BENTHIC MACROFAUNAL COMMUNITY STRUCTURE IN OCEAN DREDGED MATERIAL DISPOSAL SITES IN LOUISIANA: PRELIMINARY ANALYSIS

by

David A. Flemer, James Patrick, Jr.,
James R. Clark[1]
Ecotoxicology Branch
U.S. Environmental Protection Agency
Environmental Research Laboratory
Gulf Breeze, FL 32561

Barbara F. Ruth[2]
Technical Resources, Inc.
Environmental Research Laboratory
U.S. Environmental Protection Agency
Gulf Breeze, FL 32561

Charles M. Bundrick
Institute for Statistical and
Mathematical Modeling
University of West Florida
Pensacola, FL 32514

Gary Gaston
University of Mississippi
Department of Biology
Shoemaker Hall
University, MS 38677

Submitted to:

Susan McKinney
U.S. Environmental Protection Agency
Region VI
Dallas, TX 75202

Final Report: November 1994

Present Affiliation:

[1]Dr. James R. Clark
Environmental Toxicology Division
Exxon Biomedical Sciences, Inc.
Mettlers Road, CN2350
East Millstone, NJ 08875-2350

[2]Barbara F. Ruth
Florida Department of Environmental Protection
Northwest District
760 Governmental Center
Pensacola, FL 32501

EXECUTIVE SUMMARY

An analysis of the benthic macrofaunal community structure following dredged material disposal was conducted during October 1991 at three dredged material locations in the northern Gulf of Mexico. Dredged channels and adjacent reference and disposal sites were located on the Gulf of Mexico-side of the passes of the Freshwater Bayou, Mermentau River, and Atchafalaya River systems on the Louisianian coast. Dredged channels and associated reference and disposal sites were oriented approximately at right angles to the shoreline. Reference and disposal sites at each study area were divided into parallel and approximately equally-sized sectors, including nearshore, middepth and offshore sectors. The purpose of the study was:

1. to compare benthic macrofaunal community composition and taxon abundance between reference and disposal sites and assess for possible gross effects of dredged material disposal on benthic macrofaunal recolonization and recovery;
2. to characterize sampling variance as guidance for possible future studies on effects of dredged material in comparable habitats; and,
3. to characterize the sediments for potential toxic contaminants.

Prior to this study, disposal of dredged material last occurred at the Mermentau River in June/July 1987, Freshwater Bayou in September/October 1990, and Atchafalaya River in May 1991. The study was requested by the EPA Region VI.

The ecological effects component of the investigation was limited because it did not include a pre-dredged material disposal assessment of benthic community composition as a reference. Inferences about causes of differences in benthic macrofaunal community structure were limited to a qualitative assessment. However, the analysis formed the basis for future sampling strategies, including a statistical power analysis, and an assessment of current benthic macrofaunal ecological status of reference and disposal sites, and the probable role of toxic chemicals as determinants of community structure.

The total number of taxa collected from reference (N = 21) and disposal (N = 21) sites at the Atchafalaya River, Freshwater Bayou, and Mermentau River study areas was 38 and 40, 21 and 18, and 14 and 17, respectively. The opportunistic polychaetes, *Mediomastus californiensis*

(reference $\bar{x} = 40.8$ and disposal $\bar{x} = 40.7/0.05\ m^2$) and *Paraprionospio pinnata* (reference $\bar{x} =$ 14.0 and disposal $\bar{x} = 10.5$) greatly dominated abundance at the Atchafalaya River and Freshwater Bayou reference and disposal sites, respectively. All benthic animal densities are referenced to 0.05 m^2 unless specified otherwise. Differences in abundance between sites of these highly dominant species on average were not significant ($P > 0.05$) but significant differences in abundance were noted within (i.e., N = 7 at all nearshore, middepth, and offshore stations) and among some reference and disposal stations. *P. pinnata* averaged 12.2 and 8.4 organisms at Atchafalaya River reference and disposal sites and *M. californiensis* averaged 6.2 and 1.2 organisms at Freshwater Bayou reference and disposal sites, respectively. These means differed significantly ($P \leq 0.05$: significance level used unless specified otherwise) between sites within study areas, and significant differences were observed within and among some reference and disposal stations. No other taxa differed significantly in abundance at Freshwater Bayou between sites, but at the scale of individual stations, seven other much less abundant taxa differed significantly within and among some stations at reference and disposal sites.

Other less abundant taxa ($\bar{x} > 5.0$ organisms/0.05 m^2 at a reference or disposal site) at the Atchafalaya River reference and disposal sites included the polychaetes, P. *ambigua, Spiochaetopterus oculatus*, and *Glycinde solitaria,* the bivalve, *Mulinia lateralis*, the nemertean, *Nemertea* sp. A., and unknown oligochaetes. Abundance of a nemertean, Oligochaeta, and a much less abundant taxa ($\bar{x} \leq 5.0$ organisms/0.05 m^2), *Polydora* sp., were significantly greater at the reference site. Abundance of *Streblospio benedicti,* another much less abundant taxa, was significantly greater at the disposal site. Overall, abundance of 10 of 17 taxa, whose mean abundance was greater than 1.0 organism/0.05 m^2, differed significantly within and among some stations at reference and disposal sites.

The polychaete, *P. pinnata*, dominated abundance at the Mermentau River (reference $\bar{x} =$ 51.2; disposal $\bar{x} = 113.5$). The difference in abundance was significant. The less abundant taxa, *P. ambigua*, averaged 5.8 and 1.0 organisms at disposal and reference sites; the difference was significant. Three additional less abundant taxa, *G. solitara*, unknown Nemertea, and *Magelona* sp., were significantly more abundant at the disposal site, and the polychaete, *Cossura soyeri*, was significantly more abundant at the reference site. Abundance of individual taxa at all three study areas did not show an apparent graphical correlation with hydrographic factors, sediment particle size, and sediment organic contaminant or metal concentrations.

The pattern in average abundance and average taxa richness differed substantially between sites at all three study areas. At the Freshwater Bayou, abundance averaged 10.0, 25.6, and 44.4 individuals at the combined reference and disposal nearshore, middepth, and offshore locations. For these same locations, taxa richness averaged 3.3, 5.5, and 7.4. For both response parameters, the nearshore averages differed significantly from both middepth and offshore averages but the middepth and offshore averages did not differ significantly. Differences in response parameters showed no apparent graphical relationship to sediment particle size (e.g., % sand) and sediment chemical contaminant concentrations or hydrographic factors.

Significant statistical interactions in average abundance and taxa richness between sites and station locations at the Atchafalaya and Mermentau Rivers required a *post hoc* analysis to appropriately distinguish differences. Abundance averaged 144.3 and 51.6, and 92.7 and 176.1, at Atchafalaya River nearshore and middepth reference and disposal stations, respectively. Differences were statistically significant. Offshore reference and disposal sites averaged 91.3 and 85.0 organisms; this difference was not significant. Taxa richness difference which averaged 11.9 at the Atchafalaya River nearshore reference and 6.1 at the nearshore disposal station was significant. Significant differences were also observed among non-paired stations. Differences in benthic community structure showed no apparent relationship to measured sediment properties and hydrographic factors.

Abundance averaged approximately two to three times more individuals at disposal than reference stations at the Mermentau River study area (e.g., offshore reference and disposal stations averaged 44.0 and 136.3 organisms, respectively), and all disposal station averages were significantly greater than reference station averages which was the basis for discounting the importance of statistically significant ($P \leq 0.043$) main effects and station location interactions. Taxa richness averaged 4.3 and 9.8 at middepth reference and disposal stations; differences were statistically significant. Although not significant, average taxa richness at nearshore and offshore disposal stations tended to average higher values than comparable reference stations.

Differences in community structure occurred between sites and among some stations within sites at the Atchafalaya River where measured sediment and hydrographic factors were relatively homogenous. The more sandy disposal compared to reference site at the Mermentau River was associated with higher abundance and numbers of taxa. Abundance and taxa richness at the Freshwater Bayou was associated with the depth gradient and not with measured reference and disposal sediment and hydrographic factors.

The consistent pattern of no significant differences in average abundance and taxa richness between reference and disposal sites at the Freshwater Bayou, after approximately one year since dredged material disposal, suggests no gross long-term sustainable effects on benthic macrofaunal community structure due to disposal activities. The higher average abundance and trend for higher taxa richness at the Mermentau River disposal site could be the result of natural factors and not necessarily the result of dredged material disposal, especially with a four year period since disposal activities last occurred.

Although differences were detected in community structure between reference and disposal sites at the Atchafalaya River study area, significant differences in average abundance and taxa richness between paired reference and disposal stations occurred only at the nearshore location. Four of five individual taxa showed significantly greater abundance at reference sites. The significant difference in abundance of the dominant species, *P. pinnata*, occurred only at the nearshore stations. Since dredged material disposal occurred only five months prior to this study, disposal effects in this case were not improbable as suggested for the Freshwater Bayou. However, if the measured differences are the result of dredged material disposal, we suggest that the magnitude is not large compared to the known seasonal and interannual variability of members of benthic macrofaunal communties reported in the literature.

The addition of the seventh replicate sample per station seldom added a new taxon to the inventory. In most cases, four or five replicates captured all but probably the most rare species. The sampling power analysis revealed that approximately 25 to 50 replicate samples would be required to achieve an accuracy of the mean for abundance +/-10%. The need for this level of accuracy should be determined on a case by case basis because of the obvious high cost for this

level of sampling. Relaxation of the accuracy requirement to 20% substantially reduced the number of samples required to achieve a significant P-value ≤ 0.05.

We recommend application of the Before-After-Control-Impact (BACI) study design in future environmental assessment studies of dredged material disposal. The design incorporates more appropriate temporal sampling and replicated control sites. Other study design recommendations are provided in the report. The BACI approach may involve a degree of indeterminacy that should be addressed in future research.

Keywords: Dredged material disposal, dredged material effects, benthic macroinvertebrates, Louisiana coast.

INTRODUCTION

"The variability of nature is legend. In simple terms, variations in biological populations and rates of processes in time and space constitute statistical noise from which the pollution effects must be deciphered" (Boesch, 1984).

Dredged material disposal in coastal waters is of ecological concern because of potential direct and indirect effects from physical smothering of benthic communities and changes in water and sediment quality (e.g., exposure of organisms to toxic chemicals and possible food web biomagnification of toxic chemicals.) Related concerns involve physical changes in the hydrographic regime and physical disruption of benthic habitats (e.g., changes in grain size and disruption of the biogeochemical integrity of sediments [Cronin, 1967; Kirby et al., 1975, Probert, 1984]). Much has been learned concerning biological effects of dredged material disposal in open waters, especially since inception of the Corps of Engineers Dredged Materials Research Program of the Waterways Experiment Station in 1973 (Kirby et al., op cit). Additional information has been acquired from site-specific studies performed to support Environmental Impact Statements under EPA's designation of ocean dumping site program.

Most early work on effects of dredged material disposal and other sources of sediments on aquatic biota was conducted in freshwaters. Wilson (1957) and Cordone and Kelley (1961) reviewed some of the many studies of effects of sediments on aquatic life in freshwater streams and rivers. In the marine environment, Ingle et al. (1955) studied chemical effects of dredging in Mobile Bay, AL. Lunz (1938), working in coastal South Carolina, found that oyster mortality occurred only when organisms were smothered with silt; otherwise, no important physiological damage was detected due to dredged material placement. In a field study in coastal Louisiana, Mackin (1961) also showed that oysters tolerated silt concentrations from 5 to 700 mg $liter^{-1}$ Markey and Putnam (1976) concluded that no benthic community effects from dredging were apparent in the Gulfport Ship Channel six weeks after maintenance dredging operations. As part of a comprehensive field study on effects of overboard disposal of dredged materials in upper

Chesapeake Bay, Flemer et al. (1968) described short-term decreases in abundance of two benthic species (e.g., a hydrobid snail and the isopod, *Cyathura polita*). Reference samples were collected approximately one month (September 1966) prior to dredge material disposal (October 1966) and compared with samples collected approximately one month after dredge material disposal (December 1966). However, Pfitzenmeyer (1970) reported for the same area that recovery in the disposal area of benthic macroinvertebrates approximated that prior to dredging operations one year later. In the early 1950's, Ingle (1952) concluded that local habitat variability required that each dredging and disposal operation warranted separate environmental impact consideration. This approach continues as current policy within the U.S. EPA.

Early laboratory studies found species-specific effects of fine sediments on benthic animals. Loosanoff and Tommers (1948) reported reduced mollusc pumping rates in molluscs with additions of small quantities of silt. In contrast, Chiba and Oshima (1957) worked with three species of marine pelecypods and reported very little effect with low concentrations of inorganic particles. Silt and clay particles less than 0.75 g liter^{-1} (actual sediment concentrations not given but presumed to be in the low milligrams liter^{-1} range) showed no significant developmental effects to the straight hinge stage of the hard clam, *Mercenaria mercenaria* (Davis, 1960) compared to seawater controls . Normal development decreased progressively at successively higher silt and clay concentrations.

In addition to early work on effects of dredging and disposal operations on biota in the Gulf of Mexico, considerable background information exists on the distribution, abundance, trophic structure, and seasonality of benthic communities and characterization of sediment physical properties and concentration of contaminants. Much of this work was sponsored in response to concerns about changes in salinity (brine) and associated water quality factors (e.g., hypoxia, toxic chemicals) following discharge from leaching of salt domes used for oil storage (e.g., Gaston et al., 1985; Gaston and Edds, 1994; Gaston, 1985; Giammona and Darnell, 1990; DeRouen et al., 1983; Parker et al., 1980). Gaston and Edds (1994) presented evidence that changes in hydrographic conditions in concert with hypoxia were major determinants of benthic macroinvertebrate community structure at the West Hackberry coastal Louisiana brine disposal site.

Numerous ecological studies, including benthic macroinvertebrate surveys, have been conducted in the northern Gulf of Mexico (e.g., Oetking et al., 1979; Ward et al., 1979) in efforts to identify effects of oil drilling and production on coastal environments. Nearshore coastal studies have characterized hydrographic and eutrophic conditions (Rabalais et al., 1986; Rabalais, 1992) and sedimentological conditions (Jones and Williams, 1979) that provide additional evidence of the importance of these factors to the survival and development of coastal marine benthic communities. Estuarine environmental monitoring programs conducted by EPA (Summers et al., 1993a and b) and NOAA (1992) provide a broad range of environmental data. Such information helps form a basis for *a priori* expectations of seasonal distribution and abundance of benthic macroinvertebrates by sediment type for many areas in estuaries and on the shelf of the northern Gulf of Mexico.

Environmental concerns about dredged material disposal and other water quality issues resulted in legislation such as the Clean Water Act of 1972 and the Marine Protection Research and Sanctuaries Act of 1972 (also known as the Ocean Dumping Act). Section 404 of the Clean Water Act authorizes the U.S. Environmental Protection agency (EPA) and the U.S. Army Corps of Engineers (COE) to jointly regulate the disposal of dredged and fill materials in bays, harbors, estuaries, rivers and lakes (waters of the U.S.). The Ocean Dumping Act authorizes the EPA and the COE to jointly regulate the disposal of dredged materials in the open ocean (seaward of the baseline). The COE uses the EPA regulations to determine the suitability of dredged material for ocean disposal.

Because of the nature of the sediments, relatively shallow coastal waters, and commercial shipping and recreational boating, the state of Louisiana has a continuing need for dredge and disposal operations (U.S. EPA 1990a,b,c). As part of EPA's environmental evaluation process and site monitoring responsibilities, a comparison of the benthic community structure at three Ocean Dredged Material Disposal Sites (ODMDS) located in the coastal Louisiana nearshore waters was conducted as an initial step in the environmental assessment of effects of disposal of dredged material adjacent to deepened channels connecting the open Gulf waters to the intra-coastal waterway. The three sites are located in the vicinity of the Mermentau River, Freshwater Bayou and near the mouth of the Atchafalaya River (Figures 1 & 2).

The purpose of the study was: 1. to compare benthic macrofaunal community composition and taxon abundance between reference and disposal sites and assess for possible gross effects of disposal of dredged material on benthic macrofaunal recolonization and recovery; 2. to characterize sampling variance as guidance for possible future studies on effects of dredge material disposal in related habitats in the northern Gulf of Mexico; and 3. to characterize the sediments for potential toxic contaminants (see Moore et al., 1994; Appendix I).

BACKGROUND AND RATIONALE

An opportunity to conduct the study occurred during October 1991. The study occurred after dredged material had previously been disposed in each of the study areas. Elapsed time since last disposal differed among the three study areas. Prior to this study, the most recent dredged material disposal at the study areas occurred as follows: Atchafalaya River--May 1991; Freshwater Bayou-- September/October 1990; and Mermentau River--June/July 1987.

Without pre-disposal data, a rigorous comparison of effects of dredged material disposal was not feasible as typically done in "before and after" environmental studies. Thus, our approach was necessarily modified. Benthic marine soft sediment macrofaunal communities are known generally to be patchy in distribution and variable in time (Pearson and Rosenberg, 1978) and this has been documented for the northern Gulf of Mexico nearshore coastal waters (Gaston and Weston, 1983). Therefore, we had no plausible basis to assume a high degree of "equivalency" between reference and disposal site benthic macrofaunal community structure. In the northern Gulf of Mexico, warm season benthic macrofaunal recolonization is known, in at least one case, to be well developed after about five months (Gaston et al., 1985). If this is generally the pattern, then changes in community structure at greater intervals of time will be difficult to ascribe to a perturbation such as dredged material disposal, unless the perturbation has a frequency less than approximately five months and occurs during the warm season.

Considering the above perspective, we applied an ANOVA model in an exploratory manner in the sense of Heck and Horwitz (1984) to examine spatial variability in benthic macrofaunal community structure, not in the sense of formal hypothesis testing with resultant causal inferences. Quantification of ecological variability is a first step in the description of the range of states possible for components of ecological communities (Duarte, 1991). We discuss significant differences in response parameters in the context of natural variability and recolonization rates. Where plausible, we speculate on the relationship of measured differences in community structure that might be attributed to dredged material disposal. Assessment of effects of dredged material disposal at the Atchafalaya River study area should be more directly related to disposal activities than is the case for the other two study areas because of the relatively short time of five months since last disposal. However, the lack of pre-disposal information remains a cardinal limitation to causal inferences in the sense of a formal field experiment for all three study areas. Given the above limitations, we believe that gross differences in response parameters between reference and disposal sites can provide provisional information concerning possible effects of dredged material disposal.

DESCRIPTION OF STUDY SITES

All three study areas are located in the nearshore shallow coastal zone of Louisiana (Figure 1). Detailed environmental characterizations are provided in separate Draft Environmental Impact Statements (U.S. EPA, 1990 a, b, c). Generally, sediments in the channels and disposal sites for all three study areas are dominated by silts and clays with some textural variability (U.S. EPA, 1990 a, b, c).

Atchafalaya River Site

The Atchafalaya River Bar Channel provides ship access to Morgan City, LA, the Gulf Intercoastal Waterway, and the Bayous Chene, Boeuf, and Black from the Gulf of Mexico (Figure 2). The proposed Ocean Dredged Material Disposal Site (disposal site) is 30.8 km (19.1 miles) long and 0.8 km (0.5 miles) wide. The center of the proposed site is approximately 26 km (16 miles) from the mainland shore. The proposed site has an average depth of approximately 5.0 m (16 ft) and a total area of approximately 2480 ha (9.57 square miles). The reference site with dimensions approximately similar to that of the disposal site is located parallel just west of the dredged channel (Figure 2). The dredged material generally consists of approximately 45%

silt, 45% clay, and 10% fine-grained sand with some temporal and spatial variability (U.S. EPA 1990a).

