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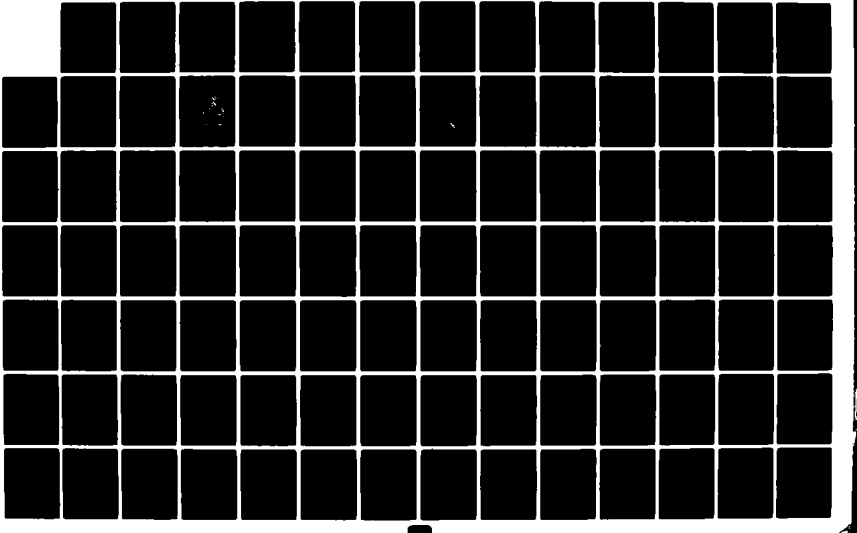

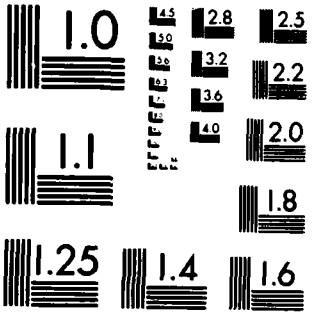

Environmental Effects of Hydraulic Dredging for Clam Shells in Lake Pontchartrain, Louisiana

by

Walter B. Sikora
Jean Pantell Sikora
Anny McK. Prior

June 1981

Prepared for

U.S. Army Engineer District, New Orleans
Contract No. DACW29-79-C-0099

Coastal Ecology Laboratory, Center for Wetland Resources,
Louisiana State University, Baton Rouge, Louisiana 70803
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PREFACE

This report presents the results of an investigation to determine the recovery rates of benthic communities in Lake Pontchartrain, Louisiana, following hydraulic dredging for clam shells. This study was sponsored by the New Orleans District of the U.S. Army Corps of Engineers (COE) under Contract No. DACW29-79-C-0099 to the Coastal Ecology Laboratory, Center for Wetland Resources, Louisiana State University, Baton Rouge, Louisiana.

The report was written and prepared by Walter B. Sikora and Jean Pantell Sikora, except for the chemistry section, which was written by Anny McK. Prior, and the appended reports by David W. Roberts and Leonard M. Bahr, Jr., and by John W. Fleeger. This report has been designated by the Coastal Ecology Laboratory as Contribution No. LSU-CEL-81-18.

Contracting Officer Representative of the COE was Frank J. Cali. John C. Weber, Sue R. Hawes, and Larry M. Hartzog assisted in review of the report.

The authors would like to acknowledge significant contributions by J. Wilkins, laboratory manager, in the field and in the laboratory; K. Westphal, figures; J. Bagur, editorial suggestions; C. Lusk and B. Grayson, typing; E. Parton, assistance with data management; K. Westphal, N. Walker, M. Lindsay, R. Robertson, and C. Cardiff, assistance in field and laboratory; R. Wilson, boat captain and Dr. Richard W. Heard, taxonomic assistance. We would also like to acknowledge the cooperation of the Lake Pontchartrain Shell Producers Association in several phases of the study and of the Greater New Orleans Expressway Commission for permitting the experimental dredging at the study site within the restricted zone of the Lake Pontchartrain Causeway.

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INTRODUCTION

Hydraulic dredging is a common industrial activity in many coastal waterbodies, including estuaries, and is often carried out on a large scale over long time periods. However, because different techniques are employed to achieve different results, the effects of dredging operations vary greatly. There is a need, therefore, to differentiate between the various types of dredging activities, which may be broadly classified into two categories: channel dredging and mining. Channel and canal dredging can alter water circulation patterns, particularly by allowing intrusion of higher salinity water into normally low salinity environments. Spoil piles from channel and canal dredging can redirect water flow, altering normal water circulation and impeding normal mixing processes. These alterations can have rapid and long lasting effects, often completely changing the ecology of large areas. Maintenance dredging, which often follows channel dredging and is undertaken to keep navigation channels at prescribed depths, frequently involves the removal of medium to coarse grained sediments in areas of high water flow. Results from studies of the effects of dredging in these systems are usually only applicable to these same environments.

Mining by hydraulic dredging, although utilizing similar equipment, is usually carried out in open, semiprotected waterbodies with low water flow, for the purpose of extracting buried shell deposits. Shell dredging on the Gulf Coast is of two different types: oyster shell dredging and clam shell dredging. The two types of dredging could have quite different environmental effects when viewed from the system level. Both types of dredging by necessity involve the drastic disturbance of the bottom sediments and the indigenous benthic communities as they pass through the processing plant of the dredge. However, because oysters are reef-forming organisms, the shell resource is concentrated in clumps or bands from less than a hundred meters to a kilometer wide and sometimes several kilometers long, often buried 5 to 15 meters or more below the sediment surface. Often a deep hole 2 to 4 meters in depth remains after the oyster shell is extracted and the spoil sediment in the hole consolidates (Harper and Hopkins 1976). These holes can remain for many years, with resulting salinity and oxygen stratification. The overall result on a system-wide basis is a potholed effect, with severe disturbance in concentrated areas, yet with large, undisturbed areas. May (1973) and Hopkins and McKinney (1976) present extensive literature reviews of oyster shell dredging.

Clam shell dredging, on the other hand, presents a different set of problems. The shells of the brackish water clam Rangia cuneata are deposited in shallow layers, 0.5 to 1 meter deep, blanketing the entire bottom of an estuary. The strategy for harvesting this resource is for the dredges to move continuously at 5 to 8 kilometers per hour, constantly dredging a shallow trench 1 to 2 meters wide. Nearly 0.25 km^2 per day will pass through the processing plant of a single dredge, with a larger area being affected by spoil. Several clam shell dredges working simultaneously could affect an entire estuary in a matter of years.

General Description of Lake Pontchartrain

Lake Pontchartrain is a shallow (mean depth 3 m, maximum 5 m) body of water of about 1630 km^2 lying in the middle of a large, southeastern Louisiana estuarine complex (Figure 1). To the west is Lake Maurepas, connected to Pontchartrain by Pass Manchac; to the east, Pontchartrain is connected to the Mississippi Sound by The Rigolets Pass and to Lake Borgne by the Chef Menteur Pass. In the southeast, the man-made Inner Harbor Navigation Canal-Mississippi River Gulf Outlet complex connects the lake to the Gulf of Mexico. The majority of freshwater input into the lake is from rivers emptying into the northwestern part of the lake; the Tickfaw River and the Amite-Comite River complex via Pass Manchac, and the Tangipahoa and Tchefuncte Rivers emptying directly into the lake. During flood years, Pontchartrain can receive Mississippi River water via the Bonnet Carré Spillway and some Pearl River water via The Rigolets. During times of normal flow, higher salinity water enters the eastern portion of the lake via the passes.

Circulation within the lake has been shown to be primarily wind driven (GSRI 1972; Gael 1980), with current speeds of about 15-20 cm/sec (U.S. Army Corps of Engineers 1962). In addition, there is a diurnal tide with a mean amplitude of 11 cm (Outlaw 1979). The salinity regime of the lake is quite low, with a mean of about 5 ppt, and it is horizontally stratified, with salinities highest in the eastern half and lowest in the western half. At times this gradient may reach 12 ppt. The average isohalines for 1978 are shown in Figure 2. Also shown in Figure 2 are the experimental and control dredging stations and the 13 stations of the concurrent benthic characterization study. Variations in salinity and temperature in the vertical are small enough to be inconsequential most of the year, which indicates that the lake is vertically a well-mixed system. Lake temperatures show a generally isothermal pattern, the maximum spatial gradient being 4°C (Swenson 1980). Water temperature generally follows air temperature, with a maximum water temperature of about 30°C in August-September and a minimum of about 6°C in January-February. Clay and silty clay sediments predominate in the central region of the lake (Barrett 1976; Bahr, Sikora, and Sikora 1980) with silty clay found at the dredging stations (Figure 3).

Because of the extensive area of the lake and hence its large fetch, wind-induced waves play an important role in the system. Swenson (1980) has shown that silty clay sediments can be resuspended by wind-induced waves of 0.75 m to 1.3 m in height, which occur with wind speeds of about 15 mph (6 m/sec). He has further shown (from Gael 1980) that winds of this speed or greater occur at least 15% of the time. Thus, one can conclude that the bottom of the lake is in motion at least 15% of the time because of natural causes.

Hydraulic Dredging for Clam Shells in Lake Pontchartrain

Hydraulic dredging for Rangia shell in Lake Pontchartrain began around 1933 and has steadily intensified in the subsequent 48 years to the present time. This trend can readily be seen from clam shell production estimates by the Louisiana Wild Life and Fisheries Commission,

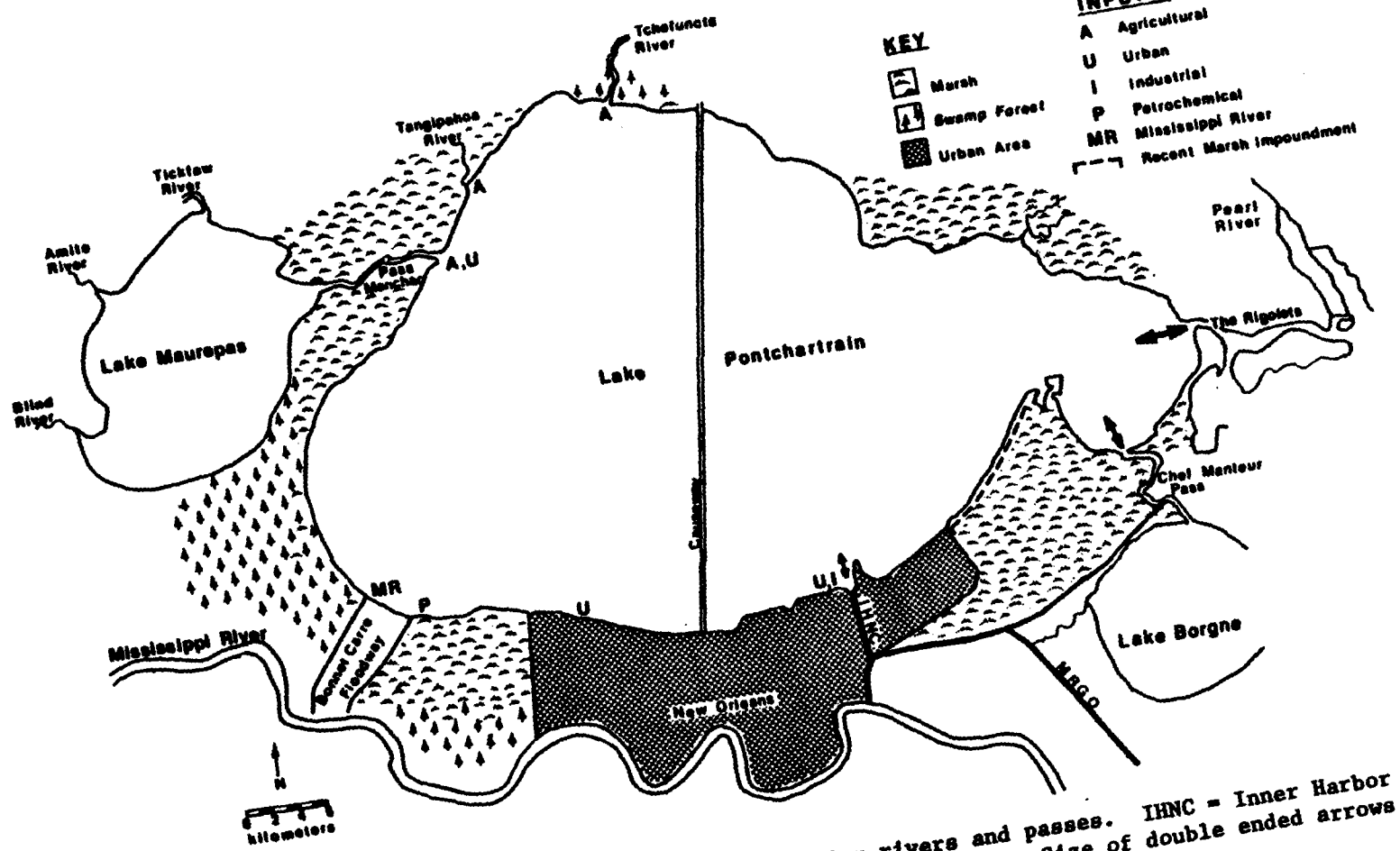


Figure 1. Lake Pontchartrain, Louisiana, showing major rivers and passes. IHNC = Inner Harbor Navigation Canal, MRGO = Mississippi River Gulf Outlet. Size of double ended arrows at passes indicate relative flows. Inputs indicated by letters,

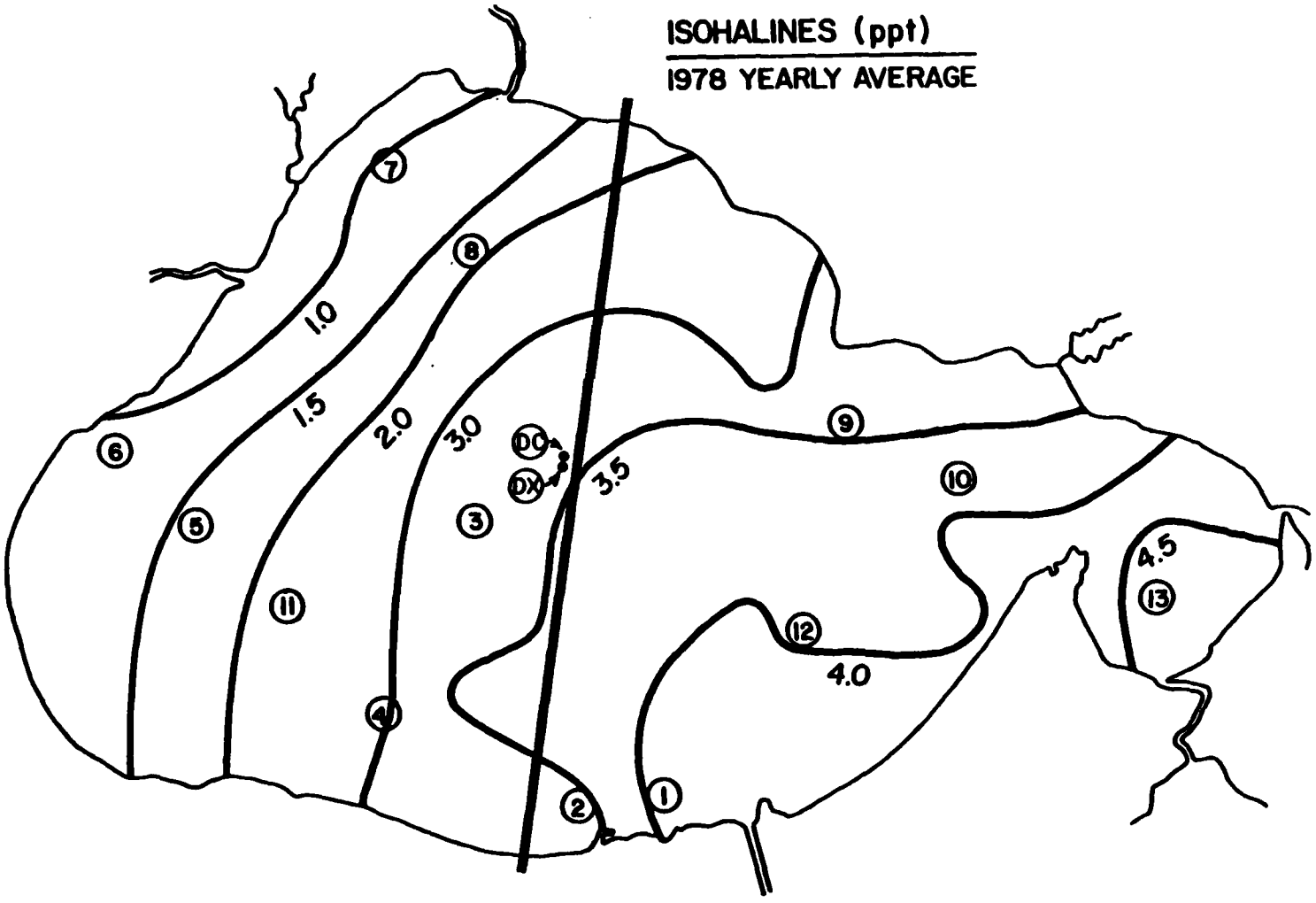


Figure 2. Lake Pontchartrain, Louisiana showing the dredging control station (DC), the dredging experimental station (DX), the thirteen stations of the benthic characterization study and the average isohalines (ppt) for 1978.

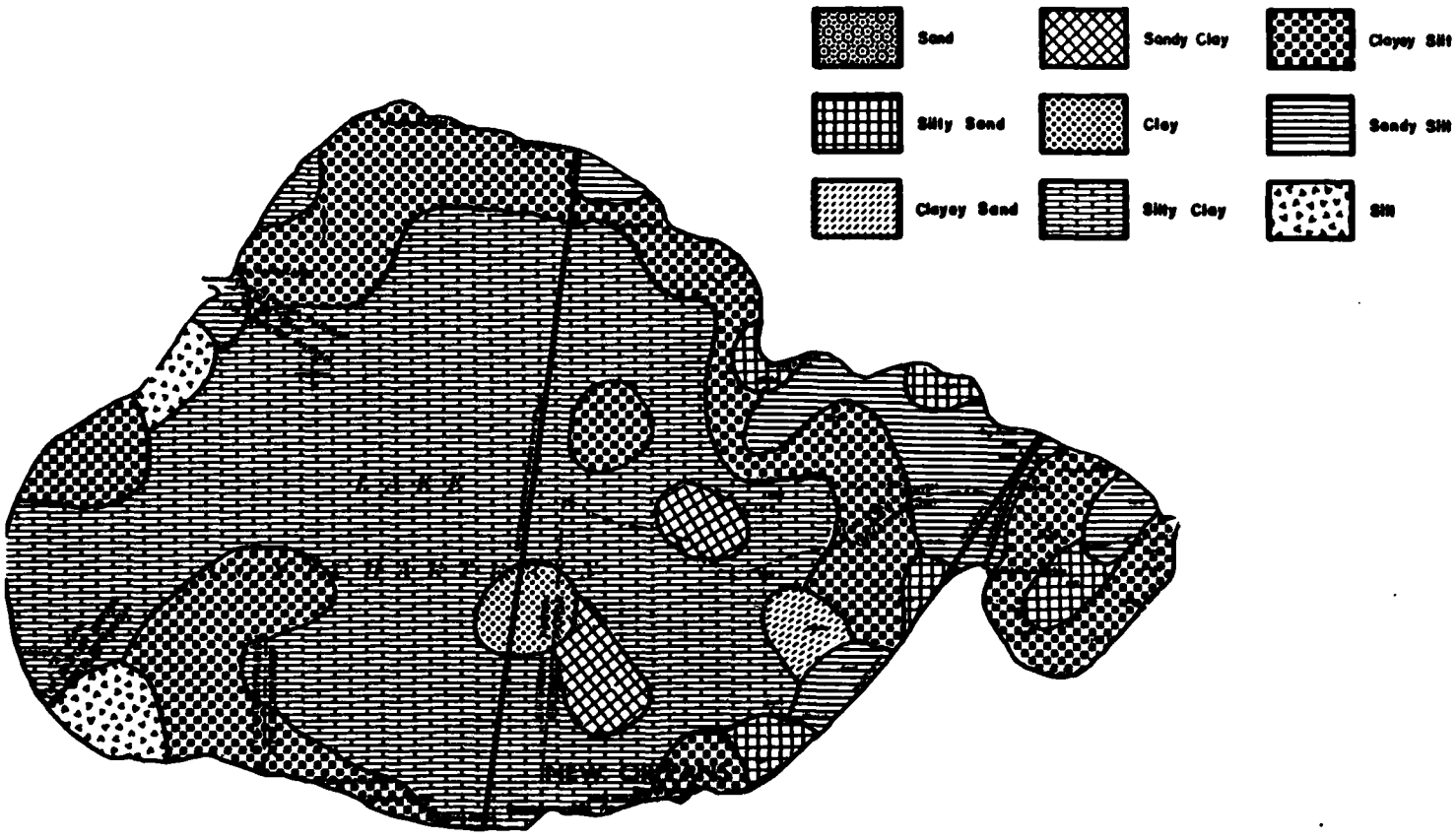


Figure 3. (From Barrett 1976). Sediment grain size distribution for 1973.

