from Chapman et al. (1990)

Chocolate Bay, Eagle Point-station 2, Lost Lake Island, Atkinson Island, and the F-1 separator platform), the Sediment Quality Triad (SQT) data suggested that unmeasured chemicals or conditions were stressing the system.

Sediment toxicity and concentrations of a variety of contaminants were observed at various locations during this broad scale survey. These "flags" should be interpreted as warning signs that contaminant problems may exist. This study has served to identify some specific areas of concern where more comprehensive monitoring needs to be conducted. Only with more extensive monitoring, remedial activities (where appropriate) and environmentally sound regulatory and management decisions, can the "health" of the Galveston Bay system be expected to improve in the future.