Freshwater Bayou Site

The Freshwater Bayou disposal site is used approximately once every three years to receive material dredged from the Freshwater Bayou Channel located approximately 23 km (13.5 miles) southwest of Intercoastal City, LA. The channel is 76 m (250 ft) wide and 5.6 km (3.5 miles) long (Figure 2). The disposal site is 610 m (2000 ft) wide, 5.6 km (3.5 miles) long, and runs parallel to the channel on its west side. A reference site of dimensions similar to the disposal site is located on the east side of the channel (Figure 2). The 343 ha (848 acre) disposal site begins at the shoreline at the mean low tide mark and extends seaward approximately perpendicular to the shoreline for 5.6 km. Seawater depth at the seaward boundary is 4.9 m (16 ft). A hydraulic pipeline dredge was used to deposit dredged material at a minimum distance of 458 m (1500 ft) west of the centerline. An average of 0.92 X 10^6 m^3 (1.2 X 10^6 cubic yds) of dredged material is disposited at the disposal site from a hydraulic cutterhead dredge. Channel-dredged sediment consists of a mixture of sand, silt and clays; silts and clays comprise about 88% and fine-grained sands make up the remainder (U.S. EPA, 1990b). *Mermentau River Site*

The Mermentau River disposal site is also used approximately once every three years to receive material dredged from the Mermentau River-Gulf of Mexico Navigation Channel, located approximately 6.4 km (4 miles) south of the town of Grand Cheniere, LA (Figure 2). The channel is 60 m (200 ft) wide and 2.0 km (1.25 miles) long. The disposal site is approximately 1.6 km (1.0 mile) long, 0.8 km (0.5 miles) wide, and runs parallel to the channel on its west side. The reference site is approximately similar in size to the disposal site and lies just to the east of the channel (Figure 2). The 135 ha (335 acre) disposal site starts at 0.6 km (0.4 mile) and extends to approximately 2.2 km (1.4 miles) offshore with depths ranging from 1.2 to 4.3 m (4 to 14 ft). Approximately 382.5 X 10^3 m^3 (500 X 10^3 cubic yds) are disposed of in the disposal site from a hydraulic cutterhead dredge. Specific information on sediment grain-size was not located, but sand apparently comprises about 5% of the sediment west of the channel and sediments are generally similar inside and outside of the channel (U.S. EPA, 1990 c). The Mermentau River

Channel was currently dredged in June/July of 1987; however, all material was used for beneficial use projects with no disposal on the disposal site.

METHODS AND MATERIALS

BENTHIC COMMUNITY

Field Sampling: Field operations and sample processing were conducted from the U.S. EPA Ocean Survey Vessel, Peter W. Anderson. Field sampling was conducted from a small support boat rigged with a boom and winch assembly. Sampling at the study areas occurred over the following days in October 1991, following some months-to-years after placement of dredged material at the respective disposal sites: Atchafalaya River (October 16), Freshwater Bayou (October 17) and Mermentau River (October 18). Each disposal site was divided into three segments or zones, representative of most landward and shallow areas (nearshore), middle region (middepth) and deepest offshore areas (offshore)(Table 1). A sampling station was established approximately near the center-point of each zone (i.e., projected as a rectangle) based on Loran C coordinates and verified in the field by channel marker buoys. Zones corresponding to those in the disposal site were established in each reference site located within "study areas" (Figure 2). For consistency of terminology, the three channel reference and disposal locations (e.g., Mermentau River, Freshwater Bayou and Atchafalaya River) are referred to as study areas. Reference and disposal locations within each study area are referred to as respective reference and disposal sites and sampling locations within reference and disposal sites are indicated by their respective depth location (e.g., nearshore reference station).

Ten 0.05 m^2 random benthic grabs were made at each reference and disposal site at approximately the center-point of each depth zone (e.g., offshore disposal station), respectively, with a modified Ponar sampler similar to that used in the EPA-EMAP Program (Summers et al., 1993). Station locations were determined by previously mapping station coordinates and confirming field location by comparing mapped coordinates with channel buoy markers. The sampler collects approximately the top 10 cm of sediment in fine sediment regimes and varying depths in more course materials. The general sampling area occupied a square of approximately 20 m on a side. An additional series of benthic grabs was taken at each sampling location within a depth zone to provide sediment for analysis of grain size and percent organic matter. Sediment grabs were sub-sampled with a Teflon-coated spoon to provide approximately one cup of sediment collected from the center of the ponar to a depth of about 5 cm. Sediment samples were composited into a one-liter polycarbonate jar with Teflon liner. While on station, but prior to sediment collection, bottom water temperature, salinity, pH, and dissolved oxygen were obtained with a HydroLab Surveyor II®. Water depth was measured with marked line and counter-weight.

A total of 180 benthic community samples (10 samples per sampling depth, e.g., nearshore station X 6 zones per study site X 3 study areas) was processed in the field. The entire contents of the ponar grab were wet-sieved through a 0.5 mm mesh. Material retained on the sieve was immersed in 0.3% propylene phenoxytol (a relaxant) for 2 to 5 min, then preserved in 10% buffered formalin containing Rose Bengal stain, and stored for transport to the ERL/Gulf Breeze. At the laboratory, samples were transferred to 60% isopropanol before sorting and taxonomic identification.

Biological analyses: Seven samples were selected randomly from the 10 samples collected in the field at each reference and disposal station. Samples were sorted into major taxonomic groups (Class or Order) with 10 X binocular dissection microscopes, identified to lowest possible taxa (species where possible) and enumerated. Data were entered into SAS data files for statistical analyses (see below).

Sediment particle size: Three replicate sediment subsamples from each channel reference and disposal sampling station were wet-sieved through a 63 μm mesh sieve to separate sands and silt/clay fractions (Folk, 1980). Each fraction was: dried at 100°C for 12 hr, cooled in a desiccator, and weighed to determine percent sand/silt and clay. Whole sediment samples were burned at 550°C to determine weight loss on ignition to estimate organic carbon.

Chemical analyses: *Chlorinated hydrocarbon pesticides and polychlorinated biphenyls (PCBs)*: A Hewlett-Packard Model 5890 gas chromatograph equipped with a ^{63}Ni electron-capture detector was used (see Moore et al., 1994, for details; Appendix I). Sediments were air-dried, weighed and extracted with a 20% (v/v) acetone and petroleum ether and then centrifuged (1600 $\bar{x}$ g) and solvent extracts extracted with 2.0% (v/v) aqueous sodium sulfate. After phase separation, the solvent was transferred to a concentrator tube, and the aqueous wash was repeated two more times. Sample extracts from the three aqueous washes were combined and concentrated to 1 ml on a nitrogen evaporator in preparation for a Florisil cleanup. Samples were shaken with 500 ml of mercury to remove sulfur before gas chromatographic analysis. Clean-up columns were prepared by adding 3 g of PR-grade Florisil and 2 g anhydrous powdered sulfate to a Chromaflex column and rinsed with 10 ml of hexane. Sediment extracts were transferred to the column with two additional 2-ml volumes of hexane. Pesticides and PCB's were eluted with 20 ml of 5% (v/v) diethyl ether in hexane. Dieldrin and endosulfan were eluted with 20 ml of 10% (v/v) isopropanol in isooctane. Separations were performed by using a 30-m (0.32 mm i.d.) RTX-5 and RTX-1 fused silica capillary column. The helium carrier gas flowed at 1.5 ml/min, column temperature was operated at 50°C (held for 2 min), ramped 10°/min to 150°C, and then 2°/min to 260°C (held for 3 min). Inlet temperature was 250°C; and detector temperature was 350°C. Makeup gas was 10% methane in argon flowing at 60 ml/min. Pesticides were analyzed with external standards; all standards were obtained from the EPA pesticide repository, Las Vegas, Nevada.

Petroleum hydrocarbons: Analyses were performed on a Perkin-Elmer gas chromatograph (GC) equipped with a flame ionization detector (see Moore et al., 1994, for details; Appendix I). Sediment extracts (Warner, 1976) were injected into the GC and separations were performed by using a fused silica capillary column. Helium carrier gas was used at a flow rate of 1.5 ml min^{-1}. Other GC parameters were: oven temperature programmed from 50°C (hold for 2 min) at a rate of 20° min^{-1}; injector temperature was 250°C, and detector temperature was 350 °C. Androstane was obtained from Sigma Chemical Co., St. Louis, MO, and used as an internal standard to quantify petroleum hydrocarbons.

Heavy metals: One to two grams of sediment were weighed and placed into a 40-ml reaction vessel. Five milliliters of concentrated nitric acid were added and the samples digested for 2 to 4 h at 90°C in a tube heater. Digestion was continued, with vessels capped, for 48 h at 70°C. After digestion, samples were transferred to 15-ml tubes and a 5-ml aliquot was diluted to 10 ml for aspiration into a Jarrell-Ash AtomComp 800 series inductively-coupled argon-plasma emission spectrometer (ICP). This instrument acquires data for 15 elements simultaneously. No detectable residues could be found in method blanks. A solution of 10% nitric acid/distilled water was aspirated between samples to prevent carryover of residues from one sample to the next. Standards in 10% nitric acid were used to calibrate the instrument initially and adjustments were made when necessary. Concentrations were reported in two significant figures as our method allowed, and were not corrected for percentage recovery. Standard solutions of metals were obtained from J. T. Baker Chemical Co., Phillipsburg, NJ, and were Instra-Analyzed quality.

For all chemical analyzes, reagent and glassware blanks were analyzed to verify that the analytical system was not contaminated with chemical residues that could interfere with quantitation (Moore et al., 1994; Appendix I).

Statistical Procedures-- Data were transferred to computer disk for statistical summary and analyses using SAS software (SAS, 1989a and b). Average total numbers of organisms per 0.05 m^2 or abundance, average total number of taxa per 0.05 m^2 or taxa richness and dominant taxa were used as response variables. This unit area quantification is used in this paper unless otherwise specified.

Statistical analyses included a Two-Way Analysis of Variance (ANOVA, Sokal and Rohlf, 1981) between reference and disposal sites and among sampling depths. If statistically significant differences ($P \leq 0.05$) among stations were indicated by ANOVA , Tukey's Studentized Range Test was used to test for differences among stations. Before ANOVA procedures were used, the number of organisms in each sample was log-transformed [i.e., log ($\bar{x}$ +1)] to stabilize variance and improve normality. The number of taxa in each observation was transformed by taking the square root. A note of clarification--we used the ANOVA as an exploratory technique to measure variation within and between reference and disposal sites. The field analysis does not control all potentially important exogenous variation and environmental heterogeneity between reference and disposal sites. Therefore, we emphasize the correlational aspect of the analysis *vs* a controlled experiment in the classical sense.

Rare taxa were qualitatively compared by grouping taxa into geometric classes by number of individuals by station for each reference and disposal site. Scale of geometric classes was 2 X, i.e., Class I = 1 individual, Class II = 2 to 3 individuals and Class III = 4 to 7 individuals, etc. This analysis facilitated comparative scaling among samples which contained different numbers of individuals (Gray and Pearson, 1982). The accuracy of average density estimates for each station for reference and disposal sites was calculated using sample variance (Eckblad, 1991).

RESULTS

HYDROGRAPHY-SEDIMENT

Atchafalaya River

Water depths increased slightly in a seaward direction in reference and disposal sites from 1.9 and 1.1 (e.g., nearshore station) to 3.3 and 3.5 m, respectively (offshore station; Table 2). Bottom water salinities differed little in magnitude between reference and disposal sites among stations (Table 2) but increased with distance from shore (e.g., salinities were 8.0 and 9.5‰ in reference and disposal sites of the nearshore stations and 24.4 and 25.8‰ in reference and disposal sites of the offshore stations). Bottom water temperatures ranged between 22.9 and 23.7°C, and small differences (e.g., equal to or less than 0.3°C) were measured between reference and disposal sites with little difference recorded over the salinity gradient. Dissolved oxygen concentrations ranged between 6.2 and 7.5 mg liter^{-1} with small differences, typically less than 0.6 mg liter^{-1}, noted among stations in reference and disposal sites (Table 2). Unusually high pH values of 8.6 to 8.8 were recorded for the nearshore stations at salinities of 8.0 to 9.5‰ and low values of 7.4 were observed at salinities of 24 to 26‰. Nominal differences of 0.2 pH units or less were noted between reference and disposal sites.

Sediment particle size was highly variable along the depth gradient for reference and disposal sites and moderately variable at comparable sampling depths between reference and disposal sites (Table 2). At nearshore stations, reference and disposal sites were characterized by course (sand) particle sizes of 60.0 and 56.6% and this size category decreased to 1.9 and 9.8% at reference and disposal middepth stations, stations with the largest differences in percent sand between reference and disposal sites. Offshore stations in reference and disposal sites both contained 3.4% sand. Weight loss on ignition (WLOI) ranged between 2.2 and 6.6% of sediment mass. Small differences in WLOI of 0.5 to 1.3% were measured between reference and disposal stations. Nearshore reference and disposal stations were characterized by lowest values of WLOI (reference = 2.2 and disposal site = 3.5%). Highest values occurred at middepth stations and values in reference (6.6%) and disposal middepth stations (6.1%) approximated each other.

Freshwater Bayou

Water depths increased from 1.6 and 1.7 m in the reference and disposal sites at nearshore stations to 4.0 and 3.0 m at offshore stations (Table 2). Salinities ranged between 26.0 and 27.2‰ with small differences (< 0.2‰) between reference and disposal sites. Highest salinities were measured at reference and disposal offshore stations (e.g., 27.2‰). Differences in bottom water temperatures were equal to or less than 0.2°C among all stations. Water temperatures ranged from a low of 21.7 at the nearshore disposal station to 22.8°C at the offshore reference station. Dissolved oxygen ranged between 6.6 and 6.8 mg $liter^{-1}$ among all stations.

Percent sand decreased in a seaward direction (Table 2) and variability was especially prevalent at the middepth stations. Values ranged from 3.0 to 4.5 at reference and disposal nearshore stations (Table 2) to 3.1 to 24.8 at middepth stations. Shell fragments were abundant at this site and resulted in the unusually high variability and large particle size. Offshore stations were characterized by percent particle sizes indicative of fine mud with 99^{+}% less than 63 μm. Percent WLOI ranged from 6.7 to 11.6 with highest values recorded for offshore stations. Differences in WLOI for stations at comparable depths between reference and disposal sites were moderately variable. For example, percent WLOI was 9.1 and 6.7% at nearshore reference and disposal stations. Differences approximated 0.5 and 0.8% for stations located at middepth and offshore reference and disposal sites (Table 2).

Mermentau River

Water depths ranged between 1.5 and 2.0 m, 1.8 and 3.0 m, and 2.6 and 3.7 m at reference and disposal stations. Salinities approximated 30.5‰ for all stations. Water temperatures ranged from 22.0 to 22.5°C among all stations. The concentration of dissolved oxygen ranged from 5.9 to 6.9 mg $liter^{-1}$ for all stations. The pH values for reference and disposal sites closely approximated each other with values for the study area ranging between 8.0 and 8.3 and differences between reference and disposal sites being less than 0.2.

Percent particles greater than 63 μm was quite variable between reference and disposal sites (Table 2). Values for the nearshore stations ranged between 17.5 (reference) and 40.8 (disposal site), middepth stations ranged between 6.7 (reference) and 10.9 (disposal site) and at offshore stations values ranged between 0.9 (reference) and 12.2% (disposal site). Values for percent WLOI were moderately variable with nominal differences among stations.

CHEMICAL RESIDUES

Selected Chlorinated Hydrocarbon Pesticides & PCBs: No chemicals in these classes were detected in sediments at the Mermentau River or Atchafalaya River study areas above detection limits of 0.010 $\mu g\ g^{-1}$ wet wt (see Table 3 for chemicals analyzed and their percent recovery). The pesticide, DDE (P,P') (0.026 $\mu g\ g^{-1}$ wet wt) was detected in sediments at the Freshwater Bayou nearshore disposal site, and this pesticide occurred in sediments at the offshore reference site at a concentration of 0.036 $\mu g\ g^{-1}$ wet wt. Hexachlorobenzene (0.018) and methoxychlor (0.047 $\mu g\ g^{-1}$ wet wt) occurred in sediments at the offshore reference site. Only four samples out of a total of 324 contained chemical contaminant concentrations above detection limits [e.g., 18 chemicals screened X 3 study areas X 2 sites/study area (i.e., reference and disposal sites) X 3 depth zones/site = 324].

Petroleum hydrocarbons: Petroleum hydrocarbons were not detected in any sediment sample above 1.0 $\mu g\ g^{-1}$ wet wt, the method detection limit.

Heavy metals: Concentrations of selected heavy metals approximated each other between reference and disposal sites (Tables 4). Concentrations of these metals from different water depth zones (e.g., Atchafalaya nearshore reference) were similar except for chromium and copper. Metal concentrations were lower in nearshore than in offshore samples at both Mermentau River disposal site and its reference site, and in the Atchafalaya River disposal and reference sites.

MACROFAUNAL DISTRIBUTION AND ABUNDANCE

Atchafalaya River

Reference and disposal sites contained 38 and 40 taxa of which sites shared 29 taxa in common (Table 5). Of five molluscan taxa, only the bivalve, *M. lateralis*, occurred at both sites. Among 10 arthropod taxa, only three co-occurred at both sites (e.g., *Balanus* sp., *Oxyurostylis smithi*, and the free-living pinnixid, *Pinnixia sayana*). Annelids dominated taxa richness;

nineteen of 27 annelids co-occurred at both sites. The more abundant taxa usually occurred at all three stations within a site and they occurred 15 out of 21 samples (Table 6). Seven taxa occurred at nearshore, middepth, and offshore stations of both reference and disposal sites. Less abundant taxa (e.g., less than 2 individuals/0.05 m^2) frequently occurred at only one or two stations at a site.

There was very little difference in average taxa richness/0.05 m^2 at the reference site ($\bar{x}$ = 13.1; sd = 2.83) compared to the disposal site ($\bar{x}$ = 11.5; sd = 4.88; Table 7). Treatment, within treatment and interactions were all statistically significant (Table 8), thus we applied Tukey's test to examine station location differences between and within treatments. Average taxa richness at the nearshore reference station ($\bar{x}$ = 11.9) was nearly twice that of the nearshore disposal station ($\bar{x}$ = 6.1). This was the only case of a significant difference ($P \leq 0.05$) between paired reference and disposal stations. Average taxa richness of 16.0 at the offshore disposal station was the maximum value for the study area and was significantly greater than averages at the nearshore reference and disposal, and middepth reference stations (Table 8).

Little difference in overall average abundance/0.05 m^2 between reference ($\bar{x}$ = 109.4; sd = 39.05) and disposal sites ($\bar{x}$ = 104.2; sd = 77.75) sites (Tables 7) was evident, but differences in abundance within sites and among stations between sites were detected. Abundance averaged approximately three times fewer individuals at the nearshore disposal ($\bar{x}$ = 51.6) compared to the nearshore reference ($\bar{x}$ = 144.3) station (Tables 9). Differences were significant ($P \leq 0.05$). The difference in abundance was due primarily to high numbers of the polychaete, *Mediomastus californiensis*, at the nearshore reference station ($\bar{x}$ = 81.1; Table 10) compared to those at the nearshore disposal station ($\bar{x}$ = 30.3). The polychaete, *Pseudoerythoe ambigua*, ranked 2nd and 3rd in abundance at disposal and reference sites but abundance was not significantly different ($P > 0.05$) between sites. Average abundances of *P. pinnata* and an unknown nemertean were significantly greater ($P \leq 0.05$) at the reference than at the disposal site (e.g., 12.2 *vs* 8.4 and 9.2 *vs* 8.8), respectively; Table 10). Five other taxa (i.e., *Polydora* sp., *Paraprionospio pinnata*, *Spiochaepterus oculatus*, unknown Nemertea, and unknown Oligochaeta; Table 10) contributed significantly more individuals to reference than to disposal sites. Nine and eight taxa, respectively, differed in average abundance within reference and disposal stations. Abundance at the middepth disposal station averaged 1.9 times ($\bar{x}$ = 176.1) that of the middepth reference station ($\bar{x}$ = 92.7); the difference was significant ($P \leq 0.05$). Again, the polychaete, *M. californiensis*, accounted for much of the difference in abundance (e.g., middepth disposal station--$\bar{x}$ = 84.3 and middepth reference-- $\bar{x}$ = 27.7).

Sample size was adequate to capture most taxa at stations within this study area based on "species area curves" (e.g., middepth disposal station; Figure 3). At this station twenty taxa were collected with three replicate samples compared to 21 taxa with six replicate samples.

A relatively large percentage of taxa contained only a few individuals (Figure 4; Table 10). The general pattern of percent taxa and abundance group was approximately similar between Atchafalaya reference and disposal sites (Figure 4).