(1968), which range from 300,000 cubic yards statewide in the mid-thirties to 5,000,000 cubic yards, mainly from Lakes Pontchartrain and Maurepas in 1968. In recent times there have been two attempts at assessing the environmental effects of those operations in Lake Pontchartrain (Tarver 1973, Gulf South Research Institute 1974). Neither of these efforts, however, were concerned with possible long-term effects, and both concluded that short-term effects, particularly on the water column and on nektonic species, were negligible and transitory in nature. Any realistic evaluation of the effects of shell dredging would by necessity have to include the long-term effects on the sediment structure and chemistry as well as on the benthic infauna, both macrofauna and meiofauna, which live in the sediments. Only then could projections be made to include the entire lake ecosystem through time.

Magnitude of Shell Dredging Operations

Before considering possible long-term effects of shell dredging on the fauna or lake ecosystem as a whole, the magnitude of the perturbation resulting from shell dredging should be put in perspective. The trench width cut by the "fish mouth" of dredges operating in Lake Pontchartrain is estimated at 4 to 6 feet and the forward speed of a dredge is 3 to 5 miles per hour (GSRI 1974). For calculation purposes, we will use an average width of 5 feet (1.524 m) and an average speed of 4 miles per hour (6437 m/hr). Thus, an average dredge would cover 9810 m^2 in an hour, or $2.35 \times 10^5 m^2$ in a day. If we assume that of the seven dredges in Lake Pontchartrain (GSRI 1974), on the average, one is laid up for major repairs at any one time, and the six working dredges operate for at least 360 days a year, we have;

$$2.35 \times 10^5 \times 360 \text{ days} \times 6 \text{ dredges} = 5.08 \times 10^8 m^2$$

perturbed by dredges annually. However, according to Mr. Don Palmore of the Lake Pontchartrain Shell Producers Association, only five dredges operate an average of 270 days a year. Using this estimate we would have;

$$2.35 \times 10^5 \times 270 \text{ days} \times 5 \text{ dredges} = 3.17 \times 10^8 m^2$$

perturbed annually by dredges.

GSRI (1974) describes a series of zoning restrictions imposed by the Louisiana Wild Life and Fisheries Commission, which include, 1) a one mile band around the perimeter of the lake, 2) a one-mile strip on each side of the Lake Pontchartrain Causeway, 3) a one-half-mile strip crossing the lake diagonally to protect high pressure gas pipelines and 4) a four mile wide area encompassing the eastern end of the lake from Goose Point to New Orleans. Dredging operations are thus prohibited in 56% of the lake, leaving 44% open to dredging. The total area of the lake is estimated by Swenson (1980) to be $1.63 \times 10^9 m^2$, 44%, or $7.17 \times 10^8 m^2$, of which is open to dredging. Dividing this figure by each of the two estimates of the area covered by dredging annually, we find that an area equal to that which is open to dredging will be covered in from 1.4 years (with 7 dredges working 360 days/yr) to 2.3 years (with 5

dredges working 270 days/yr). At this rate, one can see that not only has much of the lake been dredged, but because the entire 44% of lake bottom open to dredging is not covered in any one year, much of the lake bottom is dredged and redredged, possibly several times each year.

Scope of Present Study

The approach taken in the present study was to select two sites in proximity that would also be representative of the area subjected to the perturbation caused by dredging operations. One site would serve as the control site and not be dredged; the second would be the experimental site and would be dredged in a manner approximating normal operations. Each site would be sampled in a quantitative manner both before and after dredging on a logarithmic time scale for the first year and on a quarterly basis thereafter. Ideally, the two sites would be located somewhere in the mid-lake region and would not ever have been dredged. Unfortunately, because dredging operations have occurred pretty much at random over the past 45 years with few, if any, records having been kept, it was virtually impossible to ascertain that any particular area had never been dredged. The next best alternative was to select an area that had not been dredged recently and that would be protected from being dredged again before the completion of the study. Such an area exists along a two-mile wide swath, one mile on each side of the Lake Pontchartrain Causeway, which was first opened in 1956. A site was selected one quarter mile west of the north turnaround of the 12-mile fixed bridge, Lat. 30°12'30", Long. 90°07'42", to be the experimental site (Figure 2). The control site was placed one quarter mile north of the experimental site, one quarter mile west of the Causeway. To mark the exact location of the dredged experimental site (a condition essential to the success of a long-term study of dredging effects), a nun buoy was put in place on September 26, 1978. This buoy remained in place until some time in December 1978, when it was lost because of failure of the mooring eye. The cast-iron engine blocks used to anchor the buoy were located by magnetometer and verified by a diver on April 19, 1979, and a second, can-type buoy and anchor were placed approximately 6 feet from the original buoy on that date. On December 22, 1980, the engine blocks were again located by magnetometer and verified by divers, and a third buoy was placed on the spot.

In addition to biological sampling, a number of physical parameters were also investigated, including dissolved oxygen, pH, nutrients, trace metals, and chlorinated hydrocarbons, at the experimental and control stations. Sediment bulk density was investigated at both dredging control and experimental stations as well as at the monthly stations occupied during the concurrent benthic characterization study.

SHELL DREDGING AT THE EXPERIMENTAL SITE

On September 26, 1978, the dredge Maurepas, owned and operated by Pontchartrain Materials Corp. of New Orleans, rendezvoused with the research vessel leased by LSU for the study, at the buoy-marked dredge experimental site. The dredging of the experimental site began about 1030 hrs and consisted of 46 passes as close to the buoy as possible from all directions. Dredging was completed at about 1700 hrs. Because of the large turning radius of the dredge and the increasing wind speed from the southeast in the late afternoon, slightly more passes were made on the west side of the buoy.

The dredge Maurepas is equipped with a main pump capacity of 12,000 gallons per minute (GPM) and is powered by an 860 hp diesel engine, giving it a 150 tons per hour (TPH) solids recovery capacity. One processing plant with gravity screens of 1/2 inch mesh openings and a screw classifier are supplied wash water by two 5,000 GPM pumps, thus returning about 22,000 GPM of spoil and shell fines to the lake. The hull of the Maurepas is 165 ft x 34 ft x 11 ft, and its displacement is about 453 net tons, with a draft of between 7 and 8 feet. Propulsion is by two 400 hp diesel engines, which could conceivably affect the bottom by the propeller wash. The dredge spoil issuing from the discharge pipes can be classified as fluid mud, somewhat denser than water, which it discharges with considerable velocity. The result, as illustrated in Figure 4, is that the majority of this material rapidly sinks through 5m-deep water column, spreading over the bottom rapidly, with only a small amount of material from the spoil column forming the characteristic plume behind the dredge. The immediate effects on the ecosystem of the passage of a shell dredge can be classified into three categories: (1) the effect on the area of bottom actually sucked up into the intake pipe at the "fish mouth"; (2) the effect on the water column of the discharge of the spoil column and formation of the plume; (3) the effect on the much greater area of bottom buried by varying depths of fluid mud spoil as it spreads over the bottom. Some of this fluid mud flows into the trench left by the passage of the fish mouth and, eventually, as the trenches in a heavily dredged area coalesce, all the original, naturally bedded sediment would be replaced by a lower density, thoroughly hydrated, spoil sediment.

There is a question concerning intensity of shell dredging at the experimental station, namely, whether the 46 passes represented heavier than normal dredging or lighter than might be expected in normal dredging operations. Normally, "a dredge circles continuously, crossing and recrossing a designated area until the rate of shell recovery falls below an economic minimum for a dredge" (Arndt 1976). By comparing the amount of shell still remaining at the experiment station to the amount found at the control station from samples collected in an identical manner, some relative idea of dredging intensity could be inferred Table 1 shows the dry weight in grams of washed shell from box cores, retained on one-half inch mesh screen (same mesh size used in dredge processing plants), by sampling dates, for the first year of sampling. In every instance but one, the mean weight of shells from three box cores at the experimental station exceeded that found at the control station. The overall mean of 1,620 g of shell at the experimental

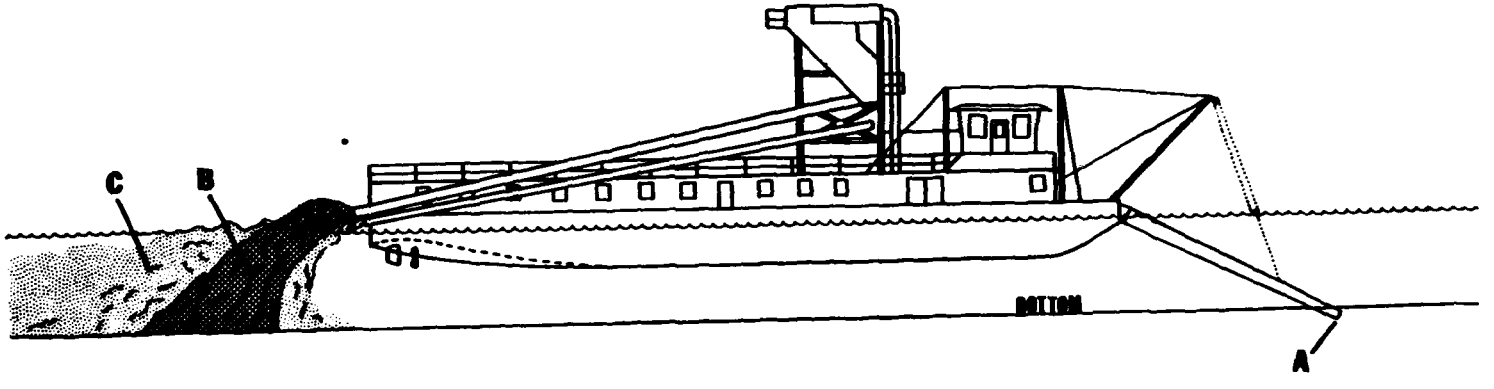


Figure 4. The Dredge Maurepas, adapted from GSRI 1974. A = "fish mouth, B = spoil column, C = plume of suspended material.

Table 1. Dry weight in grams of washed shell retained on one-half inch mesh screen from box cores

	Dredge Experimental Station DX				Dredge Control Station DC			
	Box Core A	Box Core B	Box Core C	$\bar{X}$	Box Core A	Box Core B	Box Core C	$\bar{X}$
25-26 Sept 1978	1022	1774	944	1247	1970	1755	267	1330
27 Sept 1978	1823	2471	3032	2442	---	---	---	
28 Sept 1978	548	696	1646	963	---	---	---	
1 Oct 1978	1451	2134	2201	1928	276	878	1132	762
16 Oct 1978	2576	1826	2821	2408	1303	605	1648	1185
15 Nov 1978	2760	604	424	1263	560	877	365	601
19 Apr 1979	2529	596	2295	1807	694	251	775	573
17 May 1979	646	392	1699	912	29	196	194	140
23 Aug 1979	1450	2231	40	1240	633	271	1081	662
Mean $\bar{X}$				1620				750

station is more than twice the 750 g found at the control station. There remains a considerable amount of shell at the experimental station, and under normal operations, the dredge could be expected to operate at the site until the shell is removed, thereby subjecting the site to heavier dredging intensity than was accomplished by 6½ hours of dredging on September 26, 1978.

Methods

On the day of the experimental dredging, a transect to measure dissolved oxygen in the water was made approximately one hour and fifteen minutes after the beginning of dredging. Oxygen was measured in situ with a MARTEK Model DOA dissolved oxygen meter. Three stations were sampled, beginning at approximately 150 m feet west of the Causeway, a second station approximately 275 m west of the Causeway, and the third station in the dredge plume. A mid-depth water sample was taken in the dredge plume, which was subsampled for toxic substances, nutrients, and pH. Approximately 18 liters of spoil were collected from the main discharge pipe (termed "outlet" in another section), by swinging a bucket into the discharging spoil column. This material was subsampled for toxic substances, nutrients, and meiobenthos, with the remainder being sieved for macrofauna, using the same methods described in the benthos section of this report.

Results

The results discussed in this section will be restricted to the sampling on the day of dredging. Table 2 shows the results of the dissolved oxygen transect from the west side of the Causeway to the dredge plume. Dissolved oxygen in the water column appears to be about the same, with some variation at the first two stations, with about 9.0-9.2 ppm at the surface, and about 8.4-8.6 ppm at 2 to 4 meters. In the plume the dissolved oxygen dropped about 1 ppm consistently through the water column to 8.0 ppm at the surface and 7.6 to 7.4 at 2 and 4 meters, respectively. The pH at the mid-water control site was 7.5; at the plume it was elevated to 8.3.

Study of samples of meiobenthos from discharged spoil indicates that very few animals made it through the process in recognizable condition. The actual counts of animals are given in Table 3. It should be noted, however, that because of methodology of preserving meiofauna, there is no way to determine whether the animals had been viable when collected. In the remainder of the sample, which was sieved for macrobenthos, no living animals or remains of living animals, which might have been damaged, were found.

Discussion

Oxygen uptake by resuspended anaerobic sediments could be significant. A study by Berg (1970) showed uptake rates could be as high as 56,760 to 83,040 mg/l of oxygen per hour at resuspension rates of 2,000 mg/l total

Table 2. Dissolved oxygen transect from west side of Causeway to Dredge Plume, in ppm, by depth

Station Time	150 m West 1145	275 m West 1200	Plume 1210
Surface	9.2	9.0	8.0
1 m	8.8	8.7	8.0
2 m	8.4	8.6	7.6
3 m	---	8.4	7.4
4 m	---	8.4	7.4

Table 3. Meiobenthos from discharged spoil (Number per 150 cm^3)

Taxa	Core				
	1	2	3	4	$\bar{X}$
Nematodes	1	0	1	0	0.5
Copepods	1	1	0	0	0.5
Nauplii	6	3	5	4	4.5
Ostracods	0	0	0	0	0.0
Rotifers	0	1	0	0	0.25
Turbellarians	0	1	0	0	0.25
Polychaetes	0	1	0	0	0.25
Oligochaetes	0	0	0	0	0.0
Bivalves	0	1	0	1	0.5
Gastropods	0	0	0	1	0.25
TOTAL	8	7	7	6	7.0

volatile solids of suspended benthic sediments. Contrary to what these results might have predicted, the dissolved oxygen in the dredge plume, although somewhat depressed (by about 1 mg/l), was not nearly as low as might have been expected. This apparent anomaly could be explained by three factors. The first is that oxygen demand of the sediments may not be very high. The second is that a large quantity of air is introduced into the sediments in the processing plant of the dredge. Billions of small air bubbles are, in a sense, beaten into sediments as they are washed from the shell. The partial pressure of a gas in a bubble is considerably higher than ambient conditions, and a lot of oxygen can thus be introduced into the fluid mud slurry. A third factor is the amount of suspended material in the dredge plume. Although we did not measure it on the day of dredging, we had the opportunity to measure the amount of suspended material in a dredge plume in April 1979. We found that at a distance of about 10-15 meters from the dredge, the concentration of suspended sediments was only 240 mg/l total. This finding also lends credence to the hypothesis that most of the spoil material goes straight to the bottom in a column and that very little actually forms the plume.

The slight elevation in pH measured in the dredge plume could be explained by the release of carbonate and ammonia. Of these two, the bicarbonate ion (HCO_3^-) is more important in elevating pH. While the shells are in the sediments, there is the likelihood that a micro-dissolution layer may form around the shell, producing Ca^{2+} and HCO_3^- ions. The sudden washing away of this layer around each shell could put a large quantity of bicarbonate ions into the plume, thus elevating the pH to 8.3, a value that nearly coincides with the equilibrium peak for bicarbonate.

This report does not offer any conclusions regarding the ability of living benthic animals to survive the dredging process. We are unable at this time to sample adequately the spoil issuing from the discharge pipe because of the great force of the existing spoil. In addition, a considerably larger sample than we were able to obtain would be necessary. If we consider that the majority of living animals are restricted to the top two centimeters of the sediment, which is thoroughly mixed, with 74 or more centimeters below, a dilution factor of 38 to 1 results. If the wash water were added, the dilution factor would greatly increase.

PHYSICAL EFFECTS OF SHELL DREDGING ON BOTTOM SEDIMENTS

Two of the most obvious, although not necessarily the only, effects of shell dredging on bottom sediments are the burial of the existing sediment surface with spoil and the lowering of sediment density. In the process of hydraulic shell dredging, the fluid mud spoil exits the discharge pipes with considerable velocity and force. The result is that the majority of this material sinks rapidly through the water column to the bottom, where it spreads out over the bottom. As this material spreads out and is redistributed, it buries the preexisting top layer of oxidized sediments and animals with shell fines and gray-colored spoil that has an oxygen demand (Figure 5). This phenomenon was observed at the experimental dredge site, and it is estimated that an area upwards of one-quarter of a mile or more in diameter was thus affected. Although the critical measurements of Eh were not made, the observations will be described. The question of exactly how this affects the biota is not yet known. Are the animals smothered outright or do they finally die as the oxygen in the buried oxidized zone is depleted? Are the animals even capable of moving up through the oxygen-poor spoil zone? If they are capable of moving through it, would the descending oxygen depletion gradient prevent them from moving upward against gravity to the new top layer of the spoil zone? To answer these questions, cores would have to be quick-frozen in the field and layer-sectioned for vertical distribution. The sequence of field sampling would have to be on the scale of hours to several days. Another approach that deserves serious consideration for future research is the use of x-radiography to determine the exact position in the sediment of organisms such as bivalves to determine whether clams can move up through the spoil covering.

The second obvious physical effect on the bottom sediments occurs in the processing plant on the dredge and is the result of washing the mud from the shell. The resulting spoil is a homogeneous mixture of sediment and wash water, a fluid mud with the consistency of latex paint. The sediment type at the dredge control and experimental sites is a silty clay, a type that predominates over most of the central lake region (Barrett 1976; Bahr, Sikora, and Sikora 1980). A quantifiable sediment property that could be used to measure extent of change in the sediment is bulk density. Bulk density is the actual weight per unit volume of intact sediment. Richards and Parks (1976) report bulk density of 1.42 g/cm^3 for silty clay in North Pacific Continental Shelf and slope sediments. Although admittedly, conditions on a continental shelf are different than those found in the lake, there is a paucity of bulk density values in the literature, and results reported in this section from a station presumed to be undredged approximate this value. Sediment bulk density is a key factor when considering or calculating the energy required to resuspend sediments. Over a large area of lake, a decreased bulk density of the sediments would be a significant factor in increasing suspended sediments and thus increasing turbidity. Lowering critical resuspension velocity would result in more sediment being resuspended more of the time. There is some evidence that this has occurred over the past twenty to thirty years (Dow and Turner 1980) and the possible implications to the lake ecosystem could be drastic, because primary production could be lowered and the siltation and smothering of benthic infauna could result. If, on the other hand, resuspended sediments carried with them certain toxicants, the possible effects could be far worse.

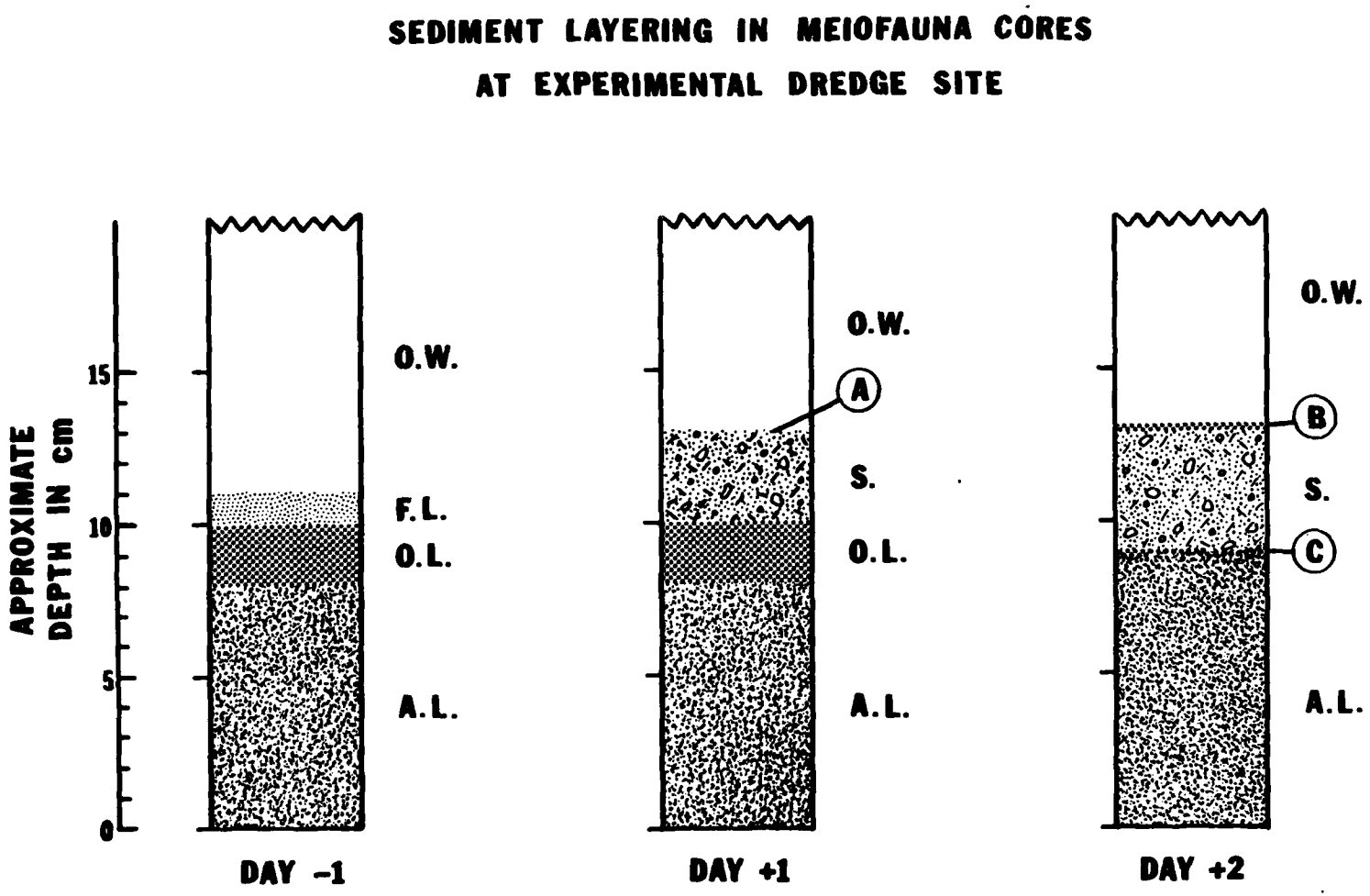


Figure 5. Burial of sediment surface by dredge spoil at experimental dredging station. Day-1 is the day before dredging; Day +1, one day after dredging; Day +2, two days after dredging. OW = overlying water, FL = fluffy layer, brown in color, OL = oxidized layer, light brown in color, AL = anaerobic layer, gray in color, S = spoil, gray in color, A = spoil surface no oxidized layer, B = spoil surface with very thin brown oxidized layer, C = remnant of original oxidized layer brown in color.

Methods

Bulk density is the actual weight per unit volume of intact sediment, with differences between sediments of same grain size and similar organic content being the result of sediment consolidation. Consolidation of sediments, corresponding most closely to compaction in geological terminology, is defined as "every process involving a decrease of the water content of a saturated sediment without replacement of the water by air" (Terzaghi 1943) or "the gradual process which involves, simultaneously, a slow escape of water and a gradual compression, and which also involves a gradual pressure adjustment" (Taylor 1948). Bulk density can be used as a measure of the consolidation state provided that the grain size distribution is taken into account. In the present study, all the sediment is of the silty clay type, thus making comparisons possible. We measured both in situ bulk densities at a number of stations maintained in the concurrent benthic characterization study and bulk density over time in a laboratory consolidation experiment on dredge spoil obtained from the dredge Lacombe while operating in the western part of the lake south of station 7 and west of station 8 on June 28, 1980, (the dredge Lacombe was previously the Kathy L., as described by GSRI 1974). Fifty polyethylene buckets with tight-sealing lids of 18 liter capacity were filled with spoil by means of a 5 cm hose fitted to the lower side of the discharge pipe of the dredge. The spoil was brought back to the laboratory and dumped into 16 columns made from 3.2 mm thick, 30 cm diameter PVC pipe with 6.4 mm PVC bottoms. The columns were placed in two racks holding eight columns each and were filled to a height of 75 cm. These columns were fitted with the same tight-sealing lids by cutting off the top portion of the buckets and riveting them to the tops of the columns and sealing them with silicone caulking. The lids, after being placed on the columns, were taped with electrical tape to produce a completely watertight seal. The columns were sampled for bulk density on days 0, 6, 12, 18, 27, and 226.

Bulk density samples were obtained using core tubes, which were made of 5 cm long core segments taped together with waterproof tape to form one core tube. The in situ samples were taken with core tubes constructed of core segments cut from standard 50 cc plastic syringes, 2.55 cm in diameter, with the leading segment beveled to form a cutting edge. Each core segment was numbered and premeasured for volume. The core tube was inserted slowly into the box core sample with a gentle rotation. After removing the core tube containing the sample, the outside was washed and the tape holding each segment was cut, and a piece of preweighed aluminum foil was inserted as the segment was removed and placed in a preweighed plastic vial with a tight-fitting cap. The samples were refrigerated, brought back to the laboratory, and weighed. Total sediment weights for each 5 cm sediment interval were calculated by subtracting plastic vial weight, core segment weight, and aluminum foil weight for each sample. Sediment bulk densities in g/cm^3 were calculated by dividing total sediment weight by the core segment volume. Bulk densities for the consolidation experiment were taken in a similar manner; glass core tube segments 5 cm long and 3.3 cm in diameter, however, were used. Also, because of the nature of the spoil, it was not possible to insert a core tube by gently rotating it. A small vibro-engraver was attached to the top of the core by rubber bands,

making a minivibracorer. This device allowed the core tube to be inserted with minimal effects on the sediment. No aluminum foil was used; rather, the tape was cut with a modified spatula, sharpened on the leading edge and with a hole cut toward the back over a screw cap. This allowed the sample and core segment to be slid back and dropped directly in the plastic jar with a tight lid. Bulk densities were determined in the same manner as the in situ samples.

Results

The initial bulk density of the spoil was 1.22 g/cm^3 and was uniform throughout the column. By day 6, the sediment surface had dropped 6 cm. The bulk densities by 5 cm depth intervals, however, were not uniform throughout the column but increased with depth to 1.26 g/cm^3 5 cm above the bottom (Figure 6). The bottommost 5 cm was not sampled for bulk density to ensure against even the slighest disruption of this segment during capping and removal of core tube from the column. The topmost 5 cm segment remained at the initial bulk density of 1.22 g/cm^3. As the experiment progressed, the sediment surface continued to drop, 8.7 cm by day 12, 11.5 by day 18, and 13.5 by day 27. The topmost 5 cm segment gained very little in bulk density, rising to 1.23 on days 12 and 18 and to 1.24 on day 27. The lower segments in the column continued to gain in bulk density from 1.29 g/cm^3 to 1.35 g/cm^3 on day 27. The spoil sediment begins consolidating at the bottom of the column, displacing water upwards through the upper segments of sediment. This movement of water apparently keeps the upper segments at a lower bulk density not significantly different from the initial density. By day 226, the bottom bulk densities are approaching reported literature values of 1.42 g/cm^2 for undredged silty clay, and the upper segments have gained considerably in density. The sediment surface has dropped 29.3 cm, or 39% of the height of the sediment column, although the rate of drop has declined to less than an average of 0.7 mm/day (Figure 7).

The in situ bulk densities are compared for the dredging control station, the dredging experimental station, and stations 2, 8, and 12 from the benthic characterization study as well as day 12 and 226 of the consolidation experiment (Table 4). These stations are all of the silty clay sediment type with similar organic contents (Table 5).

Discussion

The results of the sediment consolidation experiment were quite unexpected, particularly the non-uniform consolidation with lower segments consolidating at a faster rate. The displaced water, however, caused the upper layers of sediment to remain unconsolidated, and it is in the upper layers, particularly the top 5 cm, where the benthic fauna reside. The entire consolidation process appears to take place rather rapidly, and we believe it is not so rapid in the field. If one looks at the drop in the sediment surface, one might conclude that a trench dug by a dredge and filled with spoil which issued from the back end got deeper as time progressed. This is probably not the case; dredge cuts would tend to fill in over time. The movement of sediment by wave action and

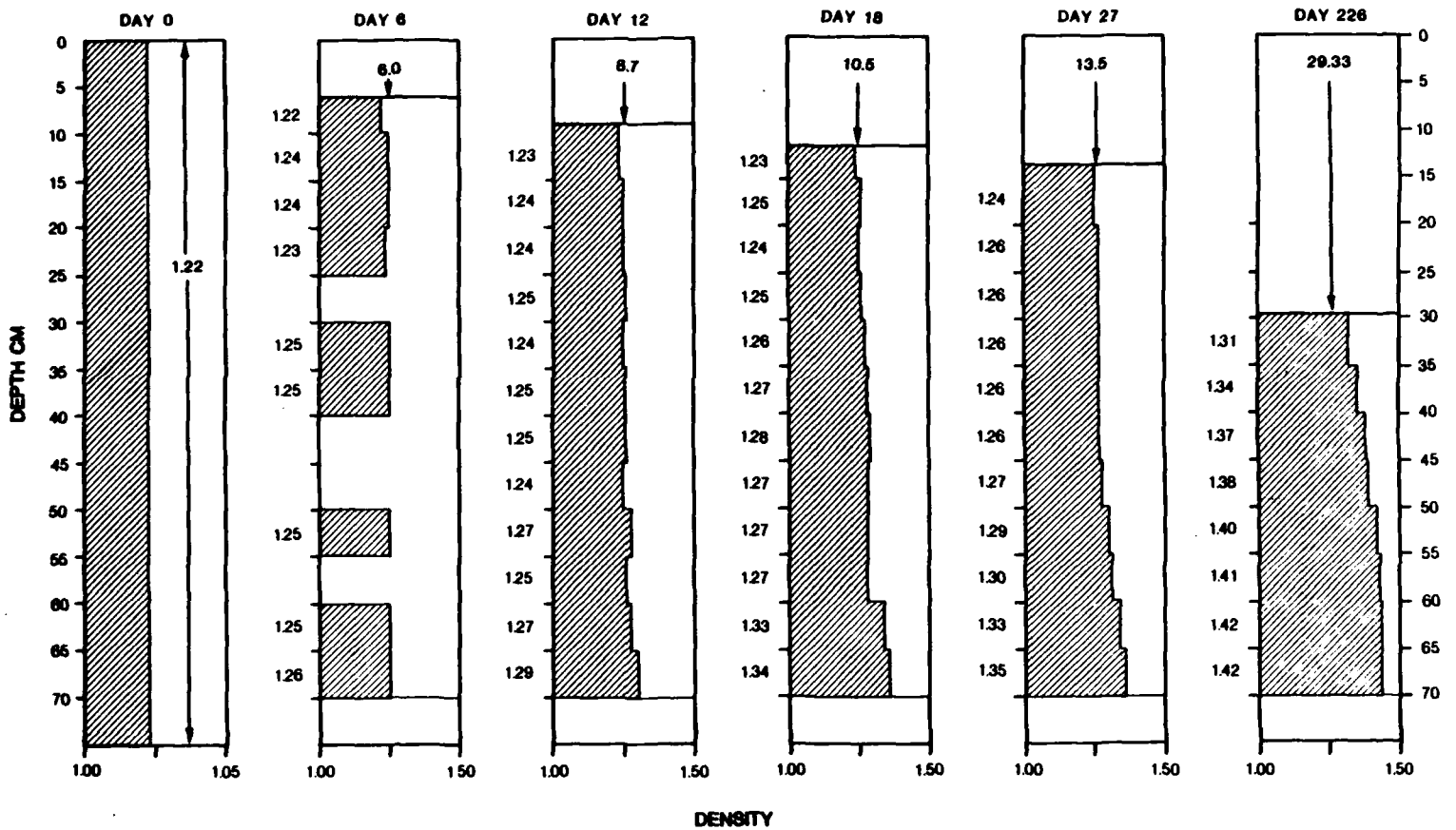


Figure 6. Spoil consolidation showing progressive days of experiment on top, sediment surface drop in centimeters and indicated by arrows. Bulk densities in $g/_3cm^3$ by 5 cm intervals in depth at left of each column. On Day 0, bulk density of 1.22 g/cm^3 uniform throughout column. On Day 6 several samples were lost. Bottom 5 cm (70–75 cm) not sampled.

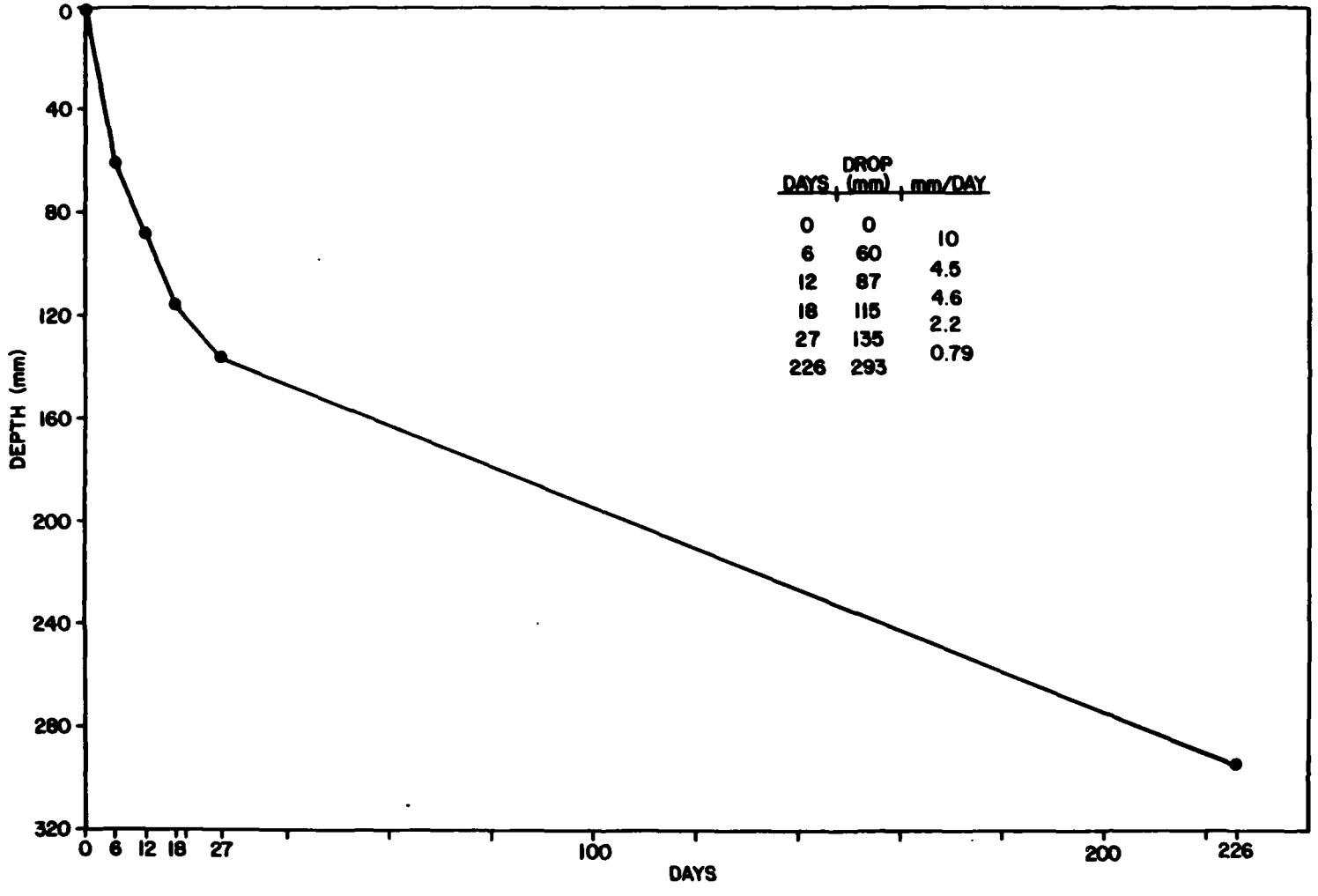


Figure 7. Spoil consolidation drop in centimeters of sediment surface by time in days.