Freshwater Bayou

Only a small numerical difference was noted in taxa richness between reference (21 taxa) and disposal sites (18 taxa). Thirteen taxa occurred in common between sites (Table 5). Molluscs contributed only two species, the bivalve, *Mulinia lateralis*, and the snail, *Nassarius vibex* . Four crustacean taxa were present (*Balanus* sp., *Corophium* sp., a paguridian and a pinnixid crab). Annelids dominated taxa richness with 14 taxa (most identified to species level); polychaetes contributed 13 taxa and the Class, Oligochaeta, may have contributed more than one species. Of the more abundant taxa, five co-occurred at all stations at both the reference and disposal sites (Table 6). The polychaete, *P. ambigua,* occurred at all three reference sites but only occurred at the middepth and offshore disposal stations. The polychaete, *P. pinnata,* occurred in 20 and 21 samples out of a total of 21 samples at the reference and disposal sites, respectively, whereas *M. californiensis* only occurred in 15 and 7 samples out of a possible total of 21 samples at these sites (Table 6). The next most frequently occurring taxa, *P. ambigua*, was present in only 10 samples at both reference and disposal sites.

The overall average taxa richness/0.05 m^2 at the Freshwater reference ($\bar{x}$ = 5.5; sd = 3.30) and disposal sites ($\bar{x}$ = 5.2; sd = 2.28; Table 7) closely resembled each other and differences were not statistically significant ($P = 0.896$). Taxa richness averaged 3.3, 5.5 and 7.4 at the combined reference and disposal nearshore, middepth, and offshore stations, respectively (Table 7). The combined average of 3.3 was significantly ($P \leq 0.05$) less than averages of 5.5 and 7.4 but the average of 5.5 and 7.4 were not significantly different (Table 8). Abundance averaged 28.8 (sd = 32.46) and 24.5 (sd = 21.21) per 0.05 m^2 at the Freshwater reference and disposal sites (Table 7). Sampling variation was too large to detect a significant difference. Abundance averaged 10.0, 25.6, and 44.4 at combined reference and disposal sites at nearshore, middepth and offshore stations, respectively (Table 9). The polychaetes, *P. pinnata* and *M. californiensis*, contributed most to the increased abundance of the combined middepth compared to the combined nearshore stations, and the polychaetes, *P. pinnata* and *Pseudoeurythoe ambigua*, enriched the abundance at the combined offshore stations compared to the combined middepth stations (Table 11). The combined average of 10.0 differed significantly ($P \leq 0.05$) from both the combined average 25.6 and 44.4, but the values of 25.6 and 44.4 were not significantly different primarily because of the high variance associated with the mean of 44.4 (Table 9). Average abundance of the polychaete, *M. californiensis*, was greater ($\bar{x} = 6.2$) at the reference site than at the disposal site ($\bar{x} = 1.2$); the difference was significant. The reverse pattern was observed for the polychaete, *P. ambigua*, where abundance averaged 3.4 and 1.9 individuals/0.05 m^2 at the disposal and reference sites; differences were significant. Average abundance of most taxa differed significantly among one or more stations within and between reference and disposal sites (Table 11).

Moderate differences were detected in species-area relationships among stations at both freshwater reference and disposal sites (Figure 3). Three samples captured 12 of a total of 13 taxa at the middepth disposal site, whereas four samples were required to capture 17 of 18 taxa at the Freshwater offshore reference site.

Abundance Groups I-IV differed greatly in percent taxa between Freshwater nearshore reference and disposal stations (Figure 4). The nearshore disposal station contained half of the taxa in Abundance Group I compared to none at the nearshore reference station. Percent taxa in abundance groups between middepth reference and disposal sites approximated each other. The offshore disposal station contained more taxa in Abundance Groups VI through VIII than the comparable reference station. The reference and disposal sites differed considerably in the overall pattern of percent taxa for each abundance group.

Mermentau River

The disposal site contained more taxa (17) than the reference site (14) (Table 5). Polychaetes dominated taxa richness with 11 taxa. Polychaete taxa contributed seven and 10 taxa at reference and disposal sites, respectively; co-occurrence was limited to six taxa. Arthropods and molluscs were limited to three (i.e., *Balanus* sp., *Corophium* sp., and *P. sayana*) and one taxa (i.e., *M. lateralis*), respectively. Three rhyncocoels (*Cerebratulus lacteus, Micrura* sp., and *Nemertea* sp. A) occurred at reference and disposal sites. Five taxa (*Glycinde solitaria, P. pinnata, P. ambigua, Balanus* sp., and Nemertea sp. A) co-occurred at all three reference and disposal stations (Table 6). The numerically dominant species, *P. pinnata*, was present in all samples (N = 42 samples).

Taxa richness per 0.05 m^2 averaged 4.7 (sd = 1.65) at the reference site and 7.0 (sd = 2.36) at the disposal site (Table 7). A significant interaction between main effects and station locations (Table 8) complicated analysis. Average taxa richness at the middepth disposal station ($\bar{x}$ = 9.0) approximated twice that of the reference ($\bar{x}$ = 4.3) station; the difference was significant ($P \leq 0.05$). Taxa richness differed little between nearshore and offshore paired reference and disposal stations. Taxa richness at the nearshore reference station ($\bar{x}$ = 4.0) was only slightly more than one half that of the offshore disposal site ($\bar{x}$ = 7.0 station; Table 8). The difference was significant ($P \leq 0.05$).

Abundance per 0.05 m^2 averaged 57.9 (sd = 19.85) at the Mermentau reference site but increased by two-fold to 131.0 (sd = 24.31) at the disposal site (Table 7). The polychaete, *P. pinnata*, dominated abundance at both reference ($\bar{x}$ = 51.2) and disposal ($\bar{x}$ = 113.5) sites (Table 12). Analysis of average abundance was complicated by a significant statistical interaction between main effects and stations ($P = 0.049$; Table 9). However, we judged the interaction to be unimportant because of the large magnitude of differences in abundance between reference and disposal site stations noted above (Table 9). Six of seven relatively abundant taxa (i.e., mean abundance > 1.0 organism/0.05 m^2) were significantly more abundant at the disposal site (Table

12). Only one taxa, *Cossura soyeri*, averaged more individuals at the reference (2.6) than at the disposal site (2.4). Three of seven relatively abundant taxa were distributed with significant differences in abundance among reference and disposal stations (i.e., *P. ambigua, Magelona* sp., and *C. soyeri*).

Two samples were required to detect eight taxa, and four samples only increased the total to nine taxa at the Mermentau nearshore reference station. The latter was representative of other stations with only small differences noted in this relationship between reference and disposal sites (Figure 3).

Percent taxa in Abundance Groups II and V differed somewhat between reference and disposal sites (Figure 4). Abundance Groups V-X approximated each other closely. Overall, the abundance patterns between reference and disposal sites were approximately equal.

DISCUSSION

Analysis of differences in benthic community structure is one of the mainstays for detecting and monitoring the biological effects of marine pollution (Warwick and Clarke, 1993). Although temporal sampling was not possible in this study, spatial variability in response parameters was the central focus. We measured variability in benthic macrofaunal community structure (i.e., mostly species) at the scale of reference and disposal sites and within sites. At the scale of individual stations, or within site variability, we initially planned to focus on comparison of paired reference and disposal stations as the smallest scale resolution on the assumption that they would represent more ecologically comparable habitats than more distant non-paired stations. The presence of a water column depth gradient from shore seaward suggested the relevance of this assumption. Sediment particle size data by visual inspection were not highly consistent between paired stations (Table 2). Thus, sediment particle size variability was of limited value as an explanatory factor in community composition and taxa abundance, except possibly at the Mermentau River study area.

Large temporal variability typical of soft sediment benthic macrofaunal communities often results in benthic populations appearing very "noisy" (e.g., Franz and Harris, 1988; Gaston and Weston, 1983). These populations often show a marked lack of concordance in their temporal trajectories from one place to another. This contributes to high spatial variance. The consequence is that considerable statistical interaction occurs between changes in average abundances from time-to-time and from place-to-place (Underwood, 1994). Statistical interaction in the ANOVA model was characteristic in community level response parameters (i.e., average taxa richness and abundance) at the Atchafalaya and Mermentau River study areas.

Taxa collected in this study correspond closely to those identified in other near-coastal and estuarine waters of the northern Gulf of Mexico, especially coastal Louisiana (Gaston and Weston, 1983; Gaston and Nasci, 1988; Giammona and Darnell, 1990; Gaston and Edds, 1994). Wright et al. (1978) reported similar dominant taxa present at stations in 10 to 15 m waters at a dredged material disposal site in the Gulf of Mexico offshore of Galveston Bay, TX. Many dominant benthic macrofauna in our study co-occur in Hillsborough Bay, an arm of Tampa Bay, FL (Santos and Simon, 1980). Gaston and Weston (1983) reported that the phoronid, *Phoronis* sp. A (later confirmed as *P. muelleri*), was a dominant species at the West Hackberry brine diffuser site in approximately 10 m of water during March through July 1981, but failed to repeat a population erruption in 1982. *Phoronis* sp. (probably *P. muelleri*) occurred in only one sample at each of the Atchafalaya River reference and disposal sites and in only one reference sample at the Freshwater Bayou study area. Presence was only at offshore stations which suggests that this taxa is adapted to a more marine than an estuarine habitat.

The total number of taxa in our three study areas ranges from relatively low values of 14 to 17 (more typical of oligohaline areas; e.g., St. E-2, Calcasieu Lake--Gaston and Weston, 1983) in reference and disposal sites at the Mermentau River study area to moderately high numbers of 38 and 40 taxa in reference and disposal sites at the Atchafalaya River study area. The higher number of taxa approximates values reported for fall samples collected at the West Hackberry Strategic Petroleum Reserve Site Brine Disposal Area (Gaston and Weston, 1983), the Texas coastal shelf off Corpus Christi (Flint and Holland, 1980) and somewhat less (e.g., 65 *vs* 40) than Parker et al. (1980) reported for Weeks Island Brine Disposal Site located 44 km (26 miles) south of Marsh Is., LA. A trend was noted for higher average taxa richness to occur at the offshore stations at all three study areas, probably a reflection of higher salinities (Carriker, 1967) and reduced sediment disturbance due to wave action.

Many species in the present study, especially the dominant polychaetes, e.g., *Mediomastus californiensis, Paraprionospio pinnata, Pseudoeurythoe ambigua*, and the bivalve, *Mulinia lateralis*, are considered opportunists because they frequently are first-colonizers following natural and anthropogenically-initiated disturbances (Grassle & Grassle, 1974; Santos & Simon, 1980; Boesch & Rosenburg, 1981; Pearson and Rosenberg, 1978). Opportunistic benthic macrofauna are characterized by high reproductive potential, and many have planktonic dispersal mechanisms that contribute to rapid population build-up often followed by rapid population declines. Natural disturbance includes physical factors such as major freshets, e.g., Tropical Storm Agnes in June 1972, Chesapeake Bay, and biotic interactions (e.g., predator effects, competition and adult/larval interactions); and human disturbance includes dredge material disposal, release of toxic chemicals, and hypoxic events in overlying waters and sediments typically associated with nutrient over-enrichment. Presumably, varying degrees of interaction occur between natural and anthropogenic stressors and among each source class which complicates assessment of strictly anthropogenic effects (e.g., brine *vs* hypoxia; Gaston et al., 1985).

Total average abundance per meter square (extrapolated from 0.05 m^2) in our study averaged 580 to 480 individuals in the reference and disposal sites at the Freshwater Bayou study area. Abundance averaged 2,180 and 2,080 at the reference and disposal sites at the Atchafalaya River study area and averaged 1,160 and 2,620 at reference and disposal sites at the Mermentau River study area. Mean densities for the Freshwater Bayou are lower than most values reported in the literature for soft sediments in the northern Gulf of Mexico (e.g., Weston and Gaston, 1982; Gaston and Weston, 1983). These low densities correspond more closely to those reported for the upper Calcasieu Estuary (Gaston and Nasci, op. cit.) and the June sampling of the Gulfport, MS, dredge material disposal site (Markey and Putnam, 1976). Mean densities of one to two thousand individuals/m^{-2} range are not uncommon for the fall season in the northern Gulf of Mexico. Abundance of benthic macrofauna averaged over an 18-month study on the Texas shelf in 22 m depth approximated 1,000 m^{-2} (see Figure 2; Flint and Holland, 1980). Differences in water depth and large expanse of Texas coastline adds considerable uncertainty to the comparison.

Community level response variables, average taxa richness and abundance, did not differ significantly between reference and disposal sites at the Freshwater Bayou study area. Significant differences in these response variables occurred only between combined paired stations, e.g., between nearshore reference and disposal stations, and their middepth and offshore counterparts (Tables 8 & 9). This response pattern was unique to the Freshwater Bayou, and we see little reason to ascribe the effects of dredged material disposal to this response pattern.

Of nine most abundant taxa at the Freshwater Bayou study area, only two, the polychaetes, (*Mediomastus californiensis* and *Pseudoeurythoe ambigua*) showed significant differences ($P \leq 0.05$) between reference and disposal sites. Abundance of *M. californiensis* was significantly less ($P \leq 0.05$) at the disposal compared to the reference site ($\bar{x} = 6.2$ *vs* 1.2). Abundance of *P. ambigua* was significantly greater at the disposal *vs* reference site (1.9 *vs*

3.4)(Table 11). Abundance of several taxa, including *M. californiensis* and *P. ambigua*, differed significantly within and among reference and disposal stations. The magnitude of the differences among stations generally was as large within sites as between sites. The magnitude of differences in abundance is within natural variability for the northern Gulf of Mexico. Thus, if the differences, in fact, are caused by dredged material disposal, the magnitude appears to be nominal.

The pattern of increased abundance of benthic macrofauna from nearshore to offshore at the Freshwater Bayou study area is not explained by measured physical sediment or hydrographic factors nor by measured chemical contaminants. It appears that unidentified environmental gradients and possibly biotic interactions explain major features of the distribution and abundance of the benthic macrofauna at Freshwater Bayou. Further work is required to explain these findings.

The Mermentau River study area showed greatest relative differences between reference and disposal sites in average abundance and taxa richness compared to the other two study areas. The disposal site samples averaged significantly higher abundance than did the reference site, an unanticipated response. Differences in average taxa richness were less striking than average abundance; however, nearshore and middepth reference values for taxa richness averaged approximately one half of the middepth and offshore disposal values (e.g., 4.0 and 4.3 *vs* 9.0 and 7.0).

Individual taxa at the Mermentau River study area averaged significantly higher abundances in five out of six comparisons of dominant taxa at the disposal compared to the reference site, as would be expected based on the response variable, average abundance. Differences in percent sand among paired stations at this study area were consistently greater than those measured for other study areas, except for the single large difference due to shell fragments measured at the middepth Freshwater disposal station (Table 2). Abundance data for individual taxa at the Mermentau disposal site were more variable on average than at the reference site. The relatively high particle size variability was consistent with the benthic macroinvertebrate variability. Other measured environmental factors do not clarify the benthic macrofaunal distributional patterns observed at this study site.

The Mermentau River study area last received dredged material in June/July 1987, an interval considerably longer than at the other two study areas. This longer time interval adds considerable uncertainty to causal inferences regarding effects of disposal of dredged material. It is not clear why this study area contained fewer taxa than the two other study areas or the enhanced abundances of taxa at the disposal site. More frequent sampling could determine whether the increased abundance detected at the disposal site was within the longterm seasonal excursion characteristic of the study area. A speculative possibility is that the timing and magnitude of disturbance from dredged material disposal at this study area acted as an "intermediate disturbance" which would increase the number of species and their abundance (Sousa, 1984).

In contrast to Freshwater Bayou, analysis of average taxa richness and abundance of taxa at the Atchafalaya River study area was complicated by statistical interaction of main and within site effects. Average abundance was greater by approximately a factor of three and significant ($P \leq 0.05$) at the nearshore reference compared to the nearshore disposal station. Average abundance at the middepth disposal station was approximately twice that of the middepth reference station and the difference was significant ($P \leq 0.05$). Average abundance at offshore reference and disposal stations approximated each other. Variability (i.e., variance) in average abundance within sites was often as large as that between sites. Average taxa richness was significantly smaller at the Atchafalaya River nearshore disposal station compared to all other stations, and this response variable was significantly larger at the offshore disposal station than at the nearshore and middepth reference, and nearshore disposal stations. These differences do not show a consistent association between reference and disposal sites or within measured environmental variables, e.g., percent sand or sediment chemical contaminant data. Evidence of recolonization by macrofauna of marine soft sediments in the northern Gulf of Mexico suggests that recolonization should be well developed within about five months (Gaston et al., 1985). The "patchy" nature of community level indicators of recolonization masked possible effects of dredged material disposal at this study area.

The distribution and abundance of individual taxa at the Atchafalaya River study area indicated that average abundance was significantly greater at the reference site compared to the disposal site in four out of five comparisons where differences were statistically significant (Table 10). However, the differences were usually less than a factor of two. Variability within reference and disposal sites usually was as large as variability between sites which would minimize whole-site differential effects of dredged material disposal on taxa abundance. These data confirm that statistically significant differences in abundances were registered at the disposal compared to the reference site on selected individual taxa. These differences could be plausibly ascribable to dredged material disposal. However, we believe that the magnitude of the differences is not large compared to other stations in this study and in the literature.

We examined several aspects of sampling variability to assist in interpretation of our results and provide a better basis for determining future sampling requirements should further assessments be conducted for these study areas. It appears that four to five samples captured 90 to 95% of the taxa at all three study areas. At a disposal site offshore of Galveston Island, the species-area curve suggested the "plateau" was not reached for several sites with 10 replicate samples (Wright et al., 1978; Harper, 1977). Variability of all response variables in our study was relatively high for most stations. The coefficient of variability often approximated 50 to 100% and occasionally higher values were measured.

We estimated with the power analysis that frequently 25 to 50 samples would be required to achieve an accuracy of mean abundance of ± 10%, and in several cases one to several hundred samples would be required to achieve this level of accuracy (Figure 5; Table 13). Relaxation of accuracy to to 20% reduced the number of samples by approximately a factor of four in most cases. This level of quantitation would still pose a burden on most sampling programs. In the Galveston disposal study cited above, instances occurred where over 1,000 samples would be required to achieve a standard error equal to 20% of the mean. Wright et al. (1978) stated that variability not compensated for by an adequate sample size may have obscured some effects and indicated changes when none occurred. In areas where sampling variability is especially high, suggestive of patchy distribution, it would be desirable to stratify the sampling effort in future assessments and focus sampling effort on the "hot-spot" areas. Ecological variability has routinely constrained the ability to ascribe significant differences to cause or distinguish differences due to chance in environmental assessments of effects of human disturbance on ecosystems (Boesch, 1984).

Reference and disposal site sediment chemical data from the three study areas resembled each other quite closely in most cases. Few of the chlorinated hydrocarbon pesticide concentrations were above detection limits. Polychlorinated biphenyls (PCB's) were not detected in any of the samples. Concentrations of trace metals measured in the sediments were within the ranges of contaminants reported for northern Gulf estuaries (NOAA, 1991; Summers et al., 1993 b). Chromium and copper were lower in nearshore samples than in samples from offshore at both Mermentau River disposal and reference sites. Available data on particle size and organic carbon for this study area provide little insight into the differences in concentrations of chromium and copper. Bulk sediment concentrations of trace metals were not unusually high compared to similar fine-grained coastal/estuarine sediments in the Northern Gulf estuaries (Brecken-Folse and Duke, unpublished report, and NOAA, 1991). However, bulk sediment concentrations typically do not allow useful estimates of biologically available metals without additional sediment chemistry (e.g., acid volatile sulfide concentration; Di Toro et al., 1990).

Ecosystem Dynamics and The Illusion of Ecosystem Recovery

Classical environmental science definition of ecosystem recovery is the return of a system to its pre-existing state (i.e., structurally, metabolically, and dynamically; Landis et al., 1993). The question at hand is what criteria will reliably allow the determination of disturbance and recovery? The answer to this question is not as straightforward as traditionally perceived. Evidence exists that ecological disturbance is expressed at various hierarchical levels (e.g., biological organizational scale--species *vs* population *vs* community and process scale--spatial and temporal patterns; Pickett et al., 1989). One consequence is that an arbitrary conversion of disturbance to non-disturbance may simply occur by shifting the scale of observation. For example, we detected, in some cases, significant differences at the scale of stations but not at the scale of sites. A biological scale-shift might involve simply changes in taxonomic resolution between reference and disposal sites. A possible remedy is to use higher taxonomic levels than species where adequate reference locations are unavailable or the taxonomy is impractical to resolve to comparable units (Warwick, 1993).

Many biotic and abiotic factors govern the composition of an ecosystem prior to and after a disturbance event. For benthic macrofauna, substrate type, changes in biogeochemical processes (e.g., indicated by redox profiles), "founder effect" or which species arrives and colonizes first after a stress, changes in the water current field, changes in exposure to toxic organic contaminants and metals, and the patchy nature of the recruitment process are but a few. Since each of the initial conditions is likely to be different from those that lead to the original system, it is very improbable that the so-called recovered system will be identical to the pre-disturbance state. Besides being statistically significant, by what criteria do we measure departure from the identical state and what importance do we attach to the differences? This question reinforces the idea that statistically significant differences need to be interpreted ecologically. The apparent recovery or movement of the disturbed system towards the reference or pre-disturbed state may be an illusion, because the systems may be moving in opposite directions and simply pass by similar endpoints during certain time intervals (see Figure 10 in Landis et al., 1993). Of course, if variance increases in the measurement parameters in reference or disturbed system, then the ability to detect a statistically significant effect between the reference and disturbed system would decrease at comparable sampling efforts. Additional theoretical complications may arise from complexity in nonlinear ecological systems (Hastings et al., 1993). There are strong theoretical reasons that recovery to a reference state may be highly unlikely because of small, often unmeasurable initial conditions (May and Oster, 1978). What is needed is a framework that allows unambiguous definitions of disturbance and the object being disturbed. The concept of "minimal structure" appears to provide greater generality and objectivity in applying the concept of disturbance to ecological systems. Pickett et al. (1989) provide an explanation of this concept. These brief comments are meant to raise awareness that ecological "impact" assessment has many dimensions that may be unaddressed by simple empirical comparisons of a few structural indicators of community change. Most of the issues raised are still under active research and further clarification will depend on research progress.

In our study, we did not find many indications of gross effects even though numerous statistically significant effects were detected.

Recommendation for Improvement in Sampling Design

In recent years, environmental impact assessment has made considerable improvement in sampling strategies designed to test hypotheses regarding effects of putative environmental stressors (Eberhardt and Thomas, 1991; Underwood and Peterson, 1988). Progress is still limited by theoretical uncertainties discussed under the section on Illusion of Ecosystem Recovery. A complete rationale for the subject at hand could involve a monograph level presentation (e.g., Green, 1979) which is beyond the scope of this paper. Our purpose here is to provide a brief description for conceptual improvement in future studies designed to assess the extent and magnitude of ecological effects of dredged material disposal on benthic macrofaunal communities in the three study areas reported herein. The approach is generalizable to other forms of ecological disturbance and other coastal ecosystems.

Ideally, one would like to employ an "experimental" design that optimized on the identification of causal relations between effects and probable cause (s). More generally, an approach that might reliably detect environmental disturbance and distinguish between natural variability and human influence is known as the BACI (Before-After-Control-Impact) approach (Underwood, 1991, and Underwood, 1994). Implicit in the approach is the need for an initial synthesis or scoping of ecosystem structure and function (i.e., scaled process focus) to determine what ecological information is relevant to the presumed problem. For example, the spatial extent of the disposal area (i.e., will the deposited dredged material be spatially homogenous in texture and will the depth of deposited material be reliably estimated?) should be estimated. Precise answers may not be possible, but an awareness of the need to estimate relevant answers to such questions is critical.

The ability to conduct pre-disposal sampling is very important to environmental assessment projects. The likelihood of local differences in benthic macrofaunal community structure to occur, especially prior to dredged material disposal, reinforces the need for such information. For example, suppose the response variables used to assess ecological change already were an order of magnitude larger at the disposal site compared to the reference site at initiation of dredged material disposal. The effect might be to reduce the quantitative value of the variables to that of the reference area. An effect would have occurred, but it would be undetected without pre-disposal information available from reference and disposal sites.

The BACI approach recommends multiple reference areas to increase discriminating power even when only a single treatment site is available. In the present study, we sampled only one reference site per study area. The papers by Underwood (*op cit*) describe the utility of temporal sampling that helps detect effects of exogenous factors that can confound interpretation of results. Elements of the BACI approach were used in the studies on brine effects at the West Hackberry site (Gaston and Weston, 1983).

Future work should randomly sample each depth zone instead of randomly sampling a central location within each depth zone (e.g., nearshore reference station). This would provide a more appropriate estimate of spatial variability in the response variables for comparison and minimize problems of spatial pseudoreplication (Hurlburt, 1984). Problems of temporal pseudoreplication should also be addressed (Stewart-Oaten et al., 1986). Non-parametric multivariate analyses should be considered, especially when data transformations are unable to satisfy parametric statistical assumptions. Categorical presence/absence data necessitate non-parametric statistical analyses (Clarke, 1993; Warwick and Clarke, 1993). A variance test for species associations looks promising for future community level analyses (McCulloch, 1985).

SUMMARY

Overall, the evidence was weak to support the hypothesis that gross differences in benthic macrofaunal community structure were present and that the differences were associated with dredged material disposal. The numbers of taxa present at these Atchafalaya River, Freshwater Bayou, and Mermentau River reference and disposal sites were: 38 and 40, 21 and 18, and 14 and 17, respectively. The opportunistic polychaetes, *Mediomastus californiensis, Paraprionospio pinnata*, and another polychaete, *Pseudoeurythoe ambigua*, dominated abundance at the Atchafalaya River and Freshwater Bayou study areas (i.e., average abundance > 5.0 individuals 0.05 m^{-2} in, at least, either a reference or a disposal site). Other abundant taxa at the Atchafayala River study area included the polychaetes, *Spiochaetopterus oculatus* and *Glycinde solitaria*, the bivalve, *Mulinia lateralis*, and the rhyncocoele, *Nemertea* sp. A. The two polychaetes, *P. pinnata* and *P. ambigua,* dominated abundance at the Mermentau River study area.

Distribution and abundance of benthic macrofauna at the Freshwater Bayou study area differed substantially from that of the other two study areas. Reference and disposal sites at the Freshwater Bayou study area did not differ significantly from each other in average abundance and taxa richness. However, the combined abundance of nearshore reference and disposal sites ($\bar{x} = 10.0$) differed significantly from the combined middepth ($\bar{x} = 25.6$) and offshore sites but the middepth and offshore sites did not differ significantly. A similar significance pattern was observed for taxa richness. Of the two dominant taxa that differed significantly between sites, abundance of the polychaete, *P. ambigua,* was greater at the disposal site but abundance of another polychaete, *M. californiensis,* was greater at the reference site. Measured sediment physical properties (e.g., particle size) and organic contaminants and metals provided no clues to explain the distributional pattern and abundance of the individual benthic macrofaunal taxa.

At the Atchafalaya study area, variability in taxa richness and numerical abundance within and among some reference and disposal stations suggests lack of a consistent increase or decrease in these parameters at reference and disposal sites. Four out of a total of 17 numerically abundant taxa at the Atchafalaya River reference site averaged significantly more individuals than occurred at the disposal site. The polychaete, *Streblospio benedictii*, was the only species to average significantly more individuals at the disposal than at the reference site. Variability in abundance within and among reference and disposal stations was high which indicated a lack of consistency in individual taxa abundance at the physical scale of stations employed in this study. The lack of consistency suggests that the possible differential influence of dredged material disposal on selected individual benthic macrofaunal taxa cannot be ruled out. However, the overall magnitude of these differences appeared relatively small between reference and disposal sites. Benthic macrofaunal response patterns did not correlate with the measured sediment physical factors, hydrographic factors or sediment chemical constituents.

Five of seven relatively abundant taxa averaged significantly more individuals at Mermentau River disposal than at the reference site. Average numerical abundance was greater at the disposal site. Taxa richness showed a statistically significantly higher value at the middepth disposal station than at the middepth reference station. This parameter did not differ significantly between the other paired reference and disposal stations. Although the response parameters were generally consistent between reference and disposal sites, we believe further work would allow more informed judgement of potential effect or lack thereof.

The addition of the seventh replicate sample per station seldom added a new taxon to the inventory. In most cases, four or five replicates captured all but probably the most rare species. The sampling power analysis revealed that approximately 25 to 50 replicate samples would be required to achieve an accuracy of the mean for abundance at +/-10%. The need for this level of accuracy should be determined on a case-by-case basis because of the obvious high cost for this level of sampling. Relaxation of the accuracy requirement to 20% substantially reduced the number of samples required to achieve a significant P-value $\leq$ 0.05.

ACKNOWLEDGMENTS

The following individuals are thanked for their laboratory and field assistance in the project: Barbara Albrecht, Brian Dorn, Diane Folse, John Harmuth, Courtney Head, Guy Herring, Rob Holbrook, Andrew Kelly, Vicki Kramer, Shannon Phifer, Bob Quarles, Sean Stangeland, Roman Stanley, David Whiting, and Ruth Yoakum. Wallace T. Gilliam provided the gas chromatographic and mass spectral analyses of sediment samples. Captain Dwight Paine and crew of the Ocean Survey vessel Peter W. Anderson exhibited a high degree of professionalism through their assistance in making the field work a success. Maureen Stubbs and Valerie Coseo are thanked for typing the manuscript. George W. Ryan assisted with some of the preliminary statistical analyses. Tom Poe and Steve Embry provided the graphical support. Andrew McErlean, Larry Goodman, C. McKenney, John Valentine and Linda Mathies are thanked for helpful comments on the manuscript. Mention of trade names or commercial products in this report does not constitute endorsement by the U.S. Environmental Protection Agency.

LITERATURE CITED

Boesch, D.F. 1984. Introduction: Field assessment of marine pollution effects: The agony and the ecstasy, pp. 643-646. *In*: H.H. White, ed. Concepts in Marine Pollution Measurements. A Maryland Seagrant Publication, University of Maryland, College Park, MD.

Boesch, D.F. and R. Rosenberg. 1981. Response to stress in marine benthic communities, pp. 179-200. *In*: G.W. Barrett and R. Rosenberg, eds. Stress Effects on Natural Ecosystems. Wiley and Sons.

Brecken-Folse, J. and T.W. Duke. undated. The Gulf of Mexico Sediments Quality Inventory. U.S. Environmental Protection Agency, Office of Science and Technology, 401 M Street, SW, Washington, DC (Unpublished report).

Carriker. M.R. 1967. Ecology of estuarine benthic invertebrates: A perspective, pp.442-487. In: G.H. Lauff, ed. Estuaries. Am. Assoc. for Adv. Sci. Publ. No. 83, Washington, DC.

Chiba, K. and Y. Oshima. 1957. Effects of suspended particles on the pumping and feeding of marine bivalves, especially of Japanese little-neck clam. Bull. Jpn. Soc. Sci. Fish. 23: 348-354.

Clarke, K.R. 1993. Non-parametric multivariate analyses of changes in community structure. Aust. J. Ecol. 18: 117-143.

Cordone, A.J. and D.W. Kelley. 1961. The influence of inorganic sediment on the aquatic life of streams. Calif. Fish and Game. 47(2): 189-228.

Cronin, L.E. 1967. The role of man in estuarine processes, pp. 667-689. *In*: G.H. Lauff, ed. Estuaries. Publ. No. 83. AAAS, Washington, DC.

Dauer, D.M. and J.L. Simon. 1976. Repopulation of the polychaete fauna of an intertidal habitat following natural defaunation: Species equilibrium. Oecologia 22: 99-117.

Davis, H.C. 1960. Effects of turbidity-producing materials in sea water on eggs and larvae of the clam (*Mercenaria mercenaria*). Biol. Bull. 118 (1): 48-54.

DeRouen, L.R., R.W. Hann, D.M. Casserly, C. Giammona, and V.J. Lascara. 1983. West Hackberry Brine Disposal Project Pre-Discharge Characterization. DOE Strategic

Petroleum Reserve Project Management Office, DOE/PO/10288, U.S. DOE Publication, Washington, DC.

Di Toro, D.M., J.D. Mahony, D.J. Hansen, K.J. Scott, M.B. Hicks, S.M. Mayr, and M.S. Redmond. 1990. Toxicity of cadmium in sediments: The role of acid volatile sulfide. Environ. Toxicol. and Chem. 9: 1487-1502.

Duarte, C.M. 1991. Variance and the description of nature. Chapter 15, pp. 301-318. *In*: Comparative Analyses of Ecosystems--Patterns, Mchanisms, and Teories. J. Cole, G. Lovett and S. Findlay, eds. Springer-Verlag, New York.

Eberhardt, L.L. and J.M. Thomas. 1991. Designing environmental field studies. Ecol. Monogr. 61: 53-73.

Eckblad, J.W. 1991. How many samples should be taken? BioScience 41(5): 346-348.

Flemer, D.A., W.L. Dovel, H.T. Pfitzenmeyer and D.E. Ritchie, Jr. 1968. Biological effects of spoil disposal in Chesapeake Bay. J. Sanit. Eng. Div. Proc., ASCE, Vol. 94 (SA4): 683-706.

Flint, R.W. and J.S. Holland. 1980. Benthic infaunal variability on a transect in the Gulf of Mexico. Estuarine and Coastal Marine Science 10: 1-14.

Folk, R.L. 1980. Petrology of Sedimentary Rocks. Hemphill Publ. Co., Austin, TX.

Franz, D.R. and W.H. Harris. 1988. Seasonal and spatial variability in Jamaica Bay, New York--An urban estuary. Estuaries 11: 15-28.

Gaston, G.R. 1985. Effects of hypoxia on macrobenthos of the inner shelf off Cameron, Louisiana. Estuarine Coastal and Shelf Sci. 20: 603-613.

Gaston, G.R. and J.C. Nasci. 1988. Trophic structure of macrobenthic communities in the Calcasieu Estuary, Louisiana. Estuaries 11(3): 201-211.

Gaston, G.R., P.A. Rutledge and M.L. Walther. 1985. The effects of hypoxia and brine on recolonization by macrobenthos off Cameron, Louisiana (USA). Contrib. Mar. Sci. 28: 79-93.

Gaston, G.R. and D.P. Weston. 1983. Benthos. Chapter 6, 153 pp. *In*: L.R. DeRouen, R.W. Hann, Jr., D.M. Casserly, C. Giammona and V.J. Lascara, eds. West Hackberry Strategic

Petroleum Reserve Site Brine Disposal Monitoring. U.S. Department of Energy Publication, Washington, DC.

Gaston, G.R. and K.A. Edds. 1994. Long-term study of benthic communities on the continental shelf off Cameron, Louisiana: A review of brine effects and hypoxia. Gulf Research Reports 9 (1): 57-64.

Giammona, C.P. and R.M. Darnell. 1990. Environmental Effects of the Petroleum Reserve Program on Louisiana Continental Shelf Communities. Am. Zool. 30: 37-43.

Green, R.H. 1979. Sampling Design and Statistical Methods for Environmental Biologists. Wiley-Interscience Publ.

Gray, J.S. and T.H. Pearson. 1982. Objective selection of sensitive species indicative of pollution-induced change in benthic communities. I. Comparative methodology. Marine Ecology Progress Series 9: 111-119.

Grassle, J.F. and J.P. Grassle. 1974. Opportunistic life histories and genetic systems in marine polychaetes. J. Mar. Res. 32: 253-284.

Harper, D.E., Jr. 1977. Aquatic disposal field investigations, Galveston, Texas, Offshore Disposal Site. Appendix C: Investigation of the effects of dredging and dredged material disposal on offshore biota. Office, Chief of Engineers, U.S. Army, Washington, DC, Technical Report D-77-20.

Hastings, A., C.L. Hom, S. Ellner, P. Turchin, and H.C.J. Godfray. 1993. Chaos in ecology: Is Mother Nature a strange attractor? Annu. Rev. Syst. 24: 1-33.

Heck, K.L., Jr. and R.J. Horwitz. 1984. Statistical analysis of sampling data to assess impact in marine environments, pp. 233-245. *In* : H.H. White, ed. Concepts in marine pollution measurements. A Maryland Seagrant Publication. University of Maryland, College Park, MD.

Hurlburt, S.H. 1984. Pseudoreplication and the design of ecological field experiments. Ecol. Monogr. 54: 187-211.

Ingle, R.M. 1952. Studies on the effect of dredging operations upon fish and shellfish. Florida Bd. Conserv. Tech. Ser. 5: 26.

Ingle, R.M., A.R. Ceurvels, and R. Leinecker. 1955. Chemical and biological studies of the muds of Mobile Bay. Univ. Miami Mar. Lab. Contrib. 139. 14 pp.

Jones, J.I. and S.E. Williams. 1979. The distribution and origin of bottom sediments in Timbalier Bay, Louisiana, and the adjacent offshore area, pp. 301-221. *In*: C.H. Ward, M.E. Bender and D.J. Reish, eds. Effects of Oil Drilling and Prediction in a Coastal Environment. Vol. 65 (Nos. 4 and 5). Rice University Studies, Houston, TX.

Kirby, C.J., J.W. Keeley and J. Harrison. 1975. An overview of the technical aspects of the Corps of Engineers National Dredged Material Research Program, pp. 523-536. *In*: L.E. Cronin, Vol. I. Chemistry, Biology, and the Estuarine System. Estuarine Research. Academic Press, New York.

Landis, W.G., R.A. Matthews, A.J. Markiewicz, and G.B. Matthews. 1993. Multivariate analysis of the impacts of the turbine fuel JP-4 in a microcosm toxicity test with implications for the evaluation of ecosystem dynamics and risk assessment. Ecotoxicology 2: 271-300.

Loosanoff, V.L. and F.D. Tommers. 1948. Effects of suspended silt and other substances on the rate of feeding of oysters. Science 107: 69.

Lunz, G.R. 1938. Oyster culture with reference to dredging operations in South Carolina. U.S. Engineers' Office, Charleston, S.C. (Part I) 43 pp.

Mackin, J.G. 1961. Canal dredging and siltation in Louisiana Bays. Publ. Inst. Mar. Sci. 7: 262-319.

Markey, J.W. and H.D. Putnam. 1976. A study of the effects of maintenance dredging on selected ecological parameters in the Gulfport Ship Channel, Gulfport, Mississippi, pp. 821-833. *In*: P.A. Krenkel, J. Harrison and J.C. Burdick III. Proc. of Specialty Conference on Dredging and its Environmental Effects. Mobile, Alabama, ASCE, New York.

May, R.M. and G.F. Oster. 1978. Bifurcations and dynamical complexity in simple ecological models. Am. Nat. 110: 573-599.

McCulloch, C.E. 1985. Variance tests for species association. Ecology 66: 1676-1681.

Moore, J.C., E.M. Lores and J. Forester. 1994. Chemical analyses of sediment from three ocean dredged material disposal sites and reference sites in Mermentau River, Freshwater Bayou and Atchafalaya River, Louisiana. Report to U.S. EPA Region VI, Dallas, Texas.

NOAA. 1991. National Status and Trends Program for Marine Environmental Quality--Second Summary of Data on Chemical Contaminants in Sediments from the National Status and Trends Program. NOAA Technical Memorandum NOS OMA 59, 29 pp., 4 appendices.

NOAA. 1992. Mussel Watch--Recent trends in coastal environmental quality. U.S. Department of Commerce, NOAA, NOS, Washington, DC.

Oetking, P., R. Back, R. Watson, and C. Merks. 1979. Surface and shallow subsurface sediments of the nearshore Continental Shelf of South Central Louisiana, pp. 181-200. *In*: C.H. Ward, M.E. Bonder and D. J.Reish, eds. Effects of Oil Drilling and Production in a Coastal Environment. The Offshore Ecology Investigation. Rice University Studies, 65(Nos. 4 and 5), Houston, TX.

Parker, R.H., A.L. Crowe and L.S. Bohme. 1980. Biological/Chemical Survey of Texoma and Capline Sector Salt Dome Brine Disposal Sites Off Louisiana, 1978-1979. Vol I. Benthos. NOAA Technical Memorandum, NMFS-SEFC-25. Final report to DOE, NMFS, Southeast Fisheries Center, Galveston, TX.

Pearson, T.H. and R. Rosenberg. 1978. Macrobenthic succession in relation to organic enrichment and pollution of the marine environment. Oceanogr. Mar. Biol. Ann. Rev. 16: 229-311.

Pfitzenmeyer, H.T. 1970. Gross physical and biological effects of overboard spoil disposal in upper Chesapeake Bay. Project C, Benthos, pp. 26-38. NRI Special Report No. 3. University of Maryland, College Park, MD.

Pickett, S.T.A., J. Kolasa, J.J. Armesto, and S.L. Collins. 1989. The ecological concept of disturbance and its expression at various hierarchial levels. Oikos 54: 129-136.