Table 4. Sediment bulk densities, by depth, at Dredging Control Station (DC) and Dredging Experimental Station (DX), Day 12 and Day 226 of the consolidation experiment and three stations from the benthic characterization study of the same silty-clay sediment type

Depth cm	1 STA 2	2 DX	3 DC	4 DAY 12	5 DAY 226	6 DAY 226	7 STA 12	8 STA 8
0								
	1.28	1.31	1.29	1.23	1.31		1.28	1.17
5								
	1.41	1.15	1.22	1.24	1.34		1.27	1.36
10								
	1.41	1.23	1.26	1.24	1.37		1.25	1.31
15								
	1.43	1.28	1.23	1.25	1.38		1.27	1.30
20								
	1.43	1.42	1.26	1.24	1.40	1.31	1.30	1.30
25								
	1.45	1.34	1.24	1.25	1.41	1.34	1.31	1.32
30								
		1.29	1.27	1.25	1.42	1.37	1.33	1.36
35								
				1.24	1.42	1.38	1.35	
40								
				1.27		1.40		
45								
				1.25		1.41		
50								
				1.27		1.42		
55								
				1.29		1.42		
60								

Table 5. Total organic carbon analysis in percent weight, by depth, at three stations from the benthic characterization study, Dredging Control Station (DC), Dredging Experimental Station (DX)

Depth cm	STA 2	STA 8	STA 12	DC	DX
0					
	1.52	2.05	1.44	1.83	1.75
5					
	1.77	0.96	1.27	1.70	1.66
10					
	1.21	1.54	1.21	1.85	1.80
15					
	1.23	1.41	1.11	1.86	1.71
20					
	1.10	1.67	1.26	1.33	1.67
25					
	1.15	0.89	1.31	1.68	1.65
30					
		0.79	1.41	1.17	1.73
35					
			1.79		
40					

its sedimentation in a trench, however, would provide a continual, low bulk density environment for benthic organisms. Large, dense benthic organisms such as large Rangia may not be able to survive in these low density sediments, which might be a contributing factor in explaining the paucity of large living Rangia presently in the lake, as found by this study and the concurrent benthic characterization study.

During a 1954 quantitative survey of the benthos Darnell (unpublished manuscript) recorded densities of 141 Rangia per m^2 larger than 20 mm at a station closest to the experimental dredging and control stations of the present study. During the present study, at the dredging control station, we found from 36 box core samples a mean of 0.9 ± 0.9 per m^2 of Rangia larger than 20 mm. In the eastern part of the lake at six stations, all of which were out from the periphery, Darnell in his 1954 survey found densities of Rangia larger than 20 mm to be 173 ± 31.1/m^2 from 12 Peterson grab samples. Tarver (1973) reported that during his study of Rangia cuneata, virtually no clams larger than 16 mm were found in areas which were continually dredged, and that eastern Lake Pontchartrain was void of this size clam although the periphery of the lake had large concentrations.

It seems certain that conditions in the open lake that are detrimental to large Rangia have arisen in the past 26 years. Both low density sediments unable to physically support large Rangia as well as increased suspended sediments in the near bottom environment could contribute to these detrimental conditions. We know that dredging results in the production of low density sediments. Future research is needed to investigate the suspended sediments in the near-bottom environment.

The precise bulk density value of the sediments at the sediment-water interface is unknown and difficult to measure. A further complicating factor is the existence of a fluffy, organic-rich layer at this interface. The bulk density values reported in Table 4 for the uppermost segment of each column are for the first 5 cm of depth as a unit. This segment does not appear very different from the in situ stations except, perhaps, station 8 with a value of 1.17. However, attention should be focused on the 5-10 cm depth that supports the upper 5 cm. The dredging experimental station, dredging control staton, and station 12 all exhibit lower densities at the 5-10 cm depth. Conceivably, a large Rangia, of 40-50 mm, lying under the sediment surface would just about protrude into the 5-10 cm segment with its shell, while its foot would penetrate well below 5 cm. Also, the activity of the animal would tend to disrupt the sediment immediately surrounding it. If a clam were to experience continual sinking, with a continuous expenditure of energy required to keep its siphons above the surface of the sediments, (a situation for which it is not adapted), its growth would be inhibited and its survival would be threatened. If additional environmental stresses were added, survival could become impossible.

A comparison of the bulk densities by depth in Table 4 yields some striking similarities and a notable exception. This exception is station 2, with bulk densities approaching and exceeding the literature value of 1.42 for silty clay (Richards and Parks 1976). These data

indicate that station 2 has not been dredged. Conversely, comparing the dredging control station to day 12 of the consolidation experiment, they appear quite similar, as do the top four segments at station 12. If we reference the sediment column of day 226 to the bottom, as in column 6, we find striking similarities at the same depths at station 12 and station 8. Station 12 seems to behave as the postulated model for a dredge cut in the field, with the original spoil surface sinking to a depth of 20 cm and additional low density material filling in on top. A similar sequence may have taken place at station 8 over a longer period of time. The dredging experimental station showed alternately very low densities and higher densities, which might be the result of slumping. Station 12 appears to have been heavily dredged; nine box cores taken quarterly yielded a mean of 32 g of shell retained on one-half inch mesh screen (compared to means of 1620 g at station DX and 750 g at station DC). In a 1954 survey of the benthos, Darnell (unpublished manuscript) had a station in almost exactly the same location, and he recorded a mean of $132/m^2$ of Rangia larger than 20 mm. Shells from these animals alone would amount to more than 32 g per box core. If we accept station 12 as having been dredged, then we must also accept the dredging control station as having been dredged at some time prior to construction of the Causeway.

In perspective then, the results found in this study obtain primarily from differences between a dredged station and a station that has not been dredged in at least 27 years. We do not have a pristine, undredged station to compare with, nor do we have data from a pristine Lake Pontchartrain before dredging. Consequently we are comparing the effects of dredging on a benthic community that has survived dredging for 48 years and may indeed be composed of only the hardiest organisms of the original community.

THE EFFECTS OF SHELL DREDGING ON THE NUTRIENT AND HEAVY METAL CHEMISTRY OF THE WATERS AND SEDIMENTS OF LAKE PONTCHARTRAIN

Introduction

A control and a test or experimental dredging site were established within a central area of the lake which had not been disturbed recently by hydraulic shell dredging. Filtered and unfiltered samples were collected prior to dredging from a depth of one meter (subsequently referred to as mid-water) and from the water/sediment interface; sediment samples were also taken. The concentrations of nitrogen and phosphorus compounds and of the heavy metals lead, zinc, copper, iron, manganese, and cadmium were determined. The same measurements were made on mid-water samples collected from the test site at the point of dredging, and one hour after completion of dredging on the same day. A full quota of samples was taken at the dredged site on Days 1 and 2 following dredging, and from Day 5 onwards, water and sediments were sampled at both control and dredged sites (see Table 6).

Literature Review

Nutrients

Mortimer (1941) he found that phosphorus was released from anaerobic sediment systems when the ferric iron content was reduced to ferrous iron. He proposed that the capacity of the sediments to hold phosphorus was related directly to the presence of ferric iron. The reduction of the mud surface corresponded with an increase in the transport of nutrients to the overlying water (Mortimer 1941 and 1942).

It has since been shown that phosphorus is released from sediments under aerobic conditions (Olsen 1964). The uptake of phosphorus in the water by algae was followed by the release of phosphorus from the sediments, and in oxidized sediments the exchange was more complete. Gahler performed laboratory experiments which showed that the concentrations of nitrogen and phosphorus increased in overlying water in both aerobic and anaerobic environments (Gahler 1969). This increase in nitrogen and phosphorus concentrations was associated with resuspension of the sediments.

The ability of sediments to supply nutrients to the water column has been much studied. Sediments and interstitial waters have been analyzed chemically. Livingstone and Boykin (1962) extracted the sediment cores from a Connecticut pond with deionized water. They found that more phosphorus was released from shallow cores (10 cm) than from deeper cores (10 m). It was observed that as the ratio of water to sediment increased, the amount of phosphorus released in solution also increased. The time basis of the phosphorus release was not reported.

Byrnes et al. (1972) measured the exchange of (NH_4^+)-N across a sediment/water interface. Sediments at rest were spiked with additional

Table 6. Date and days of sample collection

Date	Day	Dredged (Sample Nos.)			Control (Sample Nos.)		
		Mid-water	Interface	Sediment	Mid-water	Interface	Sediment
1978 Sept. 25	-1	1F 1U	2F 2U	S3			
Sept. 26	0 0 hr.*	7F 7U			4F 4U	5F 5U	S6
Sept. 26	0 1 hr.+	8F 8U					
Sept. 27	1	9F 9U	10F 10U	S11			
Sept. 28	2	12F 12U	13F 13U	S14			
Oct. 1	5	15F 15U	16F 16U	S17	18F 18U	19F 19U	S20
Oct. 16	20	21F 21U	22F 22U	S23	24F 24U	25F 25U	S26
Nov. 15	50	27F 27U	28F 28U	S29	30F 30U	31F 31U	S32
1979 Apr. 19	205			XS_1-XG_5			CE_1-CG_5
May 17	233			SDX_1-SDX_3			SDC_1-SDC_3
June 12	259	DX_1-DX_3			DC_1-DC_3		

*0 hr during dredging. +1 hr one hour after cessation of dredging

nitrogen and the (NH_4^+)-N rate of release determined. The sediments released (NH_4^+)-N at a linear rate for a period of 8 hours.

Considering the results of Gahler's experiments in which it was shown that the nutrients, artificially released into the water column, supported algal growth, there seems little doubt that sediments contain nitrogen and phosphorus in forms releasable to overlying water under either aerobic or anaerobic conditions. Also, the released phosphorus and nitrogen compounds are available as growth supporting nutrients.

Once it was thought that only thin layers of surface mud were involved in water/sediment exchange processes. However, Bricker and Troup (1975) studied the behavior of the sediment/interstitial water environment and the importance of this environment to the chemistry of the estuarine system in Chesapeake Bay. It was reported that a rapid exchange of nutrients between overlying water and the upper part of the sediment column took place to depths of 30 cm. Nutrients in solution were exchanged across the water/sediment interface in response to the physical disturbance of the sediments by tidal or storm-induced currents. Hynes and Greib (1970) found that inorganic phosphate and (NH_4^+)-N moved easily through anaerobic sediments by diffusion mechanisms, suggesting that more than a thin surface layer was involved in exchange processes with the overlying water. Porcella et al. (1970) experimented with water columns over sediment cores 15 cm long. Phosphorus was released into the water from the entire length of the core.

The behavior of phosphorus in Wisconsin lakes has been reported by Bortleson (1970) and Sridharan (1973). Lee (1970) concluded from an examination of these reports that there was evidence for the existence of a sediment mixing zone which extends to at least 10 cm below the water/sediment interface. It was suggested that the composition of the sediment and the mixing energy of the system, be it natural or artificial, governed the depth to which sediments were involved in exchange processes with overlying water.

The results of many studies of the effects of disturbance or agitation on the release of nutrients from sediments have been reported. There is general agreement that agitation of sediments causes their resuspension into the water column and promotes the release of nutrients.

Zicker et al. (1956) buried labelled phosphorus at various depths in sediment cores and measured the release of phosphorus to the water on agitation of the sediments. An increase in the release of phosphorus to the water was measured and was thought to be due to an increase in the surface area of sediment exposed to the water column. Austin and Lee (1973) found that active mixing of sediments caused the release of nitrogen to be fifty times greater than that released from undisturbed sediments.

Ruttner writes in Fundamentals of Limnology (1963) that the mixing of natural water sediments is associated with the transport of chemical materials from the sediment to the water column. Wind-induced currents, eddies, and benthic activity are cited as mixing mechanisms which induce exchange between sediments and water. Gahler (1969) observed large

increases in total phosphorus and (NH_4^+)-N concentrations of an area of Lake Klamath where sediments had been mixed by benthic algae.

Austin and Lee (1973) found that where there was little mixing of the sediment/water interface. Nutrients were held in the sediments and were not released in any quantity to the water. It was expected that the very drastic mixing of sediments occurring during shell dredging would cause large increases of nutrients to the water column.

Summary

There is much available information on the nutrient composition and behavior of water/sediment systems.

Phosphorus and nitrogen compounds are released from sediments to overlying water in forms suitable for uptake as nutrients.

Nutrients are released under aerobic and anaerobic conditions, with associated resuspension of sediments, and inorganic phosphate and (NH_4^+)-N diffuse easily through the sediments.

The transport of nutrients to the water column takes place from depths of up to 30 cm within the sediment.

Sediments at rest release added nutrients into the overlying water at a linear rate for a period of eight hours.

The natural or experimental agitation of sediments causes the release of nitrogen in quantities up to fifty times greater than those released from undisturbed sediments.

Any mixing mechanisms, no matter how minimal, induce large increases in the release of nutrients from more than the surface layer of the sediment.

It is therefore expected that a study of the effects of shell dredging on the water and sediment chemistry of Lake Pontchartrain will show increases in the nitrogen and phosphorus content of the water at times close to the dredging operation. These increases in the water may also be associated with decreases in the concentrations of nitrogen and phosphorus in the dredged sediments.

Metals

Heavy metals are known to be toxic at certain concentrations to aquatic organisms. The Environmental Protection Agency (EPA) has prepared Quality Criteria for Water (1976) giving acceptable levels of heavy metals (Table 7).

Table 7. Criteria for water quality-heavy metal concentrations (EPA 1976)

Element	Fresh Water µg/l	Marine Water µg/l
Cd	<30	10
Cu	>30	50
Fe	300	300
Mn	50	100
Pb	30	50
Zn	1	100

These criteria were developed from bioassay studies in which organisms were exposed to metals in solution. Criteria which govern the behavior of metals in solution cannot be applied directly to particulate material or to sediments. The metal composition of the sediment may not be related directly to its pollutional characteristics when resuspended in water, as happens during dredging operations (Windom 1973).

Windom prepared criteria for a few metals in sediments suitable for overboard disposal in marine waters which were used by the U.S. EPA Region IX (Table 8).

Table 8. U.S. EPA Region IX metal criteria for the suitability of sediments for overboard disposal

Element	Criteria mg/Kg dry weight
Cd	2
Cu	50
Pb	50
Zn	75

The presence or absence of oxygen is considered to have great influence on the release of metals from dredged materials (Lee 1975). Mortimer (1971) observed that a progressive decline in oxygen concentration to analytical zero at the sediment interface of Esthwaite waters could be correlated with the mobilization of iron and manganese in the sediments and also with their transfer into the water. Phosphate and ammonia were released concurrently.

Mortimer further suggested that an oxidized microlayer that existed at the sediment interface trapped iron and manganese effectively.

Gorham and Swaine (1965) found an increase in the concentrations of iron and manganese in oxidized sediments. They were of the opinion that

the ferro manganese minerals were capable of scavenging other metals such as lead and zinc from the water and accumulating them in the sediments. In 1968 Jenne wrote that hydrous iron and manganese oxides played a significant role in the control of the levels of manganese, iron, copper, and zinc in soils and water.

The changes in heavy metal concentrations resulting from maintenance dredging were explained by Windom (1973) in terms of the reaction of hydrous iron oxide to the concentration of dissolved oxygen. In separate studies, May (1973) and Lee (1975) concluded that oxygen in solution can control the release of metals from dredged material. May (1973) monitored the concentrations of various metals in filtered water samples before, during, and after dredging operations and detected no significant change.

Windom (1973) studied the effects of dredging on the water quality of several estuaries of the southeastern United States. Sediment samples were dispersed in overlying water, maintaining the sediment-to-water proportions of dredge effluent. The concentrations of iron and other metals increased initially in the water, followed by some decrease. Eventually, after several days, an increase to the original concentration levels, or higher, was observed.

These changes in the metal content of the water were explained in terms of the behavior of iron in natural waters. Reduced ferrous iron, released from dredged material, is oxidized to insoluble ferric iron in the water column and precipitated with the settling of the sediments. Other metals that are adsorbed on the insoluble ferric iron precipitate are carried down and deposited on the bottom. With a return to reduced conditions, the metals are released, and shortly thereafter the original metal concentrations are reestablished.

The release of iron from the sediments of Mobile Bay was studied during dredging. During the first day of dredging, an increase was observed, followed by a rapid decline of a further two days, and a return to the original level after two weeks (May 1973).

Lee (1975) developed an elutriate test in the laboratory, in which the environmental conditions under which metals were released during disposal of dredged sediments were simulated. The results of this experimental work indicated that under oxidizing conditions manganese was released from both freshwater and marine sediments and zinc was removed from solution. The release of manganese and the removal of zinc were proportional to the percentage of sediment in the total volume, up to 20% sediment.

In anaerobic conditions, iron, manganese, and lead were released into solution. Greater quantities of manganese were released anaerobically than were released aerobically. Variations in the oxidizing conditions did not affect significantly the concentrations of copper and cadmium.

Summary

The indications from the literature are that the response of the heavy metal content of water and sediments to dredging is controlled largely by the reactions of hydrous iron and manganese oxides. Iron in particular is more soluble in the reduced form.

When anaerobic sediments containing reduced iron oxide come into contact with air and the aerated water column during dredging, metal concentrations in the water increase initially. The reduced iron is oxidized to ferric oxide in the aerobic conditions. The insoluble ferric iron precipitates out and is deposited on the lake bottom with the resettling of the suspended sediments that follow disposal of dredged material. Other metals adsorbed on the ferric oxide precipitate are scavenged from the water column to accumulate in the bottom sediments.

Dredging activity would, therefore, very temporarily be expected to increase metal concentrations in the water column. Soon, however, under oxidizing conditions, decreases in metal concentrations should follow as they are removed by adsorption on precipitating ferric oxide. With the reestablishment of reducing conditions within the redeposited dredged material, some resolution of metals should occur, and within days, initial concentrations be restored in the water column.

Materials and Methods

Sample Collection and Preservation

Water samples were collected in 1000 ml polyethylene bottles that had been acid-washed previously in the laboratory. Raw water, filtered in the field through a 0.45μ membrane Millepore filter, constituted filtered water. All water samples, whether filtered or unfiltered, were placed on dry ice in the field and transported to the laboratory freezer, where they were stored until time of analysis.

The sediment samples were taken with a box corer. Aliquots of sediment were placed in plastic Ziploc bags, frozen on dry ice, and transferred to the laboratory freezer to await analysis. Interface samples were siphoned from the sediment-water interface into polyethylene bottles and were treated as above.

Nutrients

Pretreatment of Samples

Immediately prior to analysis, samples of filtered water and unfiltered mid-water were allowed to defrost under cold running water. The filtered water samples were analyzed for (NH_4^+)-N, total inorganic nitrogen (T.I.N.) and orthophosphate (ortho PO_4). The concentration of total phosphorus (tot-P) was measured in unfiltered mid-water samples.

Due to highly varying amounts of sediment in unfiltered interface water samples, it was decided to express the concentrations of total phosphorus (tot-P) and total Kjeldahl nitrogen (tot Kj-N) on a percentage dry weight of sediment basis rather than per unit volume of water. For this reason, unfiltered interface water was defrosted and filtered through a 0.45μ Millepore membrane filter. The particulate material was oven-dried at 60°C for 24 hours, finely ground, and stored. The dry particulate was analyzed for tot-P and tot Kj-N.

Frozen sediment samples were shattered while still in the Ziploc bags. Randomly selected pieces were oven-dried at 60°C for 24 hours, finely ground, and stored. As with the particulate from the interface water, the concentrations of tot-P and tot Kj-N were measured.

Analytical Methods

A. Filtered Water

1. (NH_4^+)-N. Filtered water that had been defrosted was shaken thoroughly, and a 25 ml aliquot was pipetted into a 100 ml distillation flask. The aliquot was made basic by the addition of 0.05 g light MgO, converting NH_4^+ ions into NH_3 molecules. The basic sample was steam distilled for 3 minutes on a generator regulated to produce distillate at a rate of 6.5 ml/min. The distillate was collected in a 50 ml volumetric flask containing 1 ml 0.1 N HCl, thus stabilizing the NH_3. The NH_3 was oxidized to NO_2^- with a sodium hypochlorite/potassium bromide/NaOH solution, and the oxidation allowed to proceed for 30 minutes. Any excess oxidizing agent was poisoned by the addition of a sodium arsenite solution. After 5 minutes, the NO_2^- was complexed with sulphanilamide in acid solution. Finally, diazotization by napthyl ethylene diamine dihydrochloride resulted in the formation of a highly colored azo dye, the absorbance of which was measured spectrophotometrically at 543 nm (Strickland and Parsons 1972, Ho and Schneider 1974).

2. T.I.N. $(NH_4^+ + NO_3^- + NO_2^-)$-N. A 25 ml volume of filtered water was measured into a 100 ml distillation flask. Light MgO was again added to convert NH_4^+ to NH_3. To reduce NO_3^- and NO_2^- to NH_3, 0.3 g Devardo's alloy was added. With all nitrogen in the form of NH_3, the treated sample was steam distilled for 4 minutes, and the distillate was collected in a 50 ml volumetric flask containing 1 ml 0.1 N HCl. The subsequent treatment and color development followed that given above for (NH_4^+)-N (Ho and Barrett 1975).

3. Orthophosphate - (Inorganic, molybdate reactive phosphorus). A 100 ml sample of filtered water was pipetted into a 250 ml separatory funnel, which had been acid-washed, and stored full of 0.4 N HCl until needed. Five ml of 25% H_2SO_4 were added, followed immediately by 5 ml ammonium molybdate solution, to

form a phosphomolybdate complex. Reaction time was 5 minutes. The phosphomolybdate complex was extracted into ethyl acetate, and the bottom aqueous layer was discarded. Ascorbic acid solution and potassium antimonyl tartrate solution were added in quick succession, the mixture was shaken, and on separation, the aqueous layer was again discarded. The resulting blue organic layer containing the reduced phosphomolybdate complex was drained off and made up to 10 ml with 90% ethyl alcohol. Absorbance was measured spectrophotometrically at 690 nm against a cell blank of 50% alcohol. The above method was a modification of Strickland and Parsons (1972) by Ho and Schneider (1974).

B. Unfiltered Mid-water

4. Total Phosphorus (tot. inorg P + tot. org P). A 50 ml aliquot of unfiltered mid-water was measured into a 125 ml Erlenmeyer flask, and 1 ml 75% H_2SO_4 added. Then, 0.2 g potassium persulphate was added, and the whole was digested for 1 hr at 90°C (Standard Methods, Am. Public Health Assoc. 1976). During the oxidation with persulphate, oxidizable organic phosphorus, whether in soluble or particulate form, was released as inorganic phosphorus. With the total phosphorus in the inorganic molybdate reactive form, the digest was diluted and transferred to a separatory funnel, and the phosphorus content was measured as for orthophosphate.

C. Interface Particulate and Sediments

5. Total Kjeldahl Nitrogen. The Kjeldahl method, essentially a wet oxidation procedure, was used (Bremner 1965). The nitrogen of the sample was converted to NH_4^+ by digestion with concentrated H_2SO_4, containing Kjeldahl catalyst, which promoted the conversion. The NH_4^+ became NH_3 on distillation of the digest with strong alkali, and the NH_3 content of the sample was determined. Fifty mg dry sediment were placed in a digestion tube, to which 1 g Kjeldahl catalyst and 3 ml concemtrated H_2SO_4 were added. The sample was digested for 1.5 hours at 150°C, followed by 3.75 hours at 375°C. On cooling, the digest was diluted to 25 ml with distilled water, and transferred to a 100 ml distillation flask. Thirteen ml 10 N NaOH were added, and the sample steam was distilled for 3.5 minutes. The distillate was collected in a 50 ml volumetric flask containing 1 ml 0.1 N HCl. A color development was affected using Nessler's reagent, and the sample absorbance was determined spectrophotometrically at 402 nm against a reference cell of distilled water (Handbook of Chemistry and Physics 1977).

6. Total Phosphorus (tot. inorg. P + tot. org. P). One hundred mg of dry sample were digested following the Kjeldahl wet oxidation procedure. This was a modification by Kemp of the

Bremner method (Kemp personal communication, Bremner 1965). As was the case with the persulphate digestion of unfiltered mid-water, organic phosphorus was released in the inorganic molybdate reactive form and therefore was measurable as for orthophosphate.

Metal Analysis

Pretreatment of Samples

Before analysis, filtered water and unfiltered mid-water samples were defrosted at room temperature. Two ml redistilled conc. HNO_3 were added to each liter of water sample, producing a pH value ≤ 2. Thus acidified, the water samples were left to stand for a period not less than 48 hr prior to analysis for metal content.

Similar to the nutrient analyses, the problem of the highly varying amounts of sediment in unfiltered interface water samples prompted the filteration of interface water through a 0.45m Millepore membrane filter. The particulate material was subsequently treated as sediment, with metal content expressed on a dry weight basis.

Frozen sediment samples were broken into small pieces, and representative portions were oven-dried at 60°C, finely ground, and stored until analysis.

Analytical Methods

A. Sediment and Interface Particulate

A 2.5 g quantity of dry, ground sample was placed in a 125 ml Erlenmeyer flask containing 25 ml distilled water and 25 ml redistilled concentrated HNO_3. The digestion mixture was heated at 50°C for 6 hr in a hot water bath, with frequent shaking.

After digestion, the sample was cooled and filtered through a 0.45μ Millepore filter. The filtrate plus washings of the digest flask were transferred to a 100 ml volumetric flask and made up to volume. A reagent blank of 25 ml distilled water/25 ml conc. HNO_3 was carried throughout the digestion and final dilution (Villa, and Johnson 1974).

The concentrations of lead, zinc, and copper in the digests were measured by flame atomic absorption techniques using a Perkin Elmer 360 Atomic Absorption Spectrophotometer. The preparation of standard solutions and the procedure for the measurement of individual metal concentrations in the digests was followed from Analytical Methods for Atomic Absorption Spectrophotometry, Perkin Elmer (1976).

To keep the concentrations of iron and manganese within the linear range of absorbance, the digests required a further 1:25 dilution before measurement by flame techniques.

The concentrations of cadmium in the digests were low and were beyond the lower limits of detection by flame methods. Cadmium in the digests was measured by flameless atomic absorption using the method of standard additions.

Ten ml volumes of digest were spiked with 50-200 µl volumes of a 1 mg/l Cd standard solution, resulting in added concentrations of metal in the 5-20 µg/l range. To reduce matrix effects due to salinity, an equal volume of 5% NH_4NO_3 solution was injected into the graphite tube before injection of a 20 or 50 µl volume of straight digest or digest spiked to a known concentration with additional cadmium. Drying, charring, and atomizing times and temperatures were as recommended by Perkin Elmer for the HGA 2100 Graphite Furnace (Perkin Elmer 1976). Absorbance readings were recorded for each concentration of added cadmium, i.e., 0, 5, 10, 15 µg/l, etc. Absorbance readings were plotted against the added concentrations of metal. An extrapolation of the resulting straight line produced an intercept on the concentration axis equal to the concentration of metal in the digest solution.

B. Filtered Water and Unfiltered Mid-water

When the method of standard additions was applied to unfiltered mid-water samples, it was found that even after several additions of metal, particularly lead, there was no increase in absorbance readings. This effect was presumed to be due to absorbance of metal on the small amounts of particulate material present in the unfiltered water since no similar effect was found with filtered water samples. Further acidification to pH 1 with conc. HNO_3 prevented the absorbance of added metal on particulate material and kept added metal in solution.

At the same time, the method of Sholkovitz was used (Sholkovitz 1978). This method gave values of metal concentrations that were equivalent to and as reproducible as those obtained from graphite heated atomization, applying standard additions. Sholkovitz's method also had the advantage of being less laborious and less time-consuming.

One hundred ml volumes of filtered water and unfiltered mid-water samples, acidified to pH 1 for a minimum of 2 days, were evaporated until almost dry. The final volume was adjusted to 5 or 10 ml with 4 N HNO_3. This ten-or twentyfold increase allowed the concentrations of lead, copper, zinc, iron, and manganese to be determined by flame atomic absorption methods.

As was done with the sediment digest preparations, cadmium in the heat concentrated samples was measured by flameless atomic absorption, applying the method of standard additions.

Results

Precision of Estimates

Data variability arose from two sources: from that inherent in the laboratory methods, and from that due to field sampling techniques. The

laboratory precision of each analytical method was computed from sets of 3 or 5 aliquots taken from one field sample. The field sampling variation was estimated from sets of 3 or 5 field samples.

The variations in laboratory method and in field sampling were expressed as mean coefficients of variation. For each set, the ratio of the standard deviation to the mean value of a measurement was calculated as a percentage. The average of the percentages was taken as an estimation of variation. The variation values are presented where relevant as part of the tables of results.

Nutrients

Mid-Water Analyses

The results of the nutrient analyses for filtered and unfiltered mid-water samples are given in Tables 9 and 10, together with Figures 8-11.

The concentration of (NH_4^+)-N ranged from 2.4 - 103 µg/l at the dredged site, and from 0 - 33 µg/l at the control site. The concentration of total inorganic nitrogen was in the range 16 - 422 µg/l at the dredged site, and 10 - 380 µg/l at the control site. Orthophosphate values were very comparable at both sites, being in the range 12 - 44 µg/l. Levels of total phosphorus were also in the same range at both sites, namely 38 - 66 µg/l.

(NH_4^+)-N

Before dredging began, equal concentrations of nitrogen (22 µg/l) in the form of (NH_4^+)-N were found at the control and dredging sites. With the onset of dredging, the concentration rose immediately to 49 µg/l at the dredging site. Within 2-5 days the level had dropped substantially to 16 µg/l (Figure 8).

As time progressed, however, increased concentrations of (NH_4^+)-N in mid-water samples from the dredging site were not matched with increases at the control site. Application of the "student's" test to the data for 20, 50 and 259 days following dredging showed that the concentration of (NH_4^+)-N was significantly higher at the dredging site. Significance was at the 0.5 confidence level.

Total Inorganic Nitrogen

The concentration of total inorganic nitrogen (T.I.N.) increased in the mid-water samples following dredging. The level went from 44 µg/l to 111 µg/l. Over the first 20 days, values for T.I.N. at the dredged site were higher than the control values, significant at the .005 level. Through a period of 50 days, levels remained higher, at the .01 level. Thereafter, no significant difference was found between control and experimental samples (Figure 9).

Table 9. Nutrient concentrations in filtered mid-water

Date	Day	Dredged			Control		
		(NH_4^+)-N µg/l	T.I.N. µg/l	Ortho PO_4-µg/l	(NH_4^+)-N µg/l	T.I.N. µg/l	Ortho PO_4 µg/l
1978							
Sept 25	-1	22	44 (22)*	20			
Sept 26	0 0 hr	49	52 (3)	21	22	24 (2)*	12
Sept 26	0 1 hr	45	44 --	32			
Sept 27	1	33	58 (25)	30			
Sept 28	2	16	16 --	19			
Oct 1	5	28	111 (83)	30	0	14 (14)	27
Oct 16	20	43	46 (3)	42	21	20 --	43
Nov 15	50	2.4	18 (15.6)	28	7.2	10 (2.8)	31
1979							
June 12	259	103.2	422	44	33	380	43
Sampling variation		19.6%	9.3%	7.9%	38.9%	3.3%	9.3%
Laboratory method variation		3.5%	3.1%	10.0%	3.5%	3.1%	10.0%

*$(NO_3^- + NO_2^-)$ values.

Figure 8.
$[NH_4]$-N-Filtered Midwater Samples

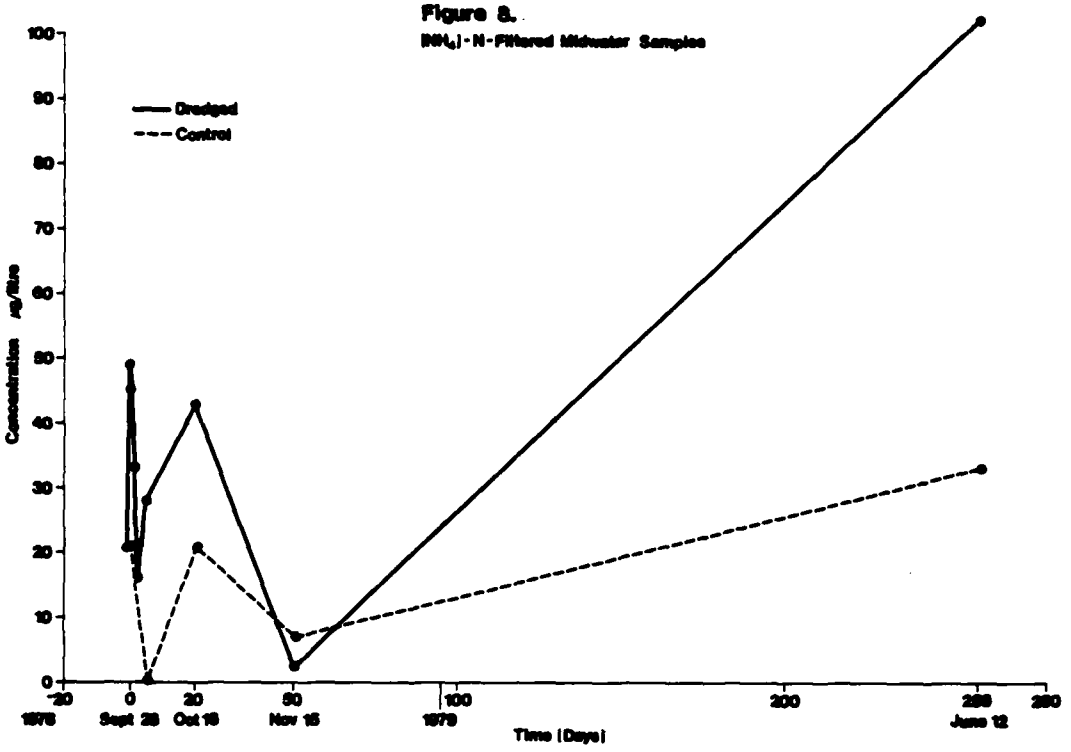


Figure 9.
Total Inorganic N-Filtered Midwater Samples

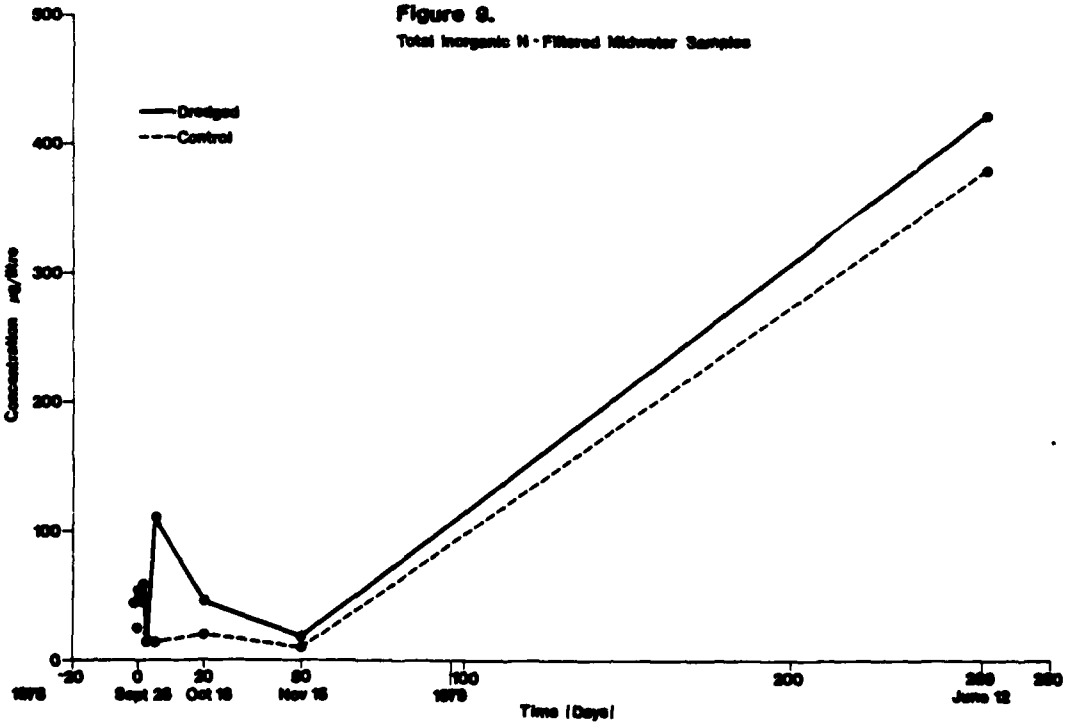


Figure 10.

Ortho PO_4 - Filtered Midwater Samples

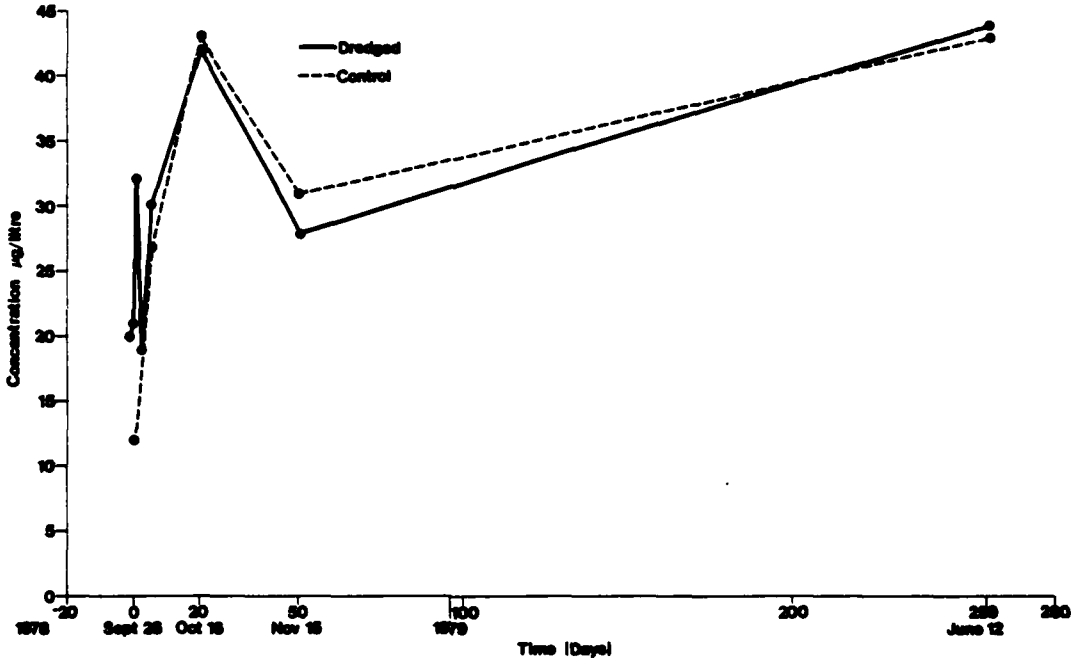
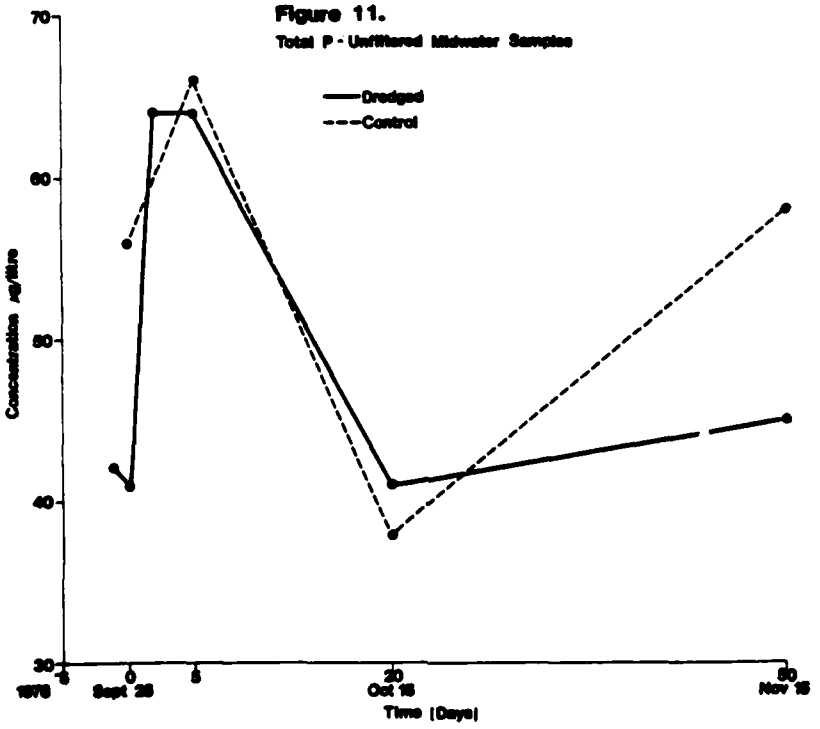


Figure 11.