Probert, P.K. 1984. Disturbance, sediment stability, and trophic structure of soft-bottom communities. J. Mar. Res. 42: 893-921.

Rabalais, N.N., R.E. Turner, W.J. Wiseman, Jr. and D.F. Boesch. 1986. Hydrographic, biological, and nutrient characteristics of the water column on the Louisiana Shelf, July and September, 1985. Louisiana Universities Marine Consortium Data Report No. 3.

Rabalais, N.N. 1992. An updated summary of status and trends in indicators of nutrient enrichment in the Gulf of Mexico. Publ. No. EPA/800-R-92-004, U.S. EPA, Office of Water, Gulf of Mexico Program, Stennis Space Center, MS.

Santos, S.L. and J.L. Simon. 1980. Response of soft-bottom benthos to annual catastrophic disturbance in a South Florida estuary. Mar. Ecol. Prog. Ser. 3: 347-355.

SAS, 1989a. Users Guide: Basics. SAS Institute, Inc. Cary, NC, 1290 pp.

SAS, 1989b. Users Guide: Statistics. SAS Institute, Inc. Cary, NC, 957 pp.

Sokal, R.R. and F.J. Rohlf. 1981. Biometry (2nd ed.). W.H. Freeman and Company, San Francisco.

Sousa, W.P. 1984. The role of disturbance in natural communities. Ann. Rev. Ecol. Syst. 15: 353-391.

Stewart-Oaten, S., W.W. Murdoch, and K.R. Parker. 1986. Environmental impact assessment: "Pseudoreplication" in time? Ecology 67: 929-940.

Summers, J.K., J.M. Macauley, V.D. Engle, G.T. Brooks, P.T. Heitmuller, and A.M. Adams. 1993 a. Louisianian Province Demonstration Report, EMAP- Estuaries-1991. U.S. Environmental Protection Agency, Environmental Research Laboratory, Gulf Breeze, FL. EPA/620/R-94/001.

Summers, J.K., J.M. Macauley, P.T. Heitmuller, V.D. Engle, A.M. Adams, and G.T. Brooks. 1993 b. Annual Statistical Summary: EMAP-Estuaries Louisianian Province -1991. U.S. Environmental Protection Agency, Office of Research and Development, Environmental Research Laboratory, Gulf Breeze, FL. EPA/600/R-93/001.

Underwood, A.J. 1994. On beyond BACI: Sampling designs that might reliably detect environmental disturbances. Ecol. Appl. 4: 3-15.

Underwood, A.J. and C.H. Peterson. 1988. Towards an ecological framework for investigating pollution. Mar. Ecol. Prog. Ser. 46: 227-234.

Underwood, A.J. 1991. Beyond BACI: Experimental designs for detecting human environmental impacts on temporal variations in natural populations. Aust. J. Mar. Freshwater Res. 42: 569-587.

U.S. EPA. 1990a. Atchafalaya River Bar Channel Ocean Dredged Material Disposal Site Designation. Supplemental EIS (Draft). U.S. EPA Region VI, Dallas, TX. EPA/906/12-90-009.

U.S. EPA 1990b. Freshwater Bayou Ocean Dredged Material Disposal Site Designation. EIS (Draft). U.S. EPA Region VI, Dallas, TX. EPA/906/07-90-008.

U.S. EPA 1990c. Mermentau River Ocean Dredged Material Disposal Site Designation. EIS (Draft). U.S. EPA Region VI, Dallas, TX. EPA/906/05-90-004.

Ward. C.H., M.E. Bender and D.S. Reish, eds. 1979. The Offshore Ecology Investigations. Effects of oil drilling and production in a coastal environment. Rice University Studies 65 (Nos. 4 and 5), Houston, TX.

Warner, J.S. 1976. Determination of aliphatic and aromatic hydrocarbons in marine organisms. Analytical Chemistry 48 (3): 578-583.

Warwick, R.M. 1993. Environmental impact studies on marine communities: Pragmatical considerations. Aust. J. Ecol. 18: 63-80.

Warwick, R.M. and K.R. Clarke. 1993. Comparing the severity of disturbance: A meta-analysis of marine macrobenthic community data. Mar. Ecol. Prog. Ser. 92: 221-231.

Weston, D.P., and G.R. Gaston. 1982. Benthos, Chapter 5. pp. 5-1 to 5-86. *In*: L.R. DeRouen, R.W. Hann, D.M. Casserly and C. Giammona, eds. West Hackberry Brine Disposal Project Pre-Discharge Characterization. DOE, Washington, DC.

Wilson, J.N. 1957. Effects of turbidity and silt on aquatic life. Biological Problems in Water Pollution. U.S. Dep. Health Educ. and Welfare. Robert A. Taft Sanitary Engineering Center, p. 235-239.

Wright T.D., D.B. Mathis and J.M. Brannon. 1978. Aquatic disposal field investigations, Galveston, Texas, Offshore Disposal Site. Evaluative Summary, Technical Report D-77-20, Office, Chief of Engineers, U.S. Army, Washington, DC, 89 pp. (NTIS No. AD-A061844).

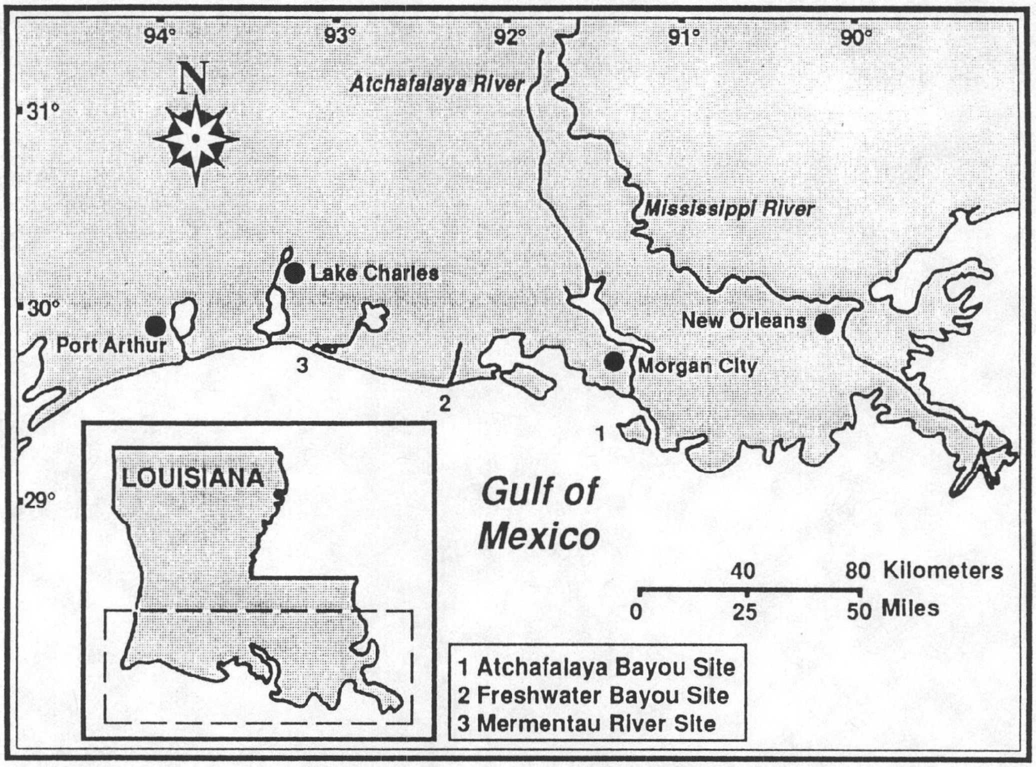
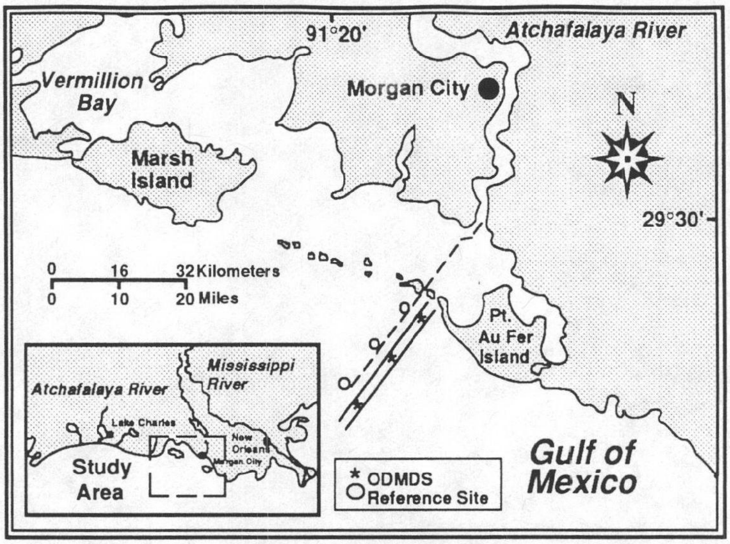
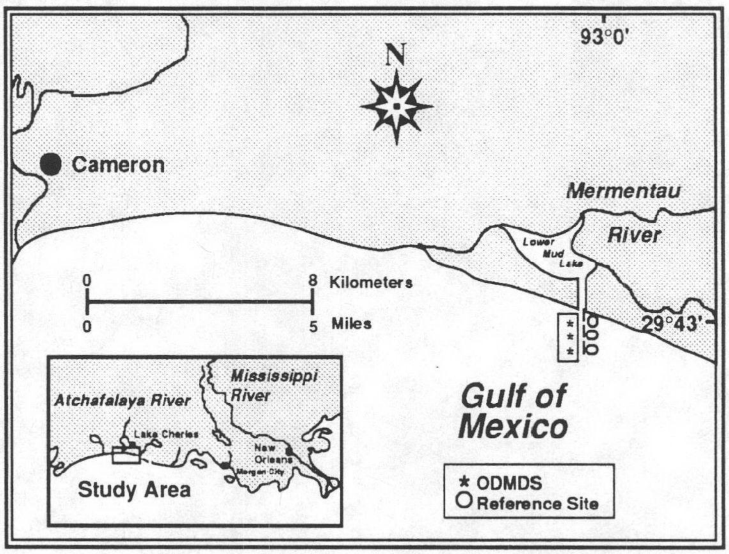
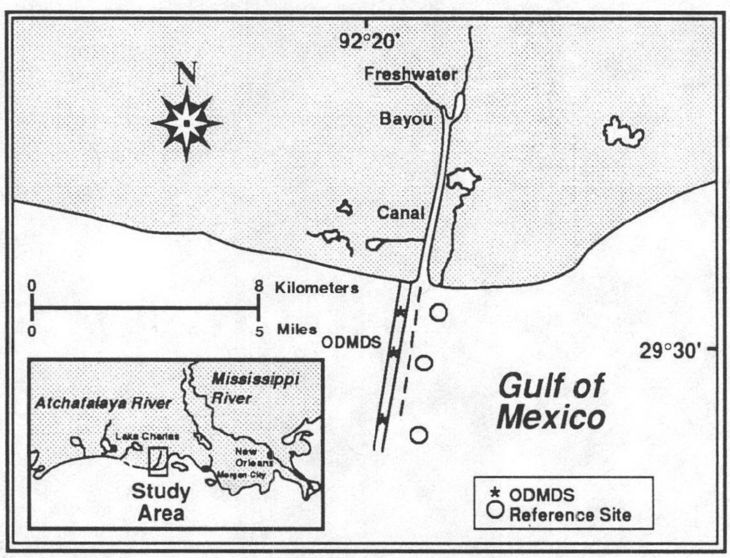


Figure 1. Map of the study areas for benthic macrofaunal community analyses.

Figure 2. Map of individual study sites including Mermentau River, Freshwater Bayou and Atchafalaya River showing sampling locations.







EPA/RL#94011-01

Figure 3. Representative taxa - area curves showing cumulative number of taxa with increased number of replicate samples from given station.

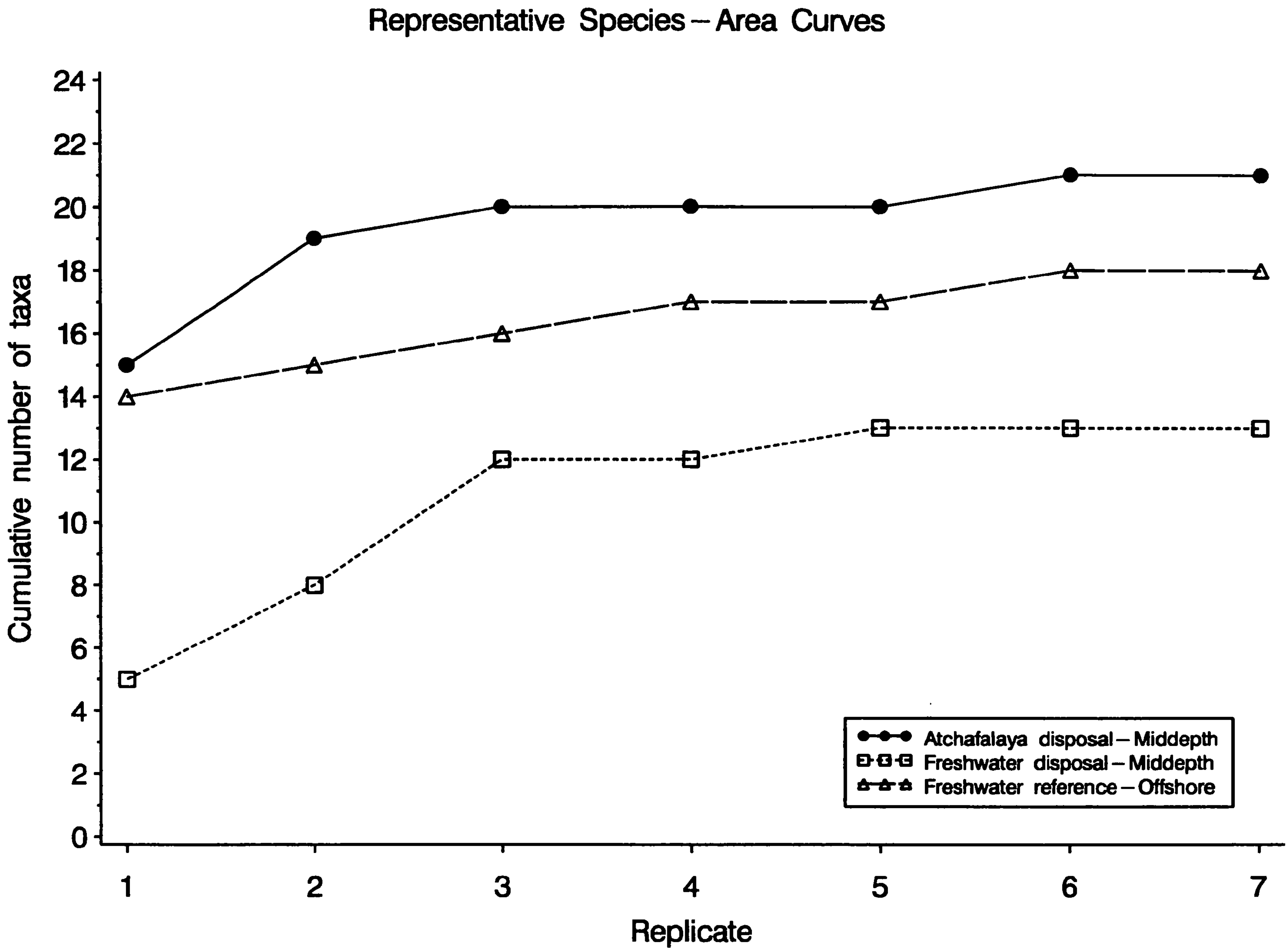
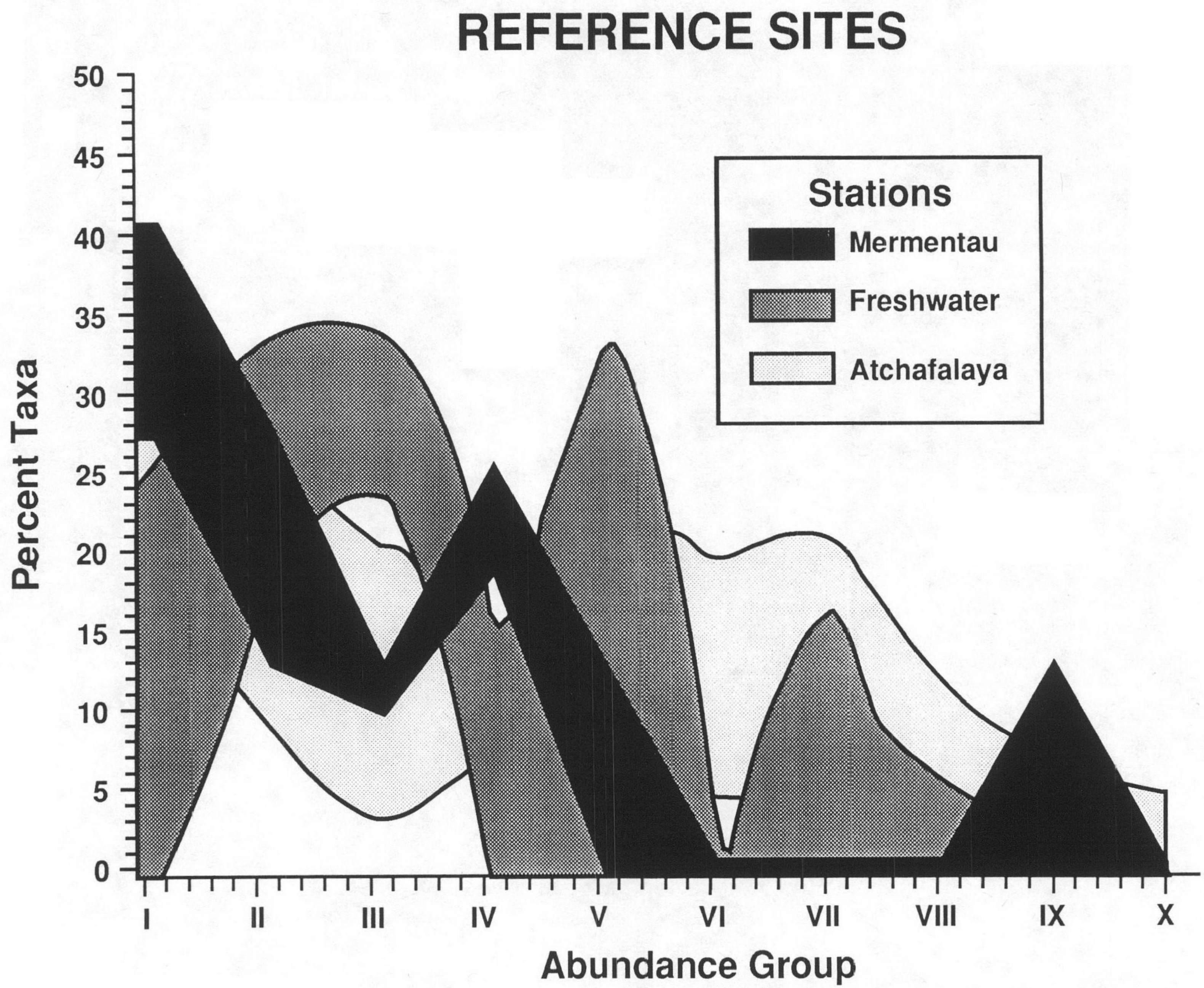
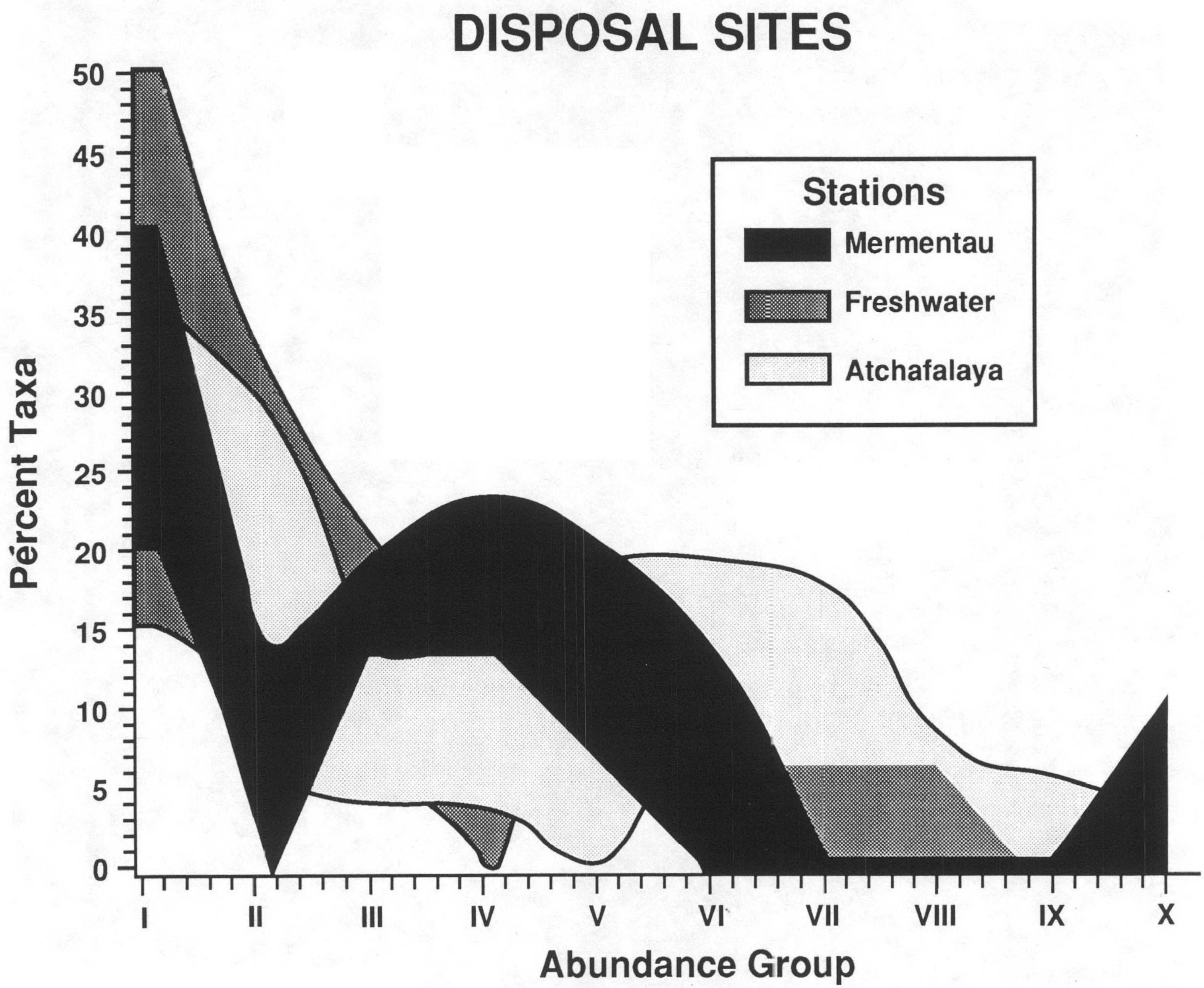