Total P - Unfiltered Midwater Samples



Orthophosphate and Total Phosphorus

As can be seen from Figures 10 and 11, dredging did not have a major effect on the concentrations of orthophosphate (Ortho PO_4) or total phosphorus (Total P) in mid-water samples. Initially, there were small increases in these measurements. However, after 2-5 days, the concentrations in control and dredged samples were equivalent. It can be assumed that dredging did not alter the levels of either Ortho PO_4 or Total P.

Interface Water Analyses

The results of the nutrient analyses run on interface water samples are given in Table 11, also Figures 12-13. Concentrations of (NH_4^+)-N ranged from 103 to 348 µg/l at the site of dredging, and from 70 to 189 µg/l at the control station. T.I.N. values were 106 355 to mg/l at the dredging site and 74 to 279 µg/l at the control station. Orthophosphate ranged from 30 to 114 µg/l at the experimental site and from 29 to 48 µg/l in the control samples. Total phosphorus values lay between 0.029 and 0.055% dry weight of particulate material at the dredging site. Control values varied between 0.044 and 0.055%. Total Kjeldahl nitrogen ranged from 0.139 to 0.279% dry weight of the particulate material at the dredging site and from 0.188 to 0.343% in control samples.

(NH_4^+)-N

Dredging resulted in an increase of (NH_4^+)-N, in filtered interface water. The increase was maintained over 5 days following dredging, but thereafter the values decreased gradually. While the concentrations of (NH_4^+)-N remained ~40 µg/l above the control values, given the sampling variation, these higher values were not considered to be significant.

The results for (NH_4^+)-N in filtered interface water indicate that in response to dredging, concentrations increased initially up to three times their original value. After 5 days, the level dropped off and remained close to those obtained at the control site (Figure 12).

Total Inorganic Nitrogen

As with (NH_4^+)-N, dredging caused the concentration of total inorganic nitrogen to double in filtered interface water over the first 5 days following dredging. The increase had disappeared essentially after 20 days, although the levels of T.I.N. remained marginally above those of the control, by 20 to 30 µg/l. There was no indication, however, of statistically significant difference between control and experimental data (Figure 13).

Orthophosphate and Total Phosphorus

The results of the analyses of orthophosphate and total phosphorus in interface water were especially interesting. Dredging produced

Table 10. Concentrations of total phosphorus in unfiltered mid-water

Date	Day	Dredged Tot. P-µg/l	Control Tot. P-µg/l
1978			
Sept 25	-1	42	
Sept 26	0 0 hr	--	56
Sept 26	0 1 hr	41	
Sept 27	1	51	
Sept 28	2	64	
Oct 1	5	64	66
Oct 16	20	41	38
Nov 15	50	45	58
Sampling variation		12.7%	15.5%
Laboratory method variation		5.3%	5.3%

Table 11. Nutrient concentrations in interface water.

Date	Day	Dredged-filtered (NH_4^+)-N µg/l	Dredged-filtered T.I.N. µg/l		Dredged-filtered Ortho PO_4 µg/l	Dredged-particulate Tot. P % dry wt.	Dredged-particulate Tot. Kj-N % dry wt.	Control-filtered (NH_4^+)-N µg/l	Control-filtered T.I.N. µg/l		Dredged-particulate Ortho PO_4 µg/l	Dredged-particulate Tot. P % dry wt.	Dredged-particulate Tot. Kj-N % dry wt.
1978													
Sept. 25	-1	124	183	(59)*	30	.044	.279						
Sept. 26	0	---	---		--	---	---	70	100	(30)*	29	.053	.207
Sept. 27	1	348	355	(7)	114	.031	.159						
Sept. 28	2	202	260	(58)	80	.029	.139						
Oct. 1	5	235	327	(92)	58	.031	.263	189	279	(90)	43	.044	.188
Oct. 16	20	144	159	(15)	49	.055	.171	133	152	(19)	30	.059	.235
Nov. 15	50	103	106	(3)	40	.036	.275	75	74	--	48	.047	.343
Sampling variation		19.6%	9.3%		7.9%	9.6%	17.2%	38.9%	3.3%		9.3%	9.6%	17.2%
Laboratory method variation		3.5%	3.1%		10.0%	14.4%	4.5%	3.5%	3.1%		10.0%	14.4%	4.5%

*$(NO_3^- + NO_2^-)$-N values.

Figure 12.

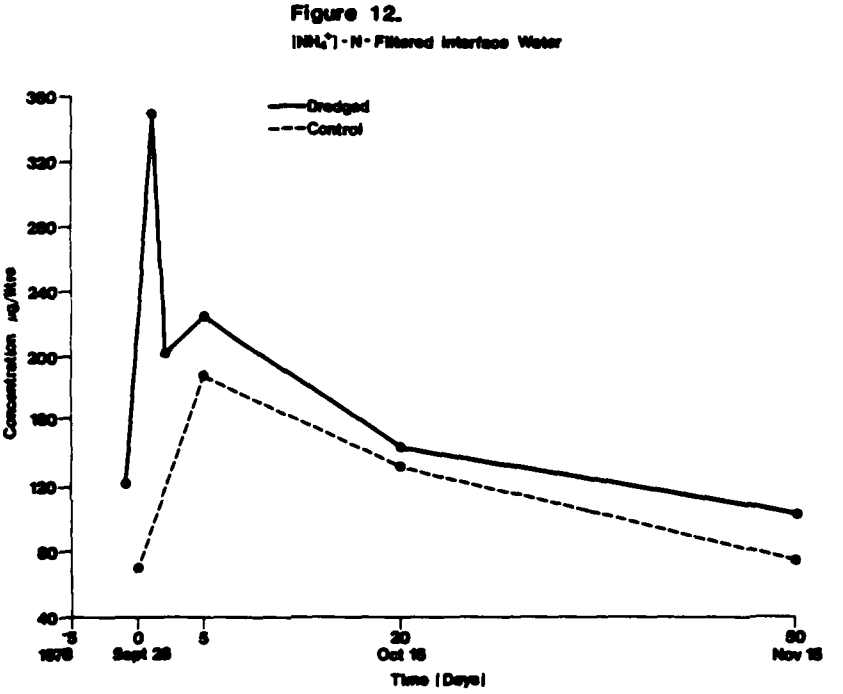
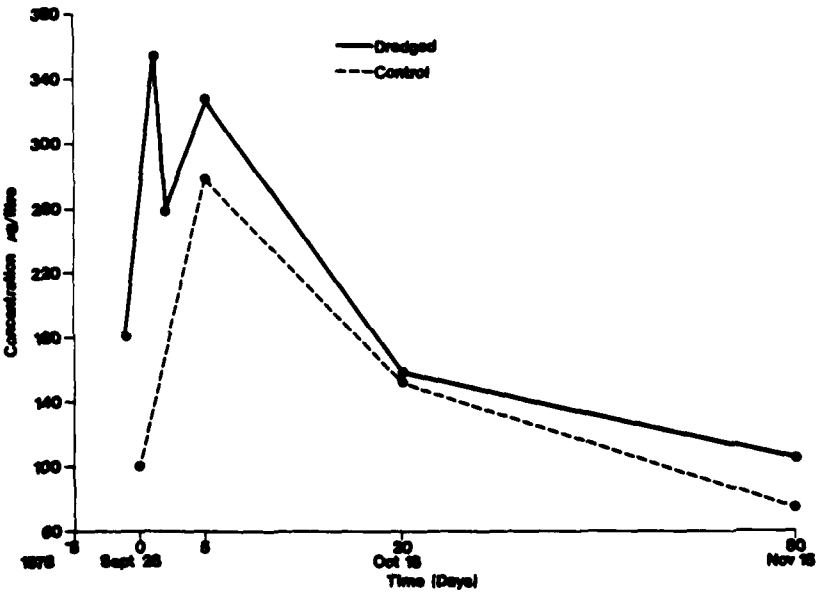


Figure 13.



concentrations of orthophosphate in filtered interface water four times higher than those obtained prior to dredging. This increase was significantly higher (.05 level) for a period of 20 days following dredging.

The levels of total phosphorus in the particulate material of unfiltered interface water decreased on dredging. Twenty days later the total phosphorus level had recovered, but levels lower than in the controls were maintained. For a 50-day period following dredging, the measurements of Total P at the dredged site were lower than those at the control and were significant at the .025 level (Figures 14 and 15).

Total Kjeldahl Nitrogen

Two days after dredging, the concentration of total Kjeldahl nitrogen in the particulate fraction of interface water had diminished to half of its original value. By day 5, the level had all but recovered. From day 5 to day 20 a second decrease occurred, which constrasted with increases in the control samples. By day 50 following dredging, increased levels were apparent again although still below control levels (Figure 16).

Sediment Analyses

The results of nutrient analyses for sediments are found in Table 12, and Figures 17 and 18. Total phosphorus varied between 0.034 and 0.055% dry weight in both control and dredged sediments. All values of total Kjeldahl nitrogen, whether control or experimental, were in the range of 0.017 to 0.303% dry weight of sediment.

Total Phosphorus

The total phosphorus content of the sediments at the dredging site dropped from 0.055 to 0.037% dry weight with the onset of dredging (dredge outlet sample). Twenty-four hours after dredging began, however, the total phosphorus concentration had returned to close to its original value. A second decrease was apparent through days 2 and 5. Thereafter, while the values remained marginally below those of the control samples, the pattern of behavior of total phosphorus in the dredged sediments did not deviate from that in the control sediments (Figure 17).

Total Kjeldahl Nitrogen

Of all the nutrients measured, dredging had the most dramatic effect on the total Kjeldahl nitrogen (Total Kj-N) content of the sediments. Before dredging began, the Total Kj-N level was 0.272% dry weight. The concentration in the sample taken from the dredge outlet was 0.017%, representing only ~5% of the initial nitrogen content of the undredged sediments. Then began a slow recovery. By day 50 after dredging, the nitrogen content of the sediments exposed to dredging was significantly lower than that of the control sediments at the 0.025 level. By days 205 and 233, nitrogen concentrations were still lower in the dredged sediments, at the 0.01 and 0.05 significance levels, respectively (Figure 18).

Figure 14.
Ortho PO_4 - Filtered Interface Water

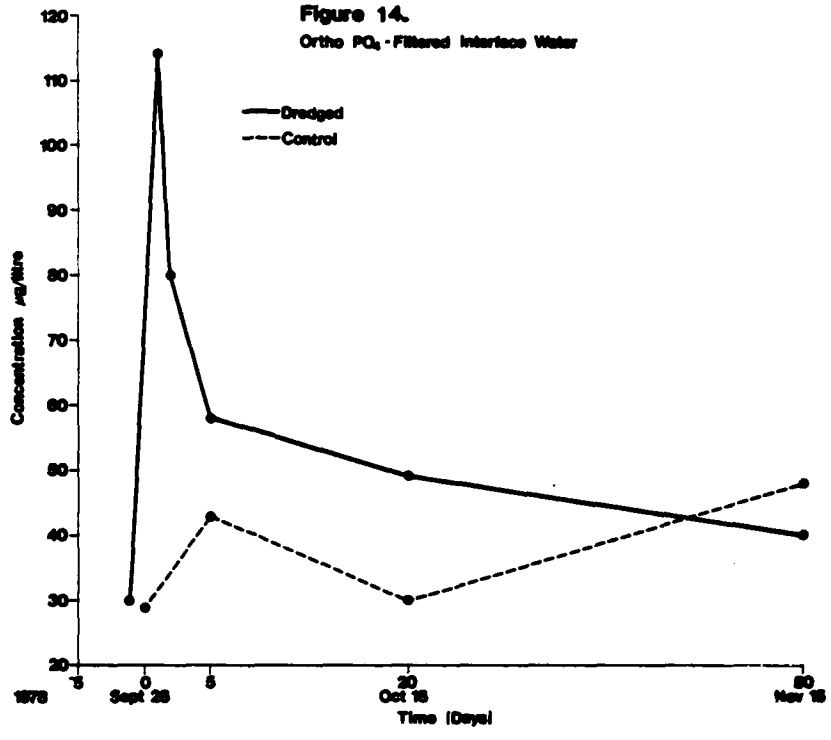


Figure 15.
Total P - Interface Particulate

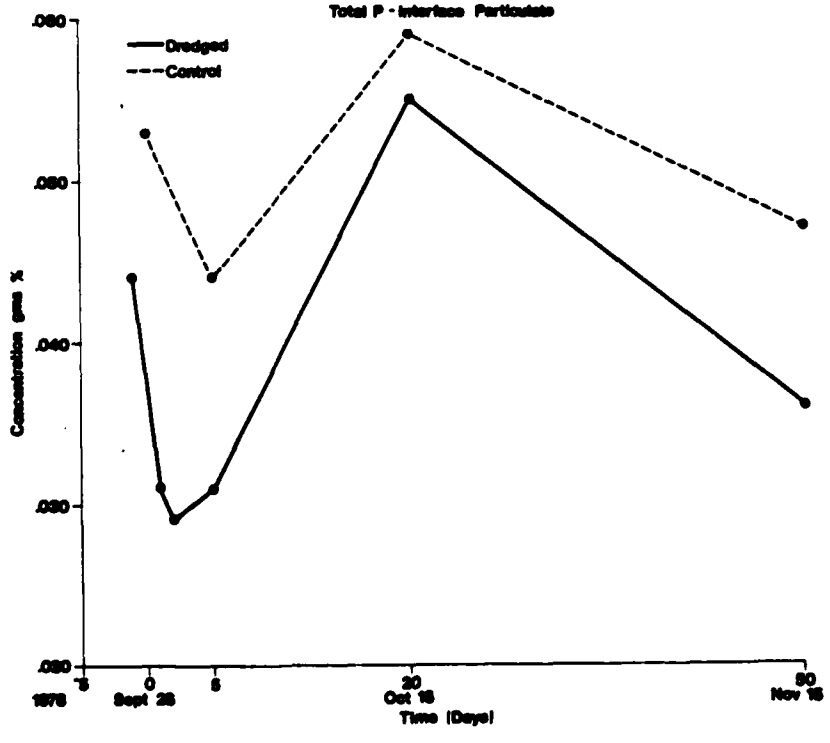


Table 12. Nutrient concentrations in sediments

Date	Day	Dredged		Control	
		Tot. P % dry wt.	Tot. Kj-N % dry wt.	Tot. P % dry wt.	Tot. Kj-N % dry wt.
1978					
Sept. 25	-1	.055	.272		
Sept. 26	0	*.037	.017	.040	.222
Sept. 27	1	.051	.058		
Sept. 28	2	.042	.055		
Oct. 1	5	.040	.130	.045	.178
Oct. 16	20	.054	.165	.055	.303
Nov. 15	50	.040	.131	.048	.204
1979					
April 19	205	.034	.159	.036	.200
May 17	233	.040	.106	.052	.089
Sampling variation		21%	26.7%	10.6%	17.8%
Laboratory method variation		14.4%	4.5%	14.4%	4.5%

*Sample taken from dredge outlet.

Figure 16.
Total Kjeldahl - N - Interface Particulate

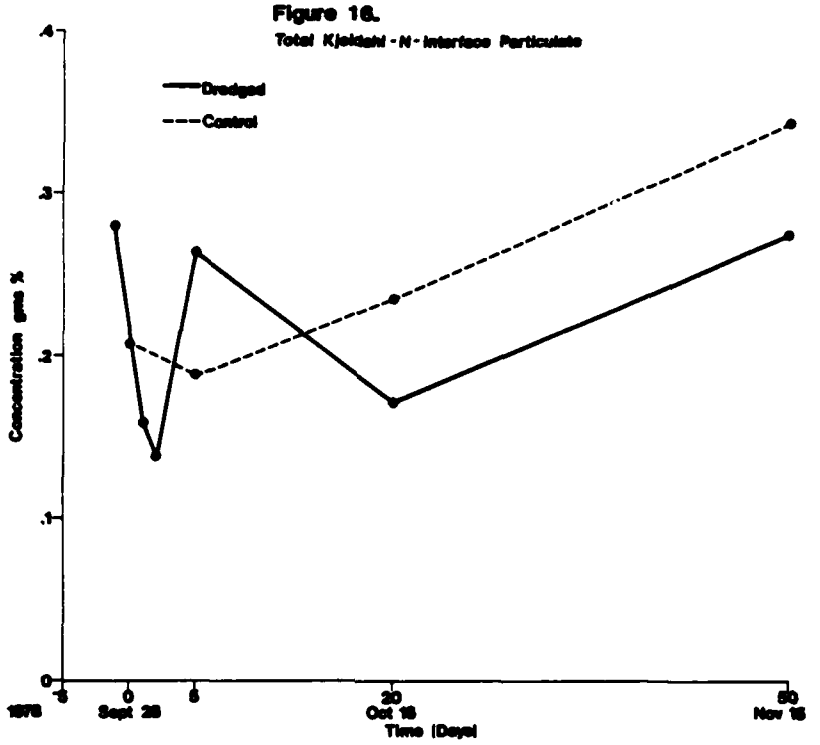
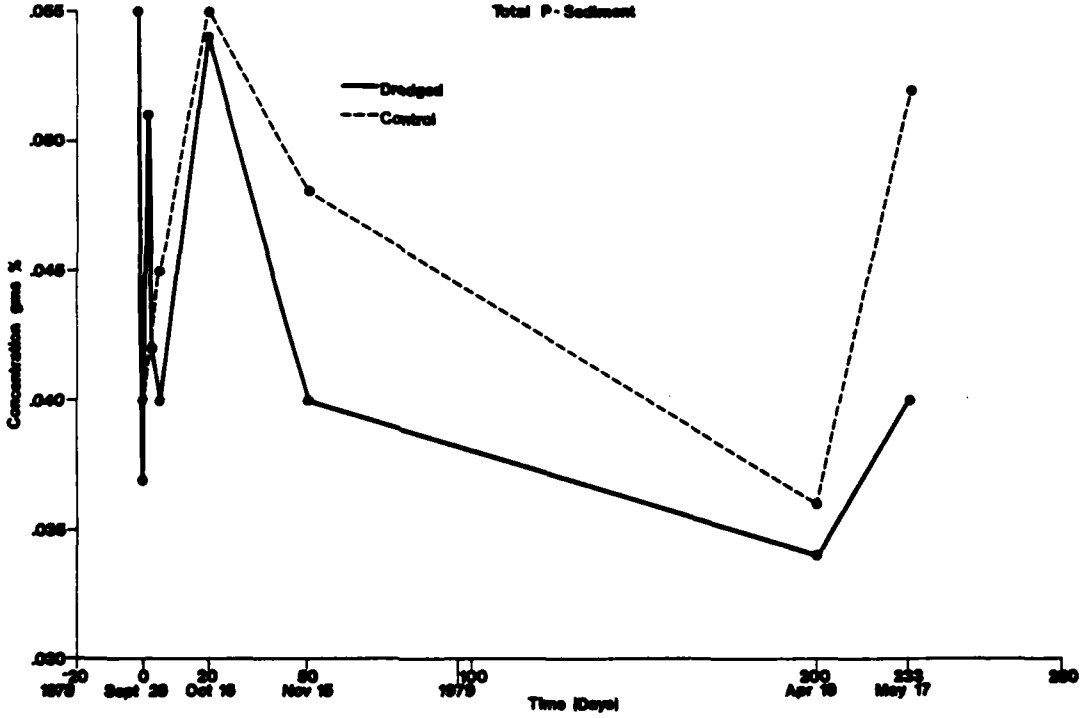


Figure 17.
Total P - Sediment



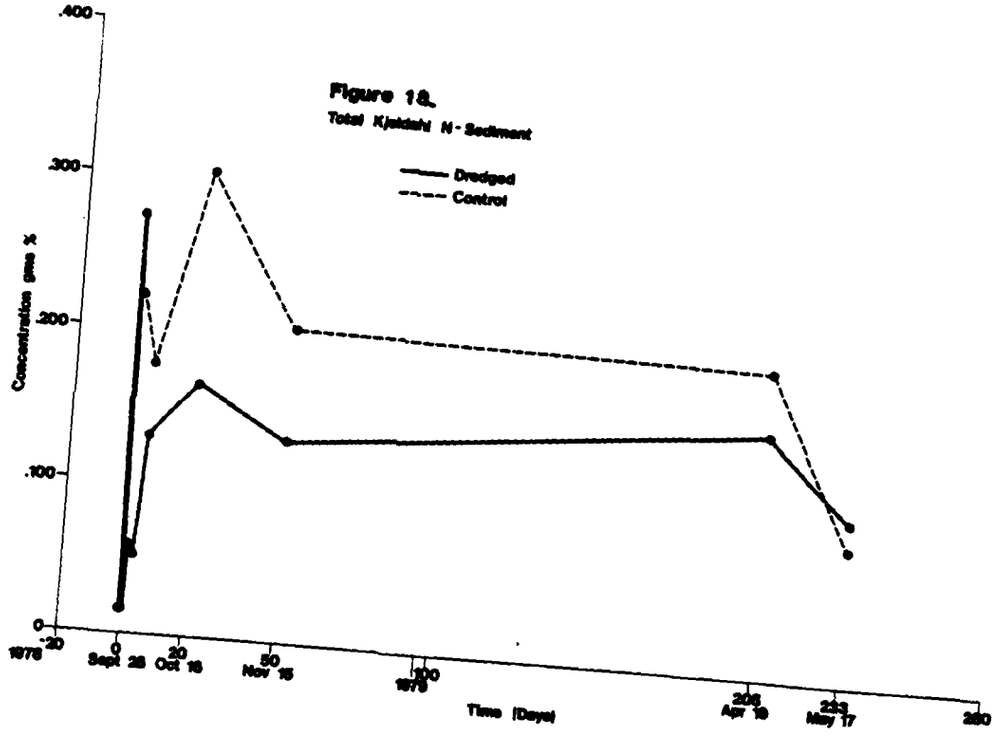


Figure 18.
Total Kjeldahl N - Sediment

Metals

In the figures of the heavy metal analyses in the sediments at the dredged site, concentration values of the interface particulate material have been substituted for sediment values for days 1 and 2 following dredging. These substitutions were necessary because immediately following dredging, it was impossible to distinguish the sediment-water interface in the fluid mud sediments at the dredged site.

The analysis of cadmium in the water samples was limited. The sensitivity of the instrumentation was 0.01 µg/ml, necessitating concentration of samples. Although the water samples were heat concentrated, analysis was at the limit of sensitivity. Often no cadmium was registered. Those levels recorded were: filtered and unfiltered mid-water (control and dredged) < 1.5 µg/l; filtered and unfiltered interface water (control and dredged) < 3.5 µg/l; sediments 1.0-3.0 µg/kg.

Iron

Iron concentrations in water and sediment samples are given in Table 13 and Figures 19-23.

Mid-Water Analyses

With the onset of dredging, the concentration of iron in filtered mid-water increased from 100 to 239 µg/l, but within one hour, it had decreased to 56 µg/l, a value that remained through day 1. By day 2, the concentration of iron in the filtered mid-water at the dredging site had increased to values slightly above those obtained before dredging began. By day 20, however, the iron content of filtered mid-water from the dredging site did not differ from control values (Figure 19).

When dredging commenced, the concentration of iron in unfiltered mid-water increased from 372 µg/l to 664 µg/l. The recovery to levels close to the original was slower than in filtered mid-water. By day 5, the concentration of iron had dropped to 266 µg/l, a value somewhat below that obtained before dredging began. Day 20 showed a second increase to 459 µg/l. By day 50, control and experimental values were very comparable.

For a 50-day period following dredging, the iron content of unfiltered mid-water was significantly higher (.05 confidence level) at the dredged site than at the control site (Figure 20).

Interface Water Analyses

Twenty-four hours after dredging, the iron content of filtered interface water had decreased to <50% of its initial value. By day 5, the original concentration level was restored and did not vary considerably thereafter (Figure 21).

Table 13. Concentrations of iron in water and sediment samples.

		IRON - DREDGED SAMPLES					IRON - CONTROL SAMPLES				
Date	Day	Filtered Mid-water µg/l	Unfiltered Mid-water µg/l	Filtered Interface µg/l	Interface Particulate g/Kg	Sediment g/Kg	Filtered Mid-water µg/l	Unfiltered Mid-water µg/l	Filtered Interface µg/l	Interface Particulate g/Kg	Sediment g/Kg
1978											
Sept 25	-1	100	372	100	15.600	17.200					
Sept 26	0 0 hr	239					100	313	276	15.894	17.500
Sept 26	0 1 hr	56	664								
Sept 27	1	58	598	40	22.293						
Sept 28	2	120	420	50	21.775						
Oct 1	5	100	266	100	18.800	16.500	66	234	100	16.300	18.500
Oct 16	20	69	459	84	19.662	20.654	66	348	160	19.576	18.670
Nov 15	50	80	134	81	15.575	13.096	63	125	116	13.096	14.172
1979											
Apr 19	205	---	---	---	---	13.402	---	---	---	---	15.594
Sampling variation			18.3%		5.0%	8.4%		18.3%		5.0%	1.9%

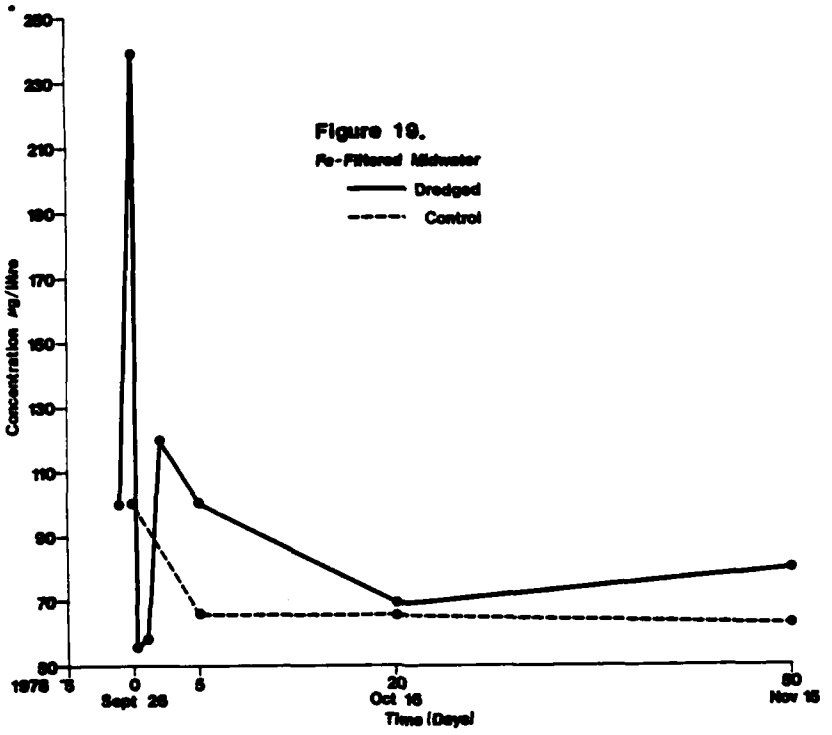


Figure 19.
Fe-Filtered Midwater
Dredged
Control
Concentration mg/litre
280
230
210
190
170
150
130
110
90
70
50
1976 -5
0
Sept 25
5
20
Oct 15
50
Nov 15
Time (Days)

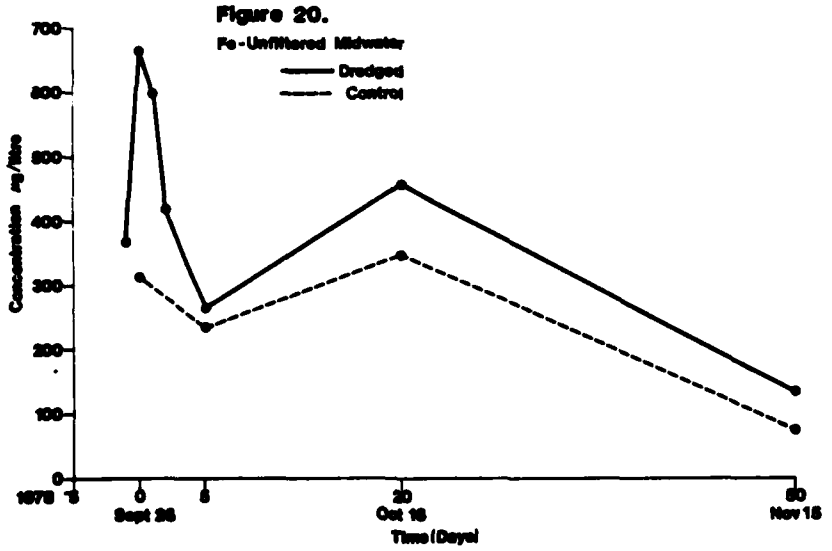


Figure 20.
Fe-Unfiltered Midwater
Dredged
Control
Concentration mg/litre
700
600
500
400
300
200
100
0
1976 -5
0
Sept 25
5
20
Oct 15
50
Nov 15
Time (Days)

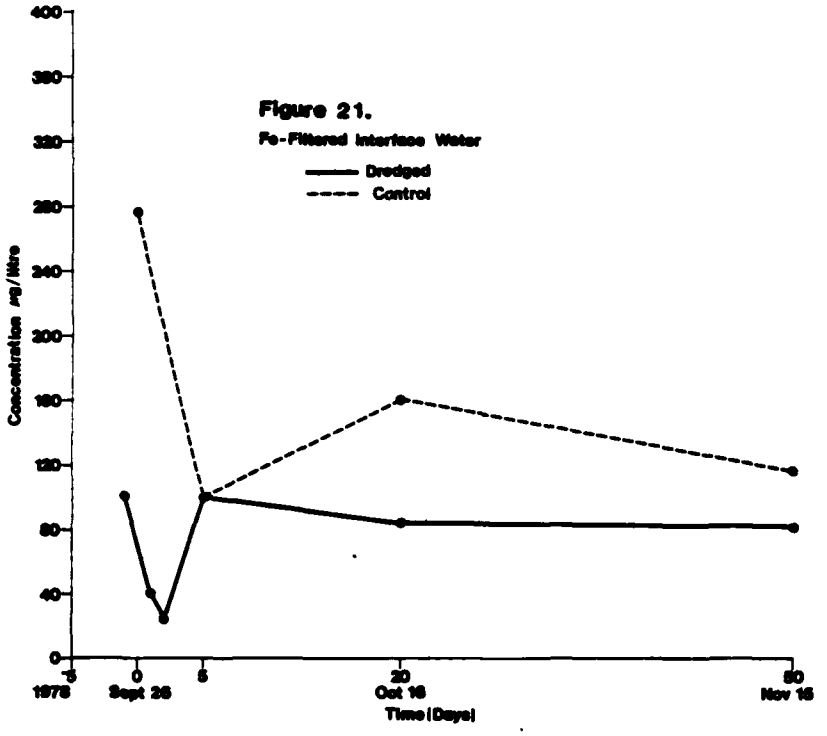


Figure 21.
Fe-Filtered Interface Water
Dredged
Control
Concentration µg/litre
400
380
360
340
320
300
280
260
240
220
200
180
160
140
120
100
80
60
40
20
0
-5
0
5
20
50
1978
Sept 26
Oct 16
Nov 15
Time (Days)

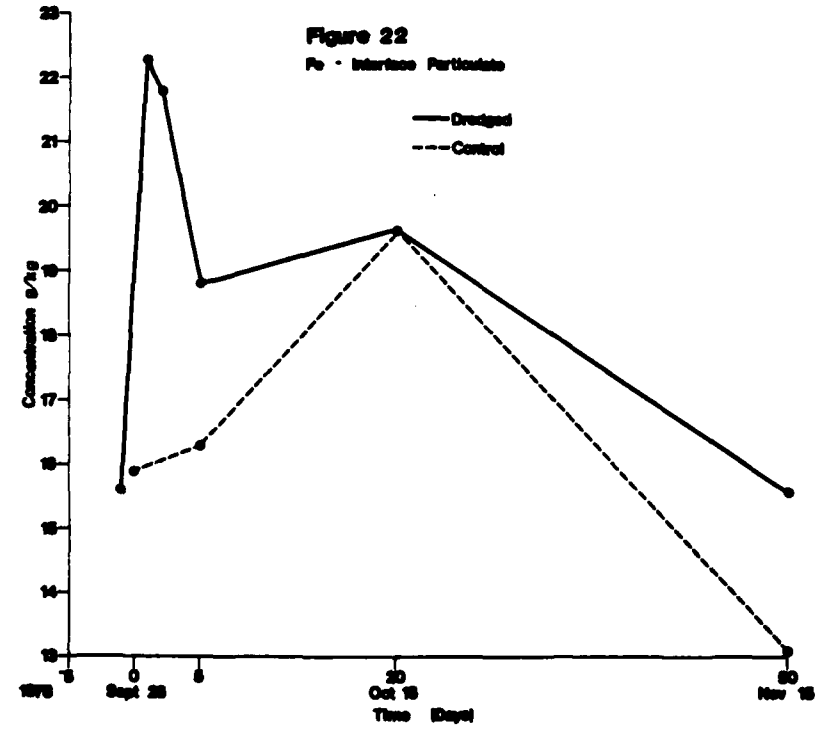


Figure 22
Fe - Interface Particulate
Dredged
Control
23
22
21
20
19
18
17
16
15
14
13
-5
0
5
20
50
1978
Sept 26
Oct 16
Nov 15
Time (Days)

The concentration of iron in interface particulate samples was 15.6 g/Kg dry weight before dredging. Values of 22.3 and 21.8 g/Kg were measured in interface particulate samples from the dredging site days 1 and 2 after dredging. Thereafter, the iron content decreased, and by day 50, levels had returned to original values (Figure 22).

Sediment Analyses

Substituting interface particulate for sediment values, it may be that the iron content of the sediments increased initially in response to dredging. By day 5 and subsequently, the concentrations of iron in dredged sediments did not differ from control values.

Manganese

The results of the manganese analyses in water and sediment samples are given in Table 14 and Figures 24-28.

Mid-Water Analyses

The onset of dredging produced an extremely large increase in the amount of manganese in filtered mid-water from the dredging site. A value of <10 µg/l initially increased to ~300 mg/l at the point of dredging. However, within an hour of dredging activity, the manganese concentration had dropped to 113 µg/l, and 24 hours later, the concentrations of manganese had returned to 5 µg/l (Figure 24).

As happened with iron, dredging caused significant changes in unfiltered mid-water samples. While some recovery took place through day 5, by days 20 and 50, manganese concentrations in unfiltered mid-water samples at the dredging site were still higher than control values at the .025 level of significance (Figure 25).

Interface Water Analyses

Any comparison between control and experimental results for filtered interface water was difficult. Before dredging began, there was substantially more manganese in filtered interface water at the control site. However, the results of the analyses of manganese concentrations in filtered interface water at the dredging site showed an initial drop in response to dredging. By days 5 and 20, the concentrations had largely recovered (Figure 26).

Unlike iron, the manganese content of the interface particulate material decreased following dredging. The initial concentration of 1.3 g/Kg decreased to 0.8 g/Kg by day 5. By day 20, the initial concentration values had been reestablished, and that situation remained through day 50, possibly indicating a long-term difference between the two sites (Figure 27).

Table 14. Concentrations of manganese in water and sediment samples.