Representative Species – Area Curves
Cumulative number of taxa
0
2
4
6
8
10
12
14
16
18
20
22
24
Replicate
1
2
3
4
5
6
7
Atchafalaya disposal – Middepth
Freshwater disposal – Middepth
Freshwater reference – Offshore

Figure 4. Relationship between percent taxa of benthic macroinvertebrates and abundance group (I = 1 individual, II = 2 to 3 individuals, III = 4 to 7 individuals, IV = 8 to 15 individuals, etc.). Results for reference and disposal sites are shown separately.

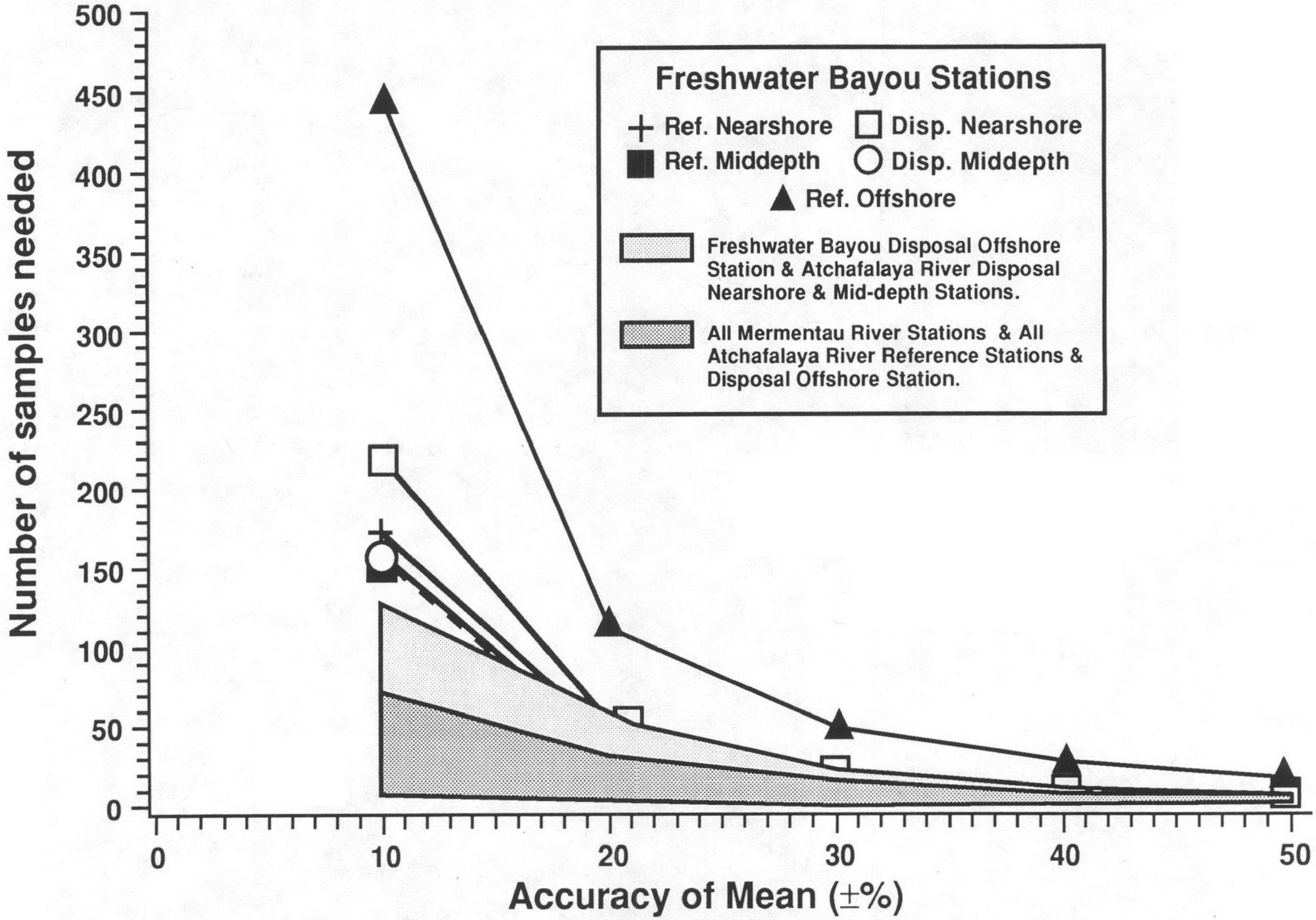


EPA/RL#94011-04



EPA/RL#94011-05

Figure 5. Number of samples required to sample benthic macroinvertebrates within a certain percent accuracy of the mean at P = 0.05.



EPA/RL#94011-03

Table 1. Latitude and longitude coordinates for Louisiana benthic community analysis sampling locations.

Site	Station		Disposal	Reference
Atchafalaya	Nearshore	N	29°19'25"	29°19'58"
		W	91°25'28"	91°26'07"
	Midsection	N	29°16'09"	29°16'36"
		W	91°28'29"	91°29'00"
	Offshore	N	29°13'58"	29°12'59"
		W	91°31'16"	91°32'25"
Freshwater	Nearshore	N	29°30'50"	29°30'53"
		W	92°21'19"	92°20'56"
	Midsection	N	29°30'11"	29°30'14"
		W	92°21'41"*	92°20'58"
	Offshore	N	29°28'48"	29°28'31"
		W	92°19'59"	92°18'10"
Mermentau	Nearshore	N	29°42'22"	29°42'17"
		W	93°08'17"	93°07'05"
	Midsection	N	29°41'05"	29°41'05"
		W	93°08'01"	93°07'11"
	Offshore	N	29°41'49"	29°41'53"
		W	93°07'35"*	93°07'00"*

* Loran malfunction probable

Table 2. Louisiana Benthic Community Analysis Physical Parameters

CHANNEL	SITE	STATION	SAL (ppt)	TEMP (°C)	DO (mg/liter)	pH	DEPTH (m)	PARTICLE SIZE %>63μm	%≤63μm	% weight loss on ignition
Atchafalaya	Reference	Nearshore	8.0	23 5	7.0	8.6	1.9	60.01	39.99	2 22
Atchafalaya	Disposal site	Nearshore	9.5	23.7	7.5	8 8	1.1	56.65	43.35	3.50
Atchafalaya	Reference	Middepth	17.6	23 3	6.2	8.8	2.1	1 89	98.11	6 62
Atchafalaya	Disposal site	Middepth	19.1	23.3	6.3	8.3	2.0	9.84	90.16	6.10
Atchafalaya	Reference	Offshore	24.4	22.9	6.3	7.4	3 3	3.37	96.63	5.48
Atchafalaya	Disposal site	Offshore	25.8	22.9	6.5	7.4	3.5	3.43	96.57	5.98
Freshwater	Reference	Nearshore	26.5	21.8	6.7	8.1	1.6	2.97	97.03	9 12
Freshwater	Disposal site	Nearshore	26.6	21.7	6.7	8.0	1.7	4.46	95.54	6.74
Freshwater	Reference	Middepth	26.0	22.0	6.8	7 9	2.5	3.11	96 89	7.53
Freshwater	Disposal site	Middepth	26.2	22.1	6.7	7 9	2.3	24.75	75.25	6.98
Freshwater	Reference	Offshore	27.2	22.8	6.7	8.2	4 0	0 49	99.51	11.63
Freshwater	Disposal site	Offshore	27.2	22.6	6.6	8.1	3 0	0 55	99.45	10.81
Mermentau	Reference	Nearshore	30 6	22.3	6.6	8.2	1.5	17 55	82.45	5.84
Mermentau	Disposal site	Nearshore	30.5	22.5	6.3	8.0	2.0	40.79	59.21	5.37
Mermentau	Reference	Middepth	30.5	22.0	5.9	8.2	1.8	6.74	93.26	8.20
Mermentau	Disposal site	Middepth	30.5	22.2	6.7	8.2	3.0	10.91	89.09	6 41
Mermentau	Reference	Offshore	30 5	22.0	6.3	8.3	2.6	0 92	99.08	9 73
Mermentau	Disposal site	Offshore	30.5	22.0	6 9	8.2	3.7	12.15	87 85	6.50

Table 3. Average percentage recovery of selected chlorinated pesticides and PCB spiked on samples of sediment. Method detection limits are also shown (From Moore et al. 1994).

Compound	Spike Concentration ($\mu g/g$)	N	Average Percentage Recovery and Standard Deviation	Method Detection Limit ($\mu g/g$ wet weight)
Aldrin	0.10	3	76 ± 2.0	0.010
BHC Isomers				
Alpha	a			0.010
Beta	a			0.010
Gamma (lindane)	0.10	4	25 ± 3.7	0.010
Chlordane (alpha)	0.10	12	55 ± 18	0.010
Chlorpyrifos (Dursban)	a			0.010
DDE (P,P')	0.10	11	91 ± 14	0.010
DDD (P,P')	0.10	6	71 ± 5.3	0.010
Dieldrin	0.10	11	87 ± 15	0.010
Endrin	a		a	0.010
Endosulfan I	0.10	11	82 ± 21	0.010
Endosulfan II	0.10	12	96 ± 7.0	0.010
Endosulfan Sulfate	a			0.010
Heptachlor	0.10	3	76 ± 2.0	0.010
Heptachlor epoxide	0.10	3	67 ± 1.0	0.010
Hexachlorobenzene	0.10	3	68 ± 2.6	0.010
Methoxychlor	a			0.010
Mirex	0.10	12	87 ± 6.4	0.010
PCBs[b]	0.10	11	88 ± 4.7	0.010
Toxaphene	a			0.010

ND = Not detected

[a] Analytes were not spiked for recovery.
[b] Percentage recovery was based on specific congener analysis.

Table 4. Concentration of selected metals in three replicate sediment samples from Atchafalaya River, Freshwater Bayou and Mermentau River dispersal and reference sites.

		Concentrations in μg/g wet weight								
Sediment Location	Depth	As[A]	Cd	Cr	Cu	Hg	Ni	Pb[A]	Se	Zn
Atchafalaya River										
Disposal Site	Nearshore	11	ND	3.8	4.0	ND	10	7.5	ND	16
	Mid-depth	34	ND	11	14	ND	22	18	ND	31
	Offshore	34	ND	11	12	ND	21	20	ND	31
Reference Site	Nearshore	11	ND	4.5	4.4	ND	11	6.8	ND	13
	Mid-depth	42	ND	14	18	ND	29	23	ND	43
	Offshore	25	ND	10	10	ND	20	16	ND	28
Freshwater Bayou										
Disposal Site	Nearshore	29	ND	11	12	ND	21	17	ND	32
	Mid-depth	31	ND	11	12	ND	24	24	ND	42
	Offshore	52	ND	18	17	ND	32	38	ND	37
Reference Site	Nearshore	49	ND	17	14	ND	27	24	ND	32
	Mid-depth	50	ND	18	17	ND	30	27	ND	39
	Offshore	43	ND	14	13	ND	28	28	ND	37
Mermentau River										
Disposal Site	Nearshore	21	ND	5.8	5.5	ND	14	14	ND	29
	Mid-depth	34	ND	12	11	ND	22	20	ND	35
	Offshore	33	ND	11	12	ND	22	20	ND	36
Reference Site	Nearshore	39	ND	13	11	ND	24	19	ND	34
	Mid-depth	41	ND	15	13	ND	25	20	ND	30
	Offshore	59	ND	20	15	ND	31	28	ND	35
		Method Detection Limit in μg/g wet weight								
		2.5	0.15	0.50	0.25	0.50	0.60	2.5	2.5	0.1

ND = Not detected; see Table 4 for detection limits.

[A] Usual background correction techniques could not be applied because of the intense interference; therefore, without subtracting background, lead and arsenic may be present but not in quantities greater than these shown.

Table 5. List of taxa with presence (X) or absence (-) at reference and dredge disposal sites.

Taxa	Atchafalaya		Freshwater		Mermentau	
	Reference	Disposal	Reference	Disposal	Reference	Disposal
ANNELIDA						
Ampharete americana	X	X	-	-	-	-
Cossura soyeri	X	X	X	X	X	X
Diopatra cuprea	X	X	-	-	-	-
Glycinde solitaria	X	X	-	-	X	X
Hesionidae (LPIL)	-	X	-	-	-	-
Lepidasthenia sp.	X	-	X	X	-	-
Magelona cf. *phyllisae*	X	X	-	-	X	X
Mediomastus californiensis	X	X	X	X	X	X
Neanthes micromma	-	X	X	X	-	X
Neanthes sp.	-	X	-	-	-	-
Onuphis emerita oculatus	-	X	-	-	-	-
Owenia fusiformes	X	X	-	-	-	-
Parandalia sp.	X	X	-	X	-	-
Paraprionospio pinnata	X	X	X	X	X	X
Pectinaria gouldii	X	-	-	-	-	-
Phyllodoce arena	X	X	-	-	-	-
Podarkiopsis levifusina	X	X	X	X	-	-
Polychaete sp A	-	-	-	-	-	X
Polychaete sp. B	X	-	X	-	-	-
Polychaete sp. C	X	-	-	-	-	-
Polydora sp.	X	X	-	-	-	-
Prionospio sp.	X	X	X	X	X	-
Pseudoeurythoe ambigua	X	X	X	X	X	X
Sigambra tentaculata	X	X	X	X	-	X
Spiochaetopterus oculatus	X	X	-	X	-	X
Streblospio benedicti	X	X	X	-	-	-
Oligochaeta (LPIL)	X	X	-	-	-	-

Table. 5 (continued) Taxa	Atchafalaya		Freshwater		Mermentau	
	Reference	Disposal	Reference	Disposal	Reference	Disposal
ARTHROPODA						
Balanus sp.	X	X	X	X	X	X
Callianassa sp.	X	-	-	-	-	-
Corophium sp.	X	-	X	-	X	-
Edotea montosa	-	X	-	-	-	-
Ischyroceridae (LPIL)	-	X	-	-	-	-
Melita sp.	-	X	-	-	-	-
Oxyurostylis smithi	X	X	-	-	-	-
Paguridae (LPIL)	-	-	-	X	-	-
Pinnixa sayana	X	X	X	X	-	X
Xanthidae (LPIL)	-	X	-	-	-	-
MOLLUSCA						
Mulinia lateralis	X	X	-	X	X	X
Nassarius vibex	-	X	X	-	-	-
Nuculana acuta	X	-	-	-	-	-
Rangia cuneata	X	-	-	-	-	-
Tellina texana	X	-	-	-	-	-
RHYNCOCOELA						
Cerebratulus lacteus	X	X	X	-	X	X
Lineus sp.	-	X	-	-	-	-
Micrura sp.	X	X	X	X	X	X
Nemertea sp. A (LPIL)	X	X	X	X	X	X
ECHINODERMATA						
Microphiopholis atra	-	X	-	-	-	-
PHORONIDA						
Phoronis sp.	X	X	X	-	-	-
HEMICHORDATA						
Enteropneusta (LPIL)	X	X	X	X	-	-
HOLOTHUROIDEA						
Holothuroidea (LPIL)	X	X	X	-	X	X
PLATYHELMITHES						
Stylochus ellipticus	X	X	-	-	-	-

Table 6. Species list by study area and site (reference and disposal) showing frequency of occurrence (Fq, location of occurrence by relative distance from shore; Loc, N = nearshore, M = middepth, O = offshore), mean abundance ($\bar{x}$) and standard deviation (sd) for 21 samples (0 05m² sample) per channel.