		MANGANESE - DREDGED SAMPLES					MANGANESE - CONTROL SAMPLES				
Date	Day	Filtered Mid-water µg/l	Unfiltered Mid-water µg/l	Filtered Interface µg/l	Interface Particulate g/Kg	Sediment g/Kg	Filtered Mid-water µg/l	Unfiltered Mid-water µg/l	Filtered Interface µg/l	Interface Particulate g/Kg	Sediment g/kg
1978											
Sept 25	-1	2	61	1225	1.30	1.07					
Sept 26	0 0 hr	299			.806		10	48	1758	1.617	1.22
Sept 26	0 1 hr	113	123								
Sept 27	1	5	78	594	.862						
Sept 28	2	6	87	538	.895						
Oct 1	5	10	16	844	.830	.76	22	23	1530	1.610	.990
Oct 16	20	6	51	896	1.185	.938	13	37	1273	3.341	1.294
Nov 15	50	13	34	502	1.051	.889	25	31	965	3.180	1.752
1979											
Apr 19	205	---	---	---	---	.979	---	---	---	---	1.802
Sampling variation			17.3%		13.9%	33.6%		17.3%		13.9%	7.6%

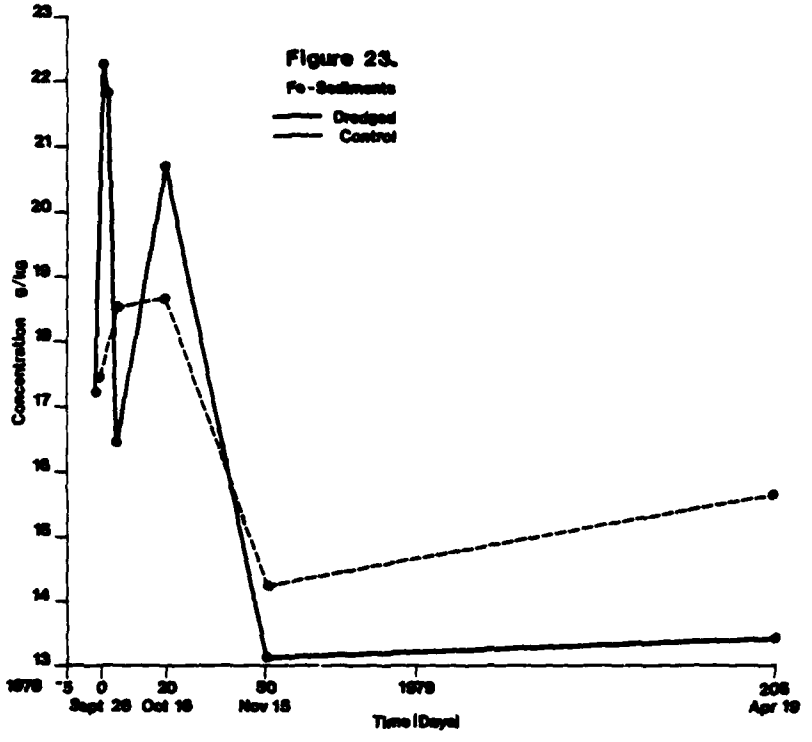


Figure 23.
Fe - Sediments
Dredged
Control
Concentration g/kg
23
22
21
20
17
14
13
-5
0
20
50
1979
205
Sept 28
Oct 18
Nov 18
Apr 19
Time (Days)

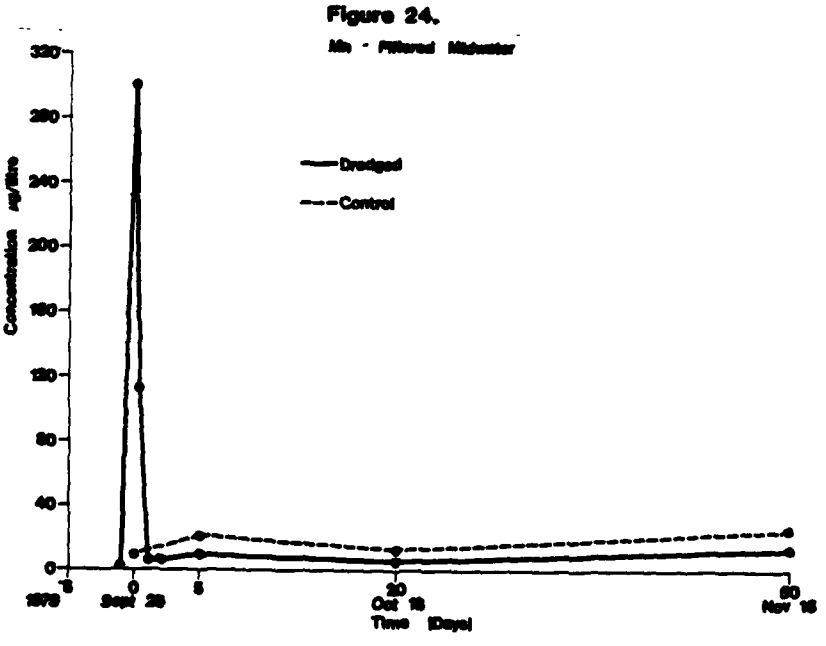


Figure 24.
Mn - Filtered Midwater
Dredged
Control
Concentration ug/litre
320
280
240
200
160
120
80
40
0
-5
0
5
20
50
1979
Sept 28
Oct 18
Nov 18
Time (Days)

Figure 25.

Mn - Unfiltered Midwater

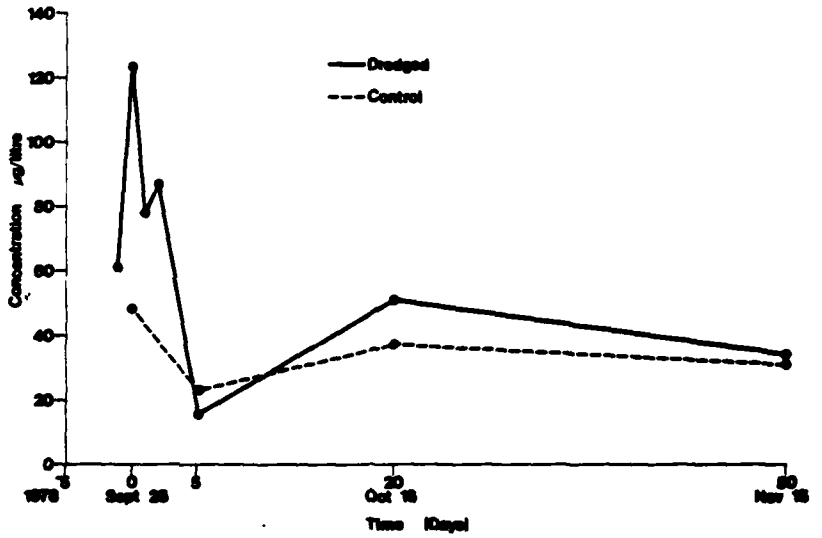


Figure 26.

Mn - Filtered Interface Water

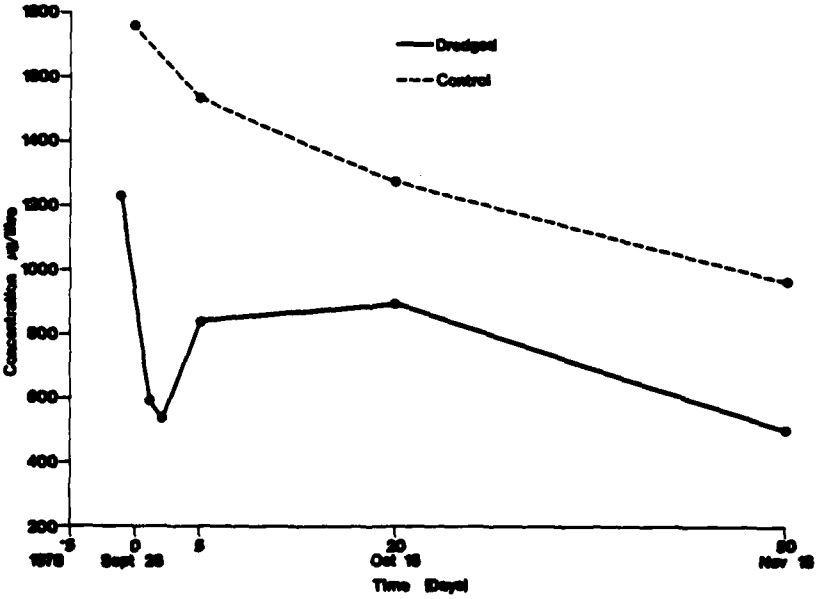


Figure 27.

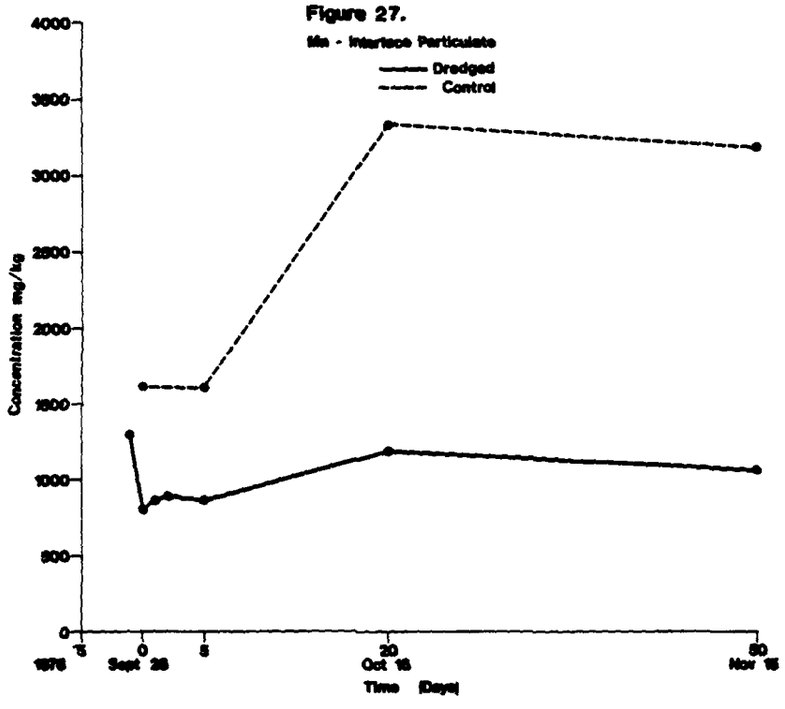
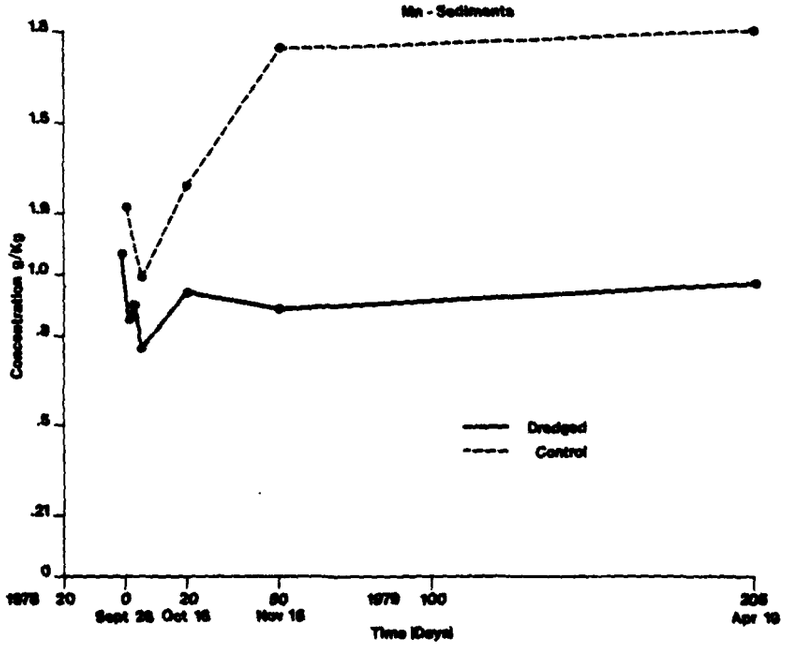


Figure 28.



Sediment Analyses

It would seem that some decrease sediment manganese occurred initially, at least through day 5. By day 20, the concentration of manganese in the dredged sediments was essentially equivalent to the before-dredging level but remained lower than control (Figure 28). This situation was similar to interface samples.

Lead

The results of the measurement of lead concentrations in water and sediment samples are given in Table 15 and Figures 29-33.

Mid-Water Analyses

The lead content of filtered mid-water increased from 25 to 110 µg/l with the onset of dredging. The higher values were maintained for a period of two days. By day 5 the level had dropped to 13 µg/l, somewhat below the initial value of 25 µg/l. A second increase to 50 µg/l was measured on day 20. The second higher value was still apparent on day 50 (Figure 29).

In response to dredging, the lead content of unfiltered mid-water increased from 36 µg/l before dredging to 66 µg/l 2 days after dredging. However, the concentration of lead in control samples varied greatly (9 - 95 µg/l), which reduced the significance of effects due to dredging (Figure 30).

Interface Water Analyses

Before dredging began, the concentrations of lead in filtered interface water at the dredging site were much below those at the control site, 9 µg/l and 27 µg/l, respectively. However, lead values in the experimental samples showed increases during a 5 day period subsequent to dredging, when control values exhibited decreases (Figure 31).

The concentration of lead in the interface particulate material increased from 32.3 to 47.4 mg/Kg by day 1 following dredging. It then decreased: values on days 2 and 5 were 35.6 mg/Kg and 21.5 mg/Kg, respectively. A second increase was measured on day 20, associated with an increase in control values. By day 50, experimental and control values were equivalent (Figure 32).

Sediment Analyses

Using substituted interface particulate values for sediment values, the effect of dredging on the lead content of sediments appears to have been minimal. Apart from a small initial increase on day 1 following dredging, the concentrations of lead in dredged and control sediments were similar (Figure 33).

Table 15. Concentrations of lead in water and sediment samples.

		LEAD - DREDGED SAMPLES					LEAD - CONTROL SAMPLES				
Date	Day	Filtered Mid-water µg/l	Unfiltered Mid-water µg/l	Filtered Interface µg/l	Interface Particulate mg/Kg	Sediment mg/Kg	Filtered Mid-water µg/l	Unfiltered Mid-water µg/l	Filtered Interface µg/l	Interface Particulate mg/Kg	Sediment mg/Kg
1978											
Sept 25	-1	25	36	9	32.3	35					
Sept 26	0 0 hr	110									
Sept 26	0 1 hr	84	44				8	37	27	34.3	24.3
Sept 27	1	87	57	40	47.4						
Sept 28	2	90	66	50	35.6						
Oct 1	5	13	31	--	21.5	30	22	9.5	15	27	29.6
Oct 16	20	50	68	84	38.8	46	73	95	57	47	52
Nov 15	50	57	34	57	30	32	29	37	70	30	30
1979											
Apr 19	205	---	---	---	---	29	---	---	---	---	29
Sampling variation			9.5%		7.5%	7.0%		9.5%		7.5%	5.7%

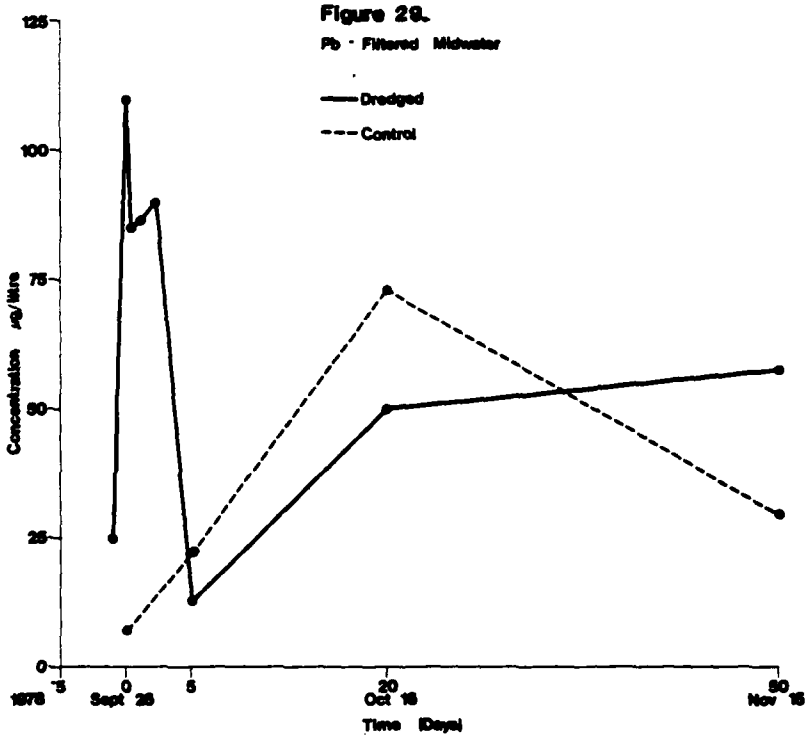


Figure 29.
Pb - Filtered Midwater
Dredged
Control
Concentration µg/litre
125
100
75
50
25
0
-5
0
5
20
50
1978
Sept 28
Oct 18
Nov 16
Time (Days)

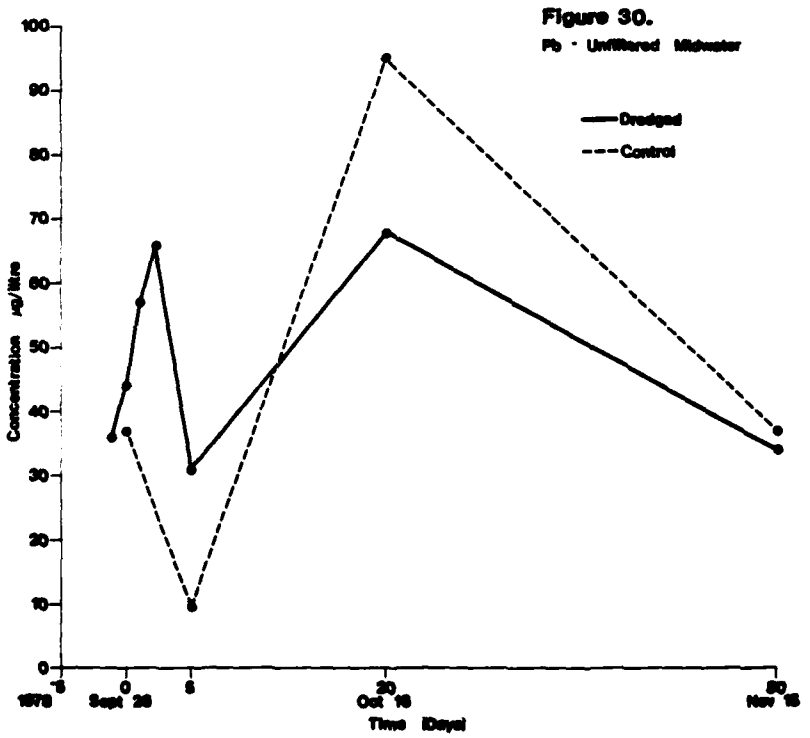


Figure 30.
Pb - Unfiltered Midwater
Dredged
Control
Concentration µg/litre
100
90
80
70
60
50
40
30
20
10
0
-5
0
5
20
50
1978
Sept 28
Oct 18
Nov 16
Time (Days)

Figure 31.

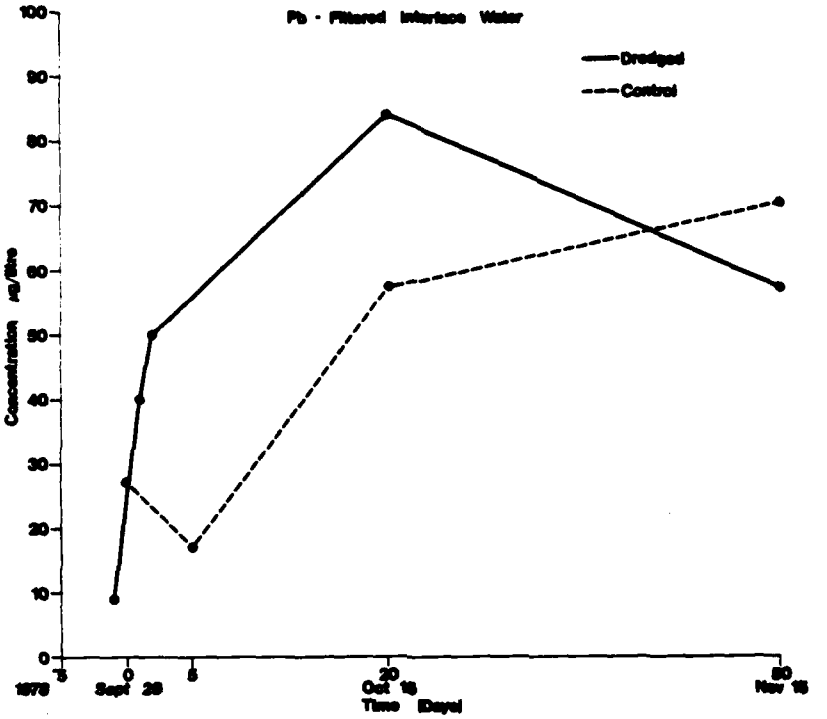