Species	Reference Fq	Loc	$\bar{x}$	sd	Disposal Fq	Loc	$\bar{x}$	sd
			ATCHAFALAYA RIVER					
Annelida								
Ampharete americana	8	N,O	0.7	1.10	5	N,O	0.7	1.98
Cossura soyeri	13	N,M,O	2.0	2 92	6	N,M,O	1 2	2.09
Diopatra cuprea	3	N,M,O	0 2	0 51	5	M	0 8	1.69
Glycinde solitaria	19	N,M,O	5.4	4.62	15	N,M,O	3 2	3.19
Hesionidae (LPIL)					1	O	0 1	0.38
Lepidasthenia sp.	1	O	0 1	0.75				
Magelona cf. *phyllis*	2	O	0.1	0.78	1	O	0.1	0 38
Mediomastus californiensis	21	N,M,O	40.8	33.48	21	N,M,O	40.7	50.43
Neanthes micromma					4	O	0.2	0.76
Neanthes sp.					3	N	0.1	0.42
Onuphis emerita oculatus					1	O	0.1	0.38
Owenia fusiformes	6	N,O	0 4	0 93	12	M,O	1.7	2 07
Parandalia sp	12	M,N,O	1.7	2.58	2	N	0.1	0.49
Paraprionospio pinnata	21	N,M,O	12.2	5.67	14	M,O	8.4	7 55
Pectinaria gouldii	1	N	0.1	0.76				
Phyllodoce arenae	2	N,M	0.1	0.36	1	O	0.1	0.38
Podarkiopsis levifusina	2	O	0.1	0.49	1	O	0.1	1 57
Polychaete sp. C (LPIL)	1	O	0 1	0 38				
Polychaete sp. B (LPIL)	1	M	0.1	0.38				
Polydora sp.	7	N,M,O	2.0	4 96	2	M,O	0.1	0.30
Prionospio sp.	2	O	0.2	1.33	3	O	0.2	0 95
Pseudoeurythoe ambigua	19	N,M,O	9 8	7 82	13	M,O	10.7	13 32
Sigambra tentaculata	9	M,O	1 3	2 00	8	M,O	1.6	2.48
Spiochaetopterus oculatus	20	N,M,O	7 2	8.15	13	M,O	7.0	10.06
Streblospio benedicti	2	N	1.4	5.51	7	N	2.5	4.35
Oligochaeta (LPIL)	16	N,M,O	4.0	4.09	11	N,M,O	2.0	3.42
Arthropoda								
Balanus sp	1	N	0.1	0.38	6	N,M,O	0.3	0.43
Callianassa sp.	1	N	0.1	0.38				
Corophium sp.	1	N	0.2	1.89				
Edotea montosa					1	N	0 1	0 75
Ischyroceridae (LPIL)					1	O	0.1	0 38
Melita sp.					1	M	0.1	0.38
Oxyurostylis smithi	3	N,O	0.2	0.43	3	O	0.2	0.79
Pinnixa sayana	3	M,O	0 2	0.43	2	M,O	0.1	0 36
Xanthidae (LPIL)					1		0 1	0.38
Mollusca								
Mulinia lateralis	17	N,M,O	5.5	5.69	19	N,M,O	7.1	5.74
Nassarius vibex					1	O	0 1	0 38
Nuculana acuta	2	O	0.1	0 49				
Rangia cuneata	1	N	0.1	0.38				
Tellina texana	1	N	0.1	0.38				
Rhyncocoela								
Cerebratulus lacteus	1	M	0.1	0.38	4	N,M	0.2	0.47
Lineus sp.					1	O	0.1	0.38
Micrura sp.	15	N,M,O	1.2	1 14	11	N,M,O	0.8	0 89
Nemertea sp. A (LPIL)	21	N,M,O	9 2	5.32	18	N,M,O	8 8	8.39
Echinodermata								
Microphiopholis atra					5	M,O	0.4	1.01

Table 6, continued

Species	Reference Fq	Loc	x	sd	Disposal Fq	Loc	x	sd
Phoronida								
Phoronis sp	1	O	0 1	0.38	1	O	0.1	0.38
Hemichordata								
Enteropneusta (LPIL)	13	M,O	2.1	2.52	12	M,O	4.1	5 90
Holothuroidea								
Holothuroidea (LPIL)	5	M,O	0.6	1.29	4	M	0.2	0.40
Platyhethminthes								
Stylochus ellipticus	1	N	0.1	0.38	2	N,M	0.2	0.83
FRESHWATER BAYOU								
Annelida								
Cossura soyeri	9	M,O	0.9	1.46	8	N,M	0.8	1.40
Lepidasthenia sp.	3	O	0.1	0.53	1	O	0 1	0 38
Mediomastus californiensis	15	N,M,O	6 2	8 69	7	N,M,O	1 2	1 95
Neanthes micromma	4	M,O	0.2	0.47	3	M,O	0.2	0.61
Parandalia sp.					1	N	0 1	0.38
Paraprionospio pinnata	20	N,M,O	13.9	24.64	21	N,M,O	10.5	9.81
Podarkiopsis levifusina	4	M,O	0 4	1.22	2	O	0 1	0.79
Polychaete sp. B (LPIL)	2	O	0.1	0 49				
Prionospio sp.	1	O	0.2	1.51	3	M,O	0.2	0.84
Pseudoeurythoe ambigua	10	N,M,O	1.9	3.02	10	M,O	3 4	5.49
Sigambra tentaculata	7	M,O	0.7	1.19	7	M,O	0.9	1.69
Streblospio benedicti	1	M	0.1	0.38				
Spiochaetopterus oculatus					1	M	0.1	0.38
Arthropoda								
Balanus sp.	7	N,M,O	0.5	0.87	9	N,M,O	1.9	3.98
Corophium sp	1	O	0 1	0 38				
Paguridae (LPIL)					1	M	0.1	0.38
Pinnixa sayana	3	O	0.4	1.46	3	O	0.1	0.53
Mollusca								
Mulinia lateralis					1	M	0.1	0.76
Nassarius vibex	1	O	0.1	0.38				
Rhyncocoela								
Cerebratulus lacteus	2	O	0 1	0.49				
Micrura sp.	9	N,M,O	0.9	1.81	11	N,M,O	1.3	1 90
Nemertea sp. A (LPIL)	9	N,M,O	0.6	0.87	12	N,M,O	2 7	7.93
Phoronida								
Phoronis sp.	1	O	0.1	0.38				
Hemichordata								
Enteropneusta (LPIL)	7	M,O	1.3	2.80	5	N	0 6	1.77
Holothuroidea								
Holothuroidea (LPIL)	1	O	0.1	0.86				

Table 6, continued

Species	Reference Fq	Loc	x	sd	Disposal Fq	Loc	x	sd
MERMENTAU RIVER								
Annelida								
Cossura soyeri	20	N,M,O	2.6	1.66	14	M,O	2.4	2.34
Glycinde solitaria	5	N,M,O	0 3	0.56	17	N,M,O	1.8	1.67
Magelona cf. *phyllisea*	2	N,O	0 1	0.30	8	M,O	0 8	1.67
Mediomastus californiensis	3	N,O	0.1	0.43	4	M,O	0.3	0.90
Neanthes micromma					1	O	0.1	0.38
Paraprionospio pinnata	21	N,M,O	51.2	20.35	21	N,M,O	113.5	20.78
Polychaete sp.A					1	M	0.1	0.38
Prionospio sp	1	M	0.1	0.38				
Pseudoeurythoe ambigua	15	N,M,O	1.0	0.95	18	N,M,O	5 8	5 14
Sigambra tentaculata					9	N,M,O	0.6	0.86
Spiochaetopterus oculatus					1	M	0.1	0.38
Arthropoda								
Balanus sp.	12	N,M,O	1.3	1 83	14	N,M,O	2.3	4.46
Corophium sp.	1	N	0.1	0.38				
Pinnixa sayana					1	O	0.1	0.38
Mollusca								
Mulinia lateralis	1	N	0.1	0.38	1	N	0 1	0 38
Rhyncocoela								
Cerebratulus lacteus	3	M,O	0.1	0 43	10	N,M,O	0.6	0.68
Micrura sp.	2	O	0.1	0 78	3	N,M	0 1	0.43
Nemertea Sp. A (LPIL)	12	N,M,O	0.8	0.77	17	N,M,O	1.9	1 53
Holothuroidea								
Holothuroidea (LPIL)	1	O	0.1	0.38	4	M,O	0.7	2.20

Table 7. Mean ($\bar{x}$) of abundance and taxa richness per 0.05m^2 standard deviations (sd) for each reference and disposal site within study areas

Mean Taxa Richness

Channel	Station	N	Reference $\bar{x}$	Reference sd	Disposal $\bar{x}$	Disposal sd
Atchafalaya						
	Nearshore	7	11.9	2.12	6.1	1.95
	Midsection	7	11 9	2.19	12.4	3 60
	Offshore	7	15 7	2.43	16.0	2.16
	Overall	21	13.1	2.83	11 5	4 88
Freshwater						
	Nearshore	7	3.3	1.38	3.3	0.76
	Midsection	7	5.0	2.58	6 0	2 45
	Offshore	7	8.3	3.55	6 4	1.99
	Overall	21	5.5	3.30	5 2	2.28
Mermentau						
	Nearshore	7	4.0	2.00	4.9	1.77
	Midsection	7	4.3	0.95	9 0	1.73
	Offshore	7	5.9	1.35	7.0	1.53
	Overall	21	4 7	1.65	7 0	2.36

Mean Abundance

Channel	Station	Reference N	Disposal $\bar{x}$	sd	$\bar{x}$	sd
Atchafalaya						
	Nearshore	7	144.3	29 84	51.6	30.75
	Midsection	7	92.7	40.84	176 1	94.35
	Offshore	7	91.3	19.91	85.0	24.54
	Overall	21	109.4	39 05	104.2	77.75
Freshwater						
	Nearshore	7	9.7	6.60	10.3	7.89
	Midsection	7	32 6	21.04	18.6	12.16
	Offshore	7	44.0	48.11	44.7	23.15
	Overall	21	28.8	32.46	24.5	21.21
Mermentau						
	Nearshore	7	61.4	14.70	119 3	26.47
	Midsection	7	68.1	18.88	137.6	22.83
	Offshore	7	44.0	19.44	136.3	22.46
	Overall	21	57.9	19.85	131 0	24.31

Table 8. Results of two-way ANOVA's for taxa richness for the Atchafalaya River, Freshwater Bayou, and Mermentau River study areas. Main effects were Site (Reference site, RS; Disposal Site, DS) and Station Location (Nearshore, N; Middepth, M; and Offshore, O). Tukey's post hoc analysis was used to compare a) main effect means when effects were significant and interaction was not and b) Site by Station Location means when a significant interaction occurred. Means followed by the same letter are not significantly different (alpha = 0.05). Standard deviation are in parentheses ( ).

ATCHAFALAYA

Source	d.f	F	P
R/DS	1	6 76	0.0134
SL	2	27 59	0 0001
R/DS x SL	2	9 20	0.0006

Results of Tukey's Test. Means (x) and standard deviation (sd) are given; N = 7.

		Site	
		RS	DS
Station	N	11.9[b] (2.12)	6 1[c] (1 95)
Location	M	11.9[b] (2.19)	12 4[ab] (3.60)
	O	15.7[ab] (2.43)	16 0[a] (2.16)

FRESHWATER

Source	d.f.	F	P
R x DS	1	0.02	0.8959
SL	2	11.80	0 0001
R/DS x SL	2	1 13	0.3345

Results of Tukey's Test. Means (x) and standard deviation (sd) are given; N = 14

Station	x	sd
N	3.3[a]	1.06
M	5 5[b]	2.47
O	7.4[b]	2 92

MERMENTAU

Source	d.f.	F	P
R/DS	1	18.04	0 0001
SL	2	8.11	0.0012
R/DS x SL	2	4 49	0 0182

Results of Tukey's Test; N = 7.

		Site	
		RS	DS
Station	N	4 0[c] (2.00)	4 9[bc] (1.77)
Location	M	4.3[c] (0.95)	9.0[a] (1.73)
	O	5.9[bc] (1.35)	7.0[ab] (1.53)

Table 9. Results of two-way ANOVA's for the Atchafalaya River, Freshwater Bayou, and Mermentau River study areas. Main effects were Site (Reference site, RS; and Disposal Site, DS) and Station Location (Nearshore, N; Middepth, M; and Offshore, O). Tukey's *post-hoc* analysis was used to compare a) main effect means when main effects were significant and interaction was not and b) Site by Station Location means when a significant interaction occurred. Means followed by the same letter are not significantly different (alpha = .05)

ATCHAFALAYA

Source	d.f.	F	P
R/DS	1	2.30	0 1381
SL	2	2 08	0.1391
R/DS x SL	2	12.08	0.0001

Results of Tukey's Test. Means (x) and standard deviation (sd) are given; (N = 7).

		Site	
		RS	DS
Station	N	144.3^{ab} (29 84)	51.5^{c} (30 75)
Location	M	92.7^{bc} (40.84)	176.1^{a} (94.35)
	O	$91\ 3^{bc}$ (19.91)	85.0^{bc} (24.54)

FRESHWATER

Source	d.f.	F	P
R x DS	1	0.37	0 5450
SL	2	12.13	0.0001
R/DS x SL	2	1.42	0.2554

Results of Tukey's Test (N = 14).

Station Location	$\bar{x}$	sd
N	10 0a	6.99
M	25.6^{b}	18.04
O	44.4^{b}	36.27

MERMENTAU

Source	d.f.	F	P
R/DS	1	78.60	0 0001
SL	2	2.35	0 1095
R/DS x SL	2	3.28	0.0491

Results of Tukey's Test (N = 7).

		Site	
		RS	DS
Station	N	61.4^{b} (14.70)	119.3^{a} (26.47)
Location	M	68.1^{b} (18.88)	137.6^{a} (22.83)
	O	44.0^{b} (19 44)	136.3^{a} (22.46)

Table 10. Average abundance per $0.05 m^2$, standard deviation ( ), and relative abundance [ ] of benthic macrofauna (density >1 at, at least, one station) for the Atchafalaya River study area. Nearshore (N), Middepth (M), and Offshore (O) stations and overall mean ($\bar{x}$) by site for individual taxa are provided. Means followed by similar letters are not significantly different (alpha = 0.05) based on ANOVA and Tukey's test.

Taxa	Reference Station				Disposal Station			
	N	M	O	$\bar{x}$	N	M	O	$\bar{x}$
Ampharete americana	0.3^{a} (0.49) [12]	0.0^{a}	1.9^{a} (1.21) [13]	0.7^{e} (1.10) [16]	0.1^{a} (0.38) [9]	0.0^{a}	1.9^{a} (3.24) [13]	0.7^{e} (1.98) [16]
Cossura soyeri	0.1^{b} (0.38) [13]	2.1^{a} (1.21) [8]	3.6^{a} (4.47) [9]	1.9^{e} (2.92) [10]	0.0^{b}	0.0^{b}	3.6^{a} (2.15) [9]	1.2^{e} (2.09) [13]
Diopatra cuprea	0.1^{b} (0.38) [13]	0.3^{b} (0.76) [14]	0.1^{b} (0.38) [16]	0.2^{e} (0.51) [19]	0.0^{b}	2.4^{a} (2.22) [9]	0.0^{b}	0.8^{e} (1.69) [14]
Glycinde solitaria	5.9^{a} (3.72) [6]	7.3^{a} (6.18) [6]	3.0^{a} (2.83) [11]	5.4^{e} (4.62) [7]	1.1^{a} (1.46) [5]	5.6^{a} (3.69) [7]	3.1^{a} (2.58) [12]	3.2^{e} (3.19) [8]
Mediomastus californiensis	81.1^{a} (20.28) [1]	$27.7^{c,d}$ (18.07) [1]	$13.6^{c,d}$ (5.86) [3]	40.8^{e} (33.48) [1]	$30.3^{a,b,c}$ (23.46) [1]	$84.3^{a,b}$ (65.45) [1]	7.4^{d} (2.88) [4]	40.7^{e} (50.43) [1]
Micura sp.	1.7^{a} (1.60) [10]	0.7^{a} (0.49) [12]	1.3^{a} (0.95) [14]	1.2^{e} (1.14) [15]	0.7^{a} (0.76) [6]	0.4^{a} (0.79) [13]	1.1^{a} (1.07) [14]	0.8^{e} (0.89) [14]
Mulinia lateralis	9.0^{a} (4.83) [3]	1.0^{b} (1.16) [10]	$6.6^{a,b}$ (6.63) [4]	5.5^{e} (5.69) [6]	8.6^{a} (5.97) [2]	8.7^{a} (6.63) [6]	$4.0^{a,b}$ (3.70) [7]	7.1^{e} (5.74) [5]
Paraprionospio pinnata	6.9^{b} (4.74) [5]	13.9^{a} (5.15) [2]	16.0^{a} (2.24) [2]	12.2^{e} (5.67) [2]	0.0^{c}	$12.7^{a,b}$ (6.70) [5]	12.4^{a} (4.72) [2]	8.4^{f} (7.55) [4]
Polydora sp.	0.1^{b} (0.38) [13]	0.1^{b} (0.38) [15]	5.4^{a} (7.74) [6]	1.9^{e} (4.96) [10]	0.0^{b}	0.1^{b} (0.38) [14]	0.1^{b} (0.38) [15]	0.1^{f} (0.30) [18]
Pseudoeurothoe ambigua	1.7^{a} (1.77) [10]	10.6^{a} (7.46) [4]	16.9^{a} (3.58) [1]	9.8^{e} (7.82) [3]	0.0^{a}	16.6^{a} (16.35) [4]	15.4^{a} (11.13) [1]	10.7^{e} (13.32) [2]
Sigambra tentaulata	0.0^{a}	0.3^{a} (0.48) [14]	3.7^{a} (1.80) [8]	1.3^{e} (2.00) [14]	0.0^{a}	0.1^{a} (0.38) [14]	4.6^{a} (2.15) [6]	1.6^{e} (2.48) [12]
Spiochaetopterus oculatus	$5.9^{a,b}$ (3.71) [6]	$13.4^{a,b}$ (11.41) [3]	$.3^{b,c}$ (1.38) [12]	7.9^{e} (8.15) [5]	0.0^{c}	17.1^{a} (11.78) [3]	$3.9^{a,b}$ (3.13) [8]	7.0^{e} (10.06) [6]
Streblospio benedicti	4.3^{a} (9.32) [8]	0.0^{b}	0.0^{b}	1.4^{e} (5.51) [13]	7.4^{b} (4.50) [3]	0.0^{b}	0.0^{b}	2.5^{f} (4.35) [9]
Unknown Enteropneusta	0.0^{a}	2.6^{a} (1.51) [7]	3.9^{a} (3.31) [7]	2.1^{e} (2.52) [9]	0.0^{a}	3.1^{a} (3.44) [8]	9.1^{a} (7.34) [3]	4.1^{e} (5.90) [7]
Unknown Holothuroidea	0.0^{a}	1.6^{a} (1.90) [9]	0.1^{a} (0.38) [16]	0.6^{e} (1.29) [17]	0.0^{a}	0.6^{a} (0.53) [12]	0.0^{a}	0.2^{e} (0.40) [17]
Unknown Nemertea	$12.4^{a,b}$ (5.44) [2]	$9.4^{a,b}$ (5.61) [5]	6.1^{b} (3.24) [5]	9.2^{e} (5.33) [4]	1.7^{c} (1.89) [4]	17.7^{a} (7.87) [2]	$7.0^{a,b}$ (3.79) [5]	8.8^{f} (8.39) [3]
Unknown Oligochaeta	8.0^{a} (4.58) [4]	0.6^{c} (1.13) [13]	$3.1^{a,b}$ (0.69) [10]	3.9^{e} (4.10) [8]	0.3^{c} (0.49) [7]	2.4^{b} (3.26) [9]	3.3^{b} (4.31) [11]	2.0^{f} (3.42) [10]

Table 11. Average abundance per $0.05m^2$, standard deviation (sd) and relative abundance [ ] of benthic macrofauna (density > 1.0 at, at least, one station) for the Freshwater Bayou study area. Nearshore (N), middepth (M) and offshore (O) stations and overall mean ($\bar{x}$) by site for individual taxa are provided. Means followed by similar letters are not significantly different (alpha = 0.05) based on ANOVA and Tukey's test.

Taxa	Reference Site Station				Disposal site Station			
	N	M	O	$\bar{X}$	N	M	O	$\bar{X}$
Balanus sp.	0.4^{a} (1.13) [5]	0.6^{a} (0.79) [6]	0.6^{a} (0.79) [6]	0.5^{d} (0.87) [8]	3.7^{a} (6.40) [1]	1.7^{a} (2.06) [4]	0.1^{a} (0.38) [8]	1.9^{d} (3.98) [3]
Unknown Nemertea	0.6^{a} (0.55) [4]	0.7^{a} (1.25) [5]	0.6^{a} (0.79) [6]	0.6^{d} (0.87) [8]	1.0^{a} (1.00) [3]	1.1^{a} (0.90) [5]	5.9^{a} (13.78) [3]	1.9^{d} (3.98) [3]
Cossura soyeri	0.0^{a} [7]	2.0^{b} (2.00) [3]	0.6^{c} (0.79) [6]	0.9^{d} (1.46) [6]	0.1^{a} (0.38) [6]	2.3^{b} (1.60) [3]	0.0^{c} [9]	0.8^{d} (1.40) [8]
Micrura sp.	0.4^{a} (0.53) [5]	0.1^{a} (0.38) [9]	2.1^{b} (2.79) [4]	0.9^{d} (1.81) [5]	0.3^{a} (0.49) [5]	0.4^{a} (0.79) [7]	3.1^{b} (2.27) [4]	1.3^{d} (1.90) [5]
Paraprionospio pinnata	3.0^{a} (3.21) [2]	12.9^{b} (8.44) [2]	26.0^{b} (40.41) [1]	14.0^{d} (24.65) [1]	3.6^{a} (2.30) [2]	8.1^{b} (6.59) [1]	19.7^{b} (10.52) [1]	10.5^{d} (9.81) [1]
Sigambra tentaculata	0.0^{a} [7]	0.3^{b} (0.49) [7]	1.9^{b} (1.46) [5]	0.7^{d} (1.19) [7]	0.0^{a} [6]	0.3^{b} (0.76) [8]	2.6^{b} (0.76) [5]	1.0^{d} (1.69) [7]
Unknown Enteropneusta	0.0^{a} [7]	0.3^{b} (0.76) [7]	3.6^{b} (4.04) [3]	1.3^{d} (2.80) [4]	0.1^{a} (0.38) [6]	0.1^{b} (0.38) [9]	1.6^{b} (2.94) [6]	0.6^{d} (1.77) [9]
Mediomastus californiensis	4.3^{ab} (3.25) [1]	14.3^{a} (10.86) [1]	0.1^{c} (0.38) [9]	6.2^{d} (8.69) [2]	1.0^{bc} (1.91) [3]	2.4^{bc} (2.37) [2]	0.3^{c} (0.76) [7]	1.2^{e} (1.95) [6]
Pseudoeurythoe ambigua	1.0^{b} (2.24) [3]	0.9^{b} (1.07) [4]	3.9^{ab} (4.18) [2]	1.9^{d} (3.02) [3]	0.0^{b} [7]	0.9^{b} (1.22) [6]	9.4^{a} (5.97) [2]	3.4^{e} (5.49) [2]

Table 12 Average abundance per 0.05m^2 standard deviation (sd) and relative abundance [ ] of benthic macrofauna (density > 1.0, at least, one station) for the Mermentau River study area. Nearshore (N), middepth (M) and offshore (O) stations and overall mean ($\bar{x}$) by site for individual taxa are provided. Means followed by similar letters are not significantly different (alpha = 0.05) based on ANOVA and Tukey's test

Taxa	Reference Site				Disposal Site			
	Station				Station			
	N	M	O	$\bar{x}$	N	M	O	$\bar{x}$
Balanus sp.	1.6^{a} (2 57) [2]	1.6^{a} (1.90) [3]	0.9^{a} (0.69) [4]	1.3^{c} (1.83)	0.6^{a} (0.79) [5]	2 6^{a} (2 76) [5]	3.7^{a} (7 23) [3]	2.3^{c} (4.46)
Glycinde solitaria	0 1^{a} (0 38) [6]	0.3^{a} (0.76) [6]	0.4^{a} (0.53) [5]	0.3^{c} (0.56)	1.3^{a} (1.38) [4]	2 7^{a} (2.29) [4]	1 3^{a} (0 76) [6]	1.8^{f} (1.67)
Unknown Nemertea	0.6^{a} (0.79) [5]	0.6^{a} (0.79) [5]	1.1^{a} (0.69) [3]	0.8^{c} (0.77)	2.3^{a} (2 06) [3]	1.6^{a} (1.40) [8]	1.7^{a} (1.11) [5]	1.9^{f} (1.53)
Pseudoerythoe ambigua[1]	0.4^{b} (0 53) [4]	1.4^{a} (1.27) [4]	1 1^{b} (0.69) [3]	1.0^{e} (0.95)	4.6^{b} (5.09) [2]	8.9^{a} (5.98) [2]	4.0^{b} (3.16) [2]	5.8^{f} (5 14)
Magelona sp.	0.1^{b} (0.38) [6]	0.0^{b} [7]	0.1^{b} (0.38) [6]	0.1^{c} (0.30)	0.0^{b} [7]	2.3^{a} (1.80) [6]	0.1^{b} (0.38) [7]	0.8^{f} (1.47)
Cossura soyeri[2]	1.6^{c} (1.13) [2]	3.7^{ab} (2.06) [2]	2.4^{bc} (0.98) [2]	2.6^{e} (1.66)	0.0^{d} [7]	5 4^{a} (1.72) [3]	1.9^{bc} (1.22) [4]	2.4^{f} (2 58)
Paraprionospio pinnata	56.4^{a} (16.09) [1]	60.3^{a} (20 34) [1]	36.9^{a} (18.43) [1]	51.2^{c} (20.35)	109.3^{b} (24.32) [1]	109 4^{b} (17 58) [1]	121.7^{b} (20 52) [1]	113.5^{f} (20.78)

1 Significant zonal effects, therefore, tested with Turkey's test.

2 Significant interaction between treatments (R & DS) and depth zones, therefore tested with Tukey's test.

Table 13 Accuracy of the sample mean ± %o, for estimating theoretical population mean based on seven replicates, and coefficient of variation (%) for each Ocean Dredged Disposal Site and Reference area station

Equation used to derive estimates (Eckblad, 1991):

$$\text{Accuracy} = \frac{(\text{coefficient of variation})(\text{t-value})}{\sqrt{\text{sample size}}}$$

Note: t-value taken from t-distribution table with 0.05 level of significance and six degrees of freedom.

	Atchafalaya River				Mermentau River				Freshwater Bayou			
	Reference		Disposal		Reference		Disposal		Reference		Disposal	
	Accuracy ± %	CV (%)	Accuracy ± %	CV (%)	Accuracy ± %	CV (%)	Accuracy ± %	CV (%)	Accuracy ± %	CV (%)	Accuracy ± %	CV (%)
Nearshore	15 2	(20.7)	43.8	(59 6)	17 6	(23 9)	16 3	(22.2)	49.9	(67 9)	56.3	(76 7)
Mid-depth	32.3	(44 0)	39.3	(53.6)	20 3	(27 7)	12.2	(16.6)	47.4	(64.6)	48.1	(65 5)
Offshore	16 0	(21 8)	21 2	(28 9)	32.5	(44 2)	12.1	(16.5)	80.3	(109 3)	38 0	(51.8)

APPENDIX I

CHEMICAL ANALYSES OF SEDIMENT FROM THREE OCEAN DREDGED MATERIAL DISPOSAL SITES AND REFERENCE SITES IN MERMENTAU RIVER, FRESHWATER BAYOU AND ATCHAFALAYA RIVER, LOUISIANA

by

Analytical Chemistry Team
Ecotoxicology Branch
James C. Moore, Team Leader
E.M. Lores and Jerrold Forester

U.S. Environmental Protection Agency
Environmental Research Laboratory
Sabine Island
Gulf Breeze, FL 32561

Submitted to

Susan McKinney
U.S. Environmental Protection Agency
Region VI
6W-QM 1445 Ross Avenue
Dallas, Texas 75202

Final Report: November 1994

ABSTRACT

Sediments from Mermentau River, Freshwater Bayou and Atchafalaya River, Ocean Dredged Material Disposal Sites, and sediments from reference sites near these disposal sites were chemically analyzed for selected heavy metals, pesticides, PCBs, and petroleum hydrocarbons. Residues of PCBs or petroleum hydrocarbons were not detected in any sediment sample. Some chlorinated-hydrocarbon pesticides were detected at concentrations near the detection limit in Freshwater Bayou Ocean Dredged Material Disposal Site (ODMDS) and its nearby reference site, but none were found in Mermentau River ODMDS, in its nearby reference site, nor in Atchafalaya River ODMDS or its reference site. All sediment samples contained residues of some heavy metals. Concentrations of heavy metals in reference sediment samples were similar to concentrations in sediments from the disposal sites.

INTRODUCTION

Chemical analyses were performed on sediment from three dredged materials disposal sites and reference sites located near the disposal sites. Each disposal area and reference site was divided into three segments representing shallow, mid-depth and deepest depth of the disposal area with corresponding reference stations outside the disposal area with similar depths. Sediments from one sampling station of each area were sampled and chemically analyzed for selected chlorinated pesticides, PCBs, selected heavy metals and petroleum hydrocarbons. Chemical analyses for pesticides and petroleum hydrocarbons were performed by using a capillary-column gas chromatographs equipped with an electron-capture detector and flame-ionization detector respectively. Inductively coupled argon plasma emission spectroscopy was used to analyze sediments for heavy metals.

METHODS OF CHEMICAL ANALYSES

A. Chlorinated Hydrocarbon Pesticides

Sediments were air dried and blended as necessary, then weighed into 150-mm by 25-mm screw top test tube with teflon liner. Twenty milliliters of 20% (v/v) acetone in petroleum ether were added and then the samples were tumbled on a rotorack at 60-90 rpm for 30 minutes. Samples were then centrifuged (1600 x g) for 10 minutes. The solvent extracts were transferred to an oil sample bottle containing 50 mℓ of 2.0% (v/v) aqueous sodium sulfate and gently shook for one minute. After the two phases separated the solvent was transferred to a 25 mℓ-concentrator tube and the aqueous wash was repeated two more times. Sample extracts from the three aqueous washes were combined and concentrated to 1 mℓ on a nitrogen evaporator in preparation for Florisil cleanup. Samples analyzed with electron-capture detectors were shaken with 500 μℓ of mercury to remove sulfur before gas chromatographic analysis. Cleanup columns were prepared by adding 3 g of PR-grade Florisil (stored at 130°C) and 2 g of anhydrous sodium sulfate (powder) to a 200-mm by 9-mm i.d. Chromaflex column (Kontes Glass Co., Vineland, NJ) and rinsing with 10 ml of hexane. Sediment extracts were transferred to the column with two additional 2-ml volumes of hexane. Pesticides and PCBs were eluted with 20 ml of 5% (v/v) diethyl ether in hexane. Dieldrin and Endosulfan were eluted with 20 ml of 10% (v/v) isopropanol in isooctane.

Pesticides were analyzed with external standard methods. All standards were obtained from the EPA pesticide repository. Analyses were performed on a Hewlett-Packard Model 5890 gas chromatograph equipped with a ^{63}Ni electron-capture detector. Separations were performed by using a 30-m (0.32 mm i.d.) RTX-5 and RTX-1 fused silica capillary column. Other gas chromatographic parameters were: helium carrier gas flowing 1.5 mℓ/min; column temperature, 50°C (hold for 2 min),10°/min to 150°C, 2°/min to 260°C (hold for 3 min); inlet temperature 250°C; and detector temperature, 350°C; 10% methane in argon makeup gas flowing at 60 mℓ/min.

B. Heavy Metals

One to two grams of sediment were weighed and placed into a 40-ml reaction vessel. Five milliliters of concentrated nitric acid were added and the samples digested for 2 to 4 h at 90°C in a tube heater. Digestion was continued, with vessels capped, for 48 h at 70°C. After digestion, samples were transferred to 15-ml tubes and a 5-ml aliquot was diluted to 10 ml for aspiration into a Jarrell-Ash AtomComp 800 Series inductively-coupled argon-plasma emission spectrometer (ICP). This instrument acquires data for 15 elements simultaneously. No detectable residues could be found in method blanks. A solution of 10% nitric acid/distilled water was aspirated between samples to prevent carryover of residues from one sample to the next. Standards in 10% nitric acid were used to calibrate the instrument initially and adjustments were made when necessary. Concentrations were reported in two significant figures as our method allowed, and were not corrected for percentage recovery.

C. Petroleum Hydrocarbons

Ten grams of sediment were weighed into culture tubes and extracted as described by J.S. Warner (1976). Sample extracts were concentrated to approximately 0.50 ml for gas chromatographic analyses. Analyses were performed on a Perkin-Elmer gas chromatograph (GC) equipped with flame ionization detectors (FID). Separations were performed by using a 30-m, 0.32-mm i.d. RTX-5 fused silica capillary column. Helium carrier gas was used at a flow of 1.5 ml/min. Other gas chromatographic parameters were: oven temperature programmed from 50°C (hold for 2 min) at a rate of 20°/min to 315°C (hold for 5 min); injector temperature was 250°C and detector temperature was 350°C.

Quality Assurance of Chemical Analyses

All standards used for quantitations of pesticides were obtained from EPA's repository in Las Vegas, Nevada. Standard solutions of metals were obtained from J.T. Baker Chemical Co., Phillipsburg, NJ, and were Instra-Analyzed quality. Androstane was obtained from Sigma Chemical Company, St. Louis, MO, and was used as an internal standard to quantify petroleum hydrocarbons. Reagent and glassware blanks were analyzed to verify that the analytical system was not contaminated with chemical residues that could interfere with quantitations.

RESULTS AND DISCUSSION

Detection limits for chlorinated hydrocarbon pesticides and PCBs are shown in Table 1 along with recovery efficiencies for these compounds. Some compounds were volatile and could not be recovered efficiently as shown by low recovery. Tables 2 and 3 show results of chlorinated hydrocarbon pesticide and PCBs analysis. No residues of these compounds were detected in sediments from Mermentau River ODMDS and nearby reference site or in Atchafalaya Bayou ODMDS and its reference site. One sediment sample from Freshwater Bayou ODMDS and one sample from its reference site had detectable concentrations of pesticides (Table 3). Since these concentrations were near the method detection limit and are subject to unknown interferences at these low concentrations, these compounds may be falsely identified by the electron-capture detector.

Concentrations of selected heavy metals are shown in Tables 4, 5, and 6. Concentrations of metals in sediment from the disposal sites were similar to concentrations of metals in their reference sites. Concentrations of metals in sediment samples from different depths were similar for most metals except for chromium and copper. Concentrations of these metals were lower in nearshore samples than in samples from offshore at both Mermentau River disposal site and its reference site, and in Atchafalaya River ODMDS and its reference site.

Petroleum hydrocarbons were not detected in any sediment sample above 1.0 $\mu g/g$ wet weight, the method detection limit.

REFERENCE

Warner, J.S. 1976. Determination of Aliphatic and aromatic Hydrocarbons in Marine Organisms. Analytical Chemistry, 48, No. 3, 578-583.

Table 1. Average percentage recovery of selected chlorinated pesticides and PCB spiked on samples of sediment. Method detection limits are also shown.

Compound	Spike Concentration (μg/g)	N	Average Percentage Recovery and Standard Deviation	Method Detection Limit (μg/g wet weight)
Aldrin	0.10	3	76 ± 2.0	0.010
BHC Isomers				
Alpha	a			0.010
Beta	a			0.010
Gamma (lindane)	0.10	4	25 ± 3.7	0.010
Chlordane (alpha)	0.10	12	55 ± 18	0.010
Chlorpyrifos (Dursban)	a			0.010
DDE (P,P')	0.10	11	91 ± 14	0.010
DDD (P,P')	0.10	6	71 ± 5.3	0.010
Dieldrin	0.10	11	87 ± 15	0.010
Endrin	a		a	0.010
Endosulfan I	0.10	11	82 ± 21	0.010
Endosulfan II	0.10	12	96 ± 7.0	0.010
Endosulfan Sulfate	a			0.010
Heptachlor	0.10	3	76 ± 2.0	0.010
Heptachlor epoxide	0.10	3	67 ± 1.0	0.010
Hexachlorobenzene	0.10	3	68 ± 2.6	0.010
Methoxychlor	a			0.010
Mirex	0.10	12	87 ± 6.4	0.010
PCBs[b]	0.10	11	88 ± 4.7	0.010
Toxaphene	a			0.010

ND = Not detected

[a] Analytes were not spiked for recovery.
[b] Percentage recovery was based on specific congener analysis.

Table 2. Concentration (in $\mu g/g$ wet weight) of selected chlorinated pesticides and PCBs in replicate samples of sediment from Atchafalaya River ODMDS and a nearby reference site.

	Atchafalaya River ODMDS			Atchafalaya Reference Site		
Replicate	A	B	C	A	B	C
Aldrin	ND	ND	ND	ND	ND	ND
BHC Isomers						
Alpha						
Beta						
Gamma (lindane)	ND	ND	ND	ND	ND	ND
Chlordane (alpha)	ND	ND	ND	ND	ND	ND
Chlorpyrifos (Dursban)	ND	ND	ND	ND	ND	ND
DDE (P,P')	ND	ND	ND	ND	ND	ND
DDD (P,P')	ND	ND	ND	ND	ND	ND
Dieldrin	ND	ND	ND	ND	ND	ND
Endrin	ND	ND	ND	ND	ND	ND
Endosulfan I	ND	ND	ND	ND	ND	ND
Endosulfan II	ND	ND	ND	ND	ND	ND
Endosulfan Sulfate	ND	ND	ND	ND	ND	ND
Heptachlor	ND	ND	ND	ND	ND	ND
Heptachlor epoxide	ND	ND	ND	ND	ND	ND
Hexachlorobenzene	ND	ND	ND	ND	ND	ND
Methoxychlor	ND	ND	ND	ND	ND	ND
Mirex	ND	ND	ND	ND	ND	ND
PCBs	ND	ND	ND	ND	ND	ND
Toxaphene	ND	ND	ND	ND	ND	ND

ND = Not detected, see Table 1 for detection limits.

Table 3. Concentration (in μg/g wet weight) of selected chlorinated pesticides and PCBs in samples of sediment from Freshwater Bayou ODMDS, and a nearby reference site.

	Freshwater Bayou ODMDS			Freshwater Bayou Reference Site		
Replicate	A	B	C	A	B	C
Aldrin	ND	ND	ND	ND	ND	ND
BHC Isomers						
Alpha						
Beta						
Gamma (lindane)	ND	ND	ND	ND	ND	ND
Chlordane (alpha)	ND	ND	ND	ND	ND	ND
Chlorpyrifos (Dursban)	ND	ND	ND	ND	ND	ND
DDE (P,P')	0.026	ND	ND	ND	ND	0.036
DDD (P,P')	ND	ND	ND	ND	ND	ND
Dieldrin	ND	ND	ND	ND	ND	ND
Endrin	ND	ND	ND	ND	ND	ND
Endosulfan I	ND	ND	ND	ND	ND	ND
Endosulfan II	ND	ND	ND	ND	ND	ND
Endosulfan Sulfate	ND	ND	ND	ND	ND	ND
Heptachlor	ND	ND	ND	ND	ND	ND
Heptachlor epoxide	ND	ND	ND	ND	ND	ND
Hexachlorobenzene	ND	ND	ND	ND	ND	0.018
Methoxychlor	ND	ND	ND	ND	ND	ND
Mirex	ND	ND	ND	ND	ND	0.047
PCBs[b]	ND	ND	ND	ND	ND	ND
Toxaphene	ND	ND	ND	ND	ND	ND

ND = Not detected, see Table 1 for detection limits.

Table 4. Concentration of selected metals in three replicate sediment samples from Mermentau River ODMDS, Louisiana, and a nearby Reference Site.

Sediment Location	Depth	As[A]	Cd	Cr	Cu	Hg	Ni	Pb[A]	Se	Zn
		Concentrations in µg/g wet weight								
Mermentau River ODMDS	Nearshore	21	ND	5.8	5.5	ND	14	14	ND	29
	Mid-depth	34	ND	12	11	ND	22	20	ND	35
	Offshore	33	ND	11	12	ND	22	20	ND	36
Reference Site	Nearshore	39	ND	13	11	ND	24	19	ND	34
	Mid-depth	41	ND	15	13	ND	25	20	ND	30
	Offshore	59	ND	20	15	ND	31	28	ND	35
	Method Detection Limit in µg/g wet weight									
		2.5	0.15	0.50	0.25	0.50	0.60	2.5	2.5	0.15

ND = Not detected.

[A] Usual background correction techniques could not be applied because of the intense interference; therefore, without subtracting background, lead and arsenic may be present but not in quantities greater than these shown.

Table 5. Concentration of selected metals in three replicate sediment samples from Freshwater Bayou ODMDS, Louisiana, and nearby Reference Site.

		Concentrations in $\mu g/g$ wet weight								
Sediment Location	Depth	As[A]	Cd	Cr	Cu	Hg	Ni	Pb[A]	Se	Zn
Freshwater	Nearshore	29	ND	11	12	ND	21	17	ND	32
Bayou ODMDS	Mid-depth	31	ND	11	12	ND	24	24	ND	42
	Offshore	52	ND	18	17	ND	32	28	ND	37
Reference	Nearshore	49	ND	17	14	ND	27	24	ND	32
Site	Mid-depth	50	ND	18	17	ND	30	27	ND	39
	Offshore	43	ND	14	13	ND	28	28	ND	37

ND = Not detected; see Table 4 for detection limits.

[A] Usual background correction techniques could not be applied because of the intense interference; therefore, without subtracting background, lead and arsenic may be present but not in quantities greater than these shown.

Table 6. Concentration of selected metals in three replicate sediment samples from Atchafalaya River ODMDS, Louisiana, and a nearby Reference Site.

		Concentrations in μg/g wet weight								
Sediment Location	Depth	As[A]	Cd	Cr	Cu	Hg	Ni	Pb[A]	Se	Zn
Atchatalaya River ODMDS	Nearshore	11	ND	3.8	4.0	ND	10	7.5	ND	16
	Mid-depth	34	ND	11	14	ND	22	18	ND	31
	Offshore	34	ND	11	12	ND	21	20	ND	31
Reference Site	Nearshore	11	ND	4.5	4.4	ND	11	6.8	ND	13
	Mid-depth	42	ND	14	18	ND	29	23	ND	43
	Offshore	25	ND	10	10	ND	20	16	ND	28

ND = Not detected; see Table 4 for detection limits.

[A] Usual background correction techniques could not be applied because of the intense interference; therefore, without subtracting background, lead and arsenic may be present but not in quanities greater than these shown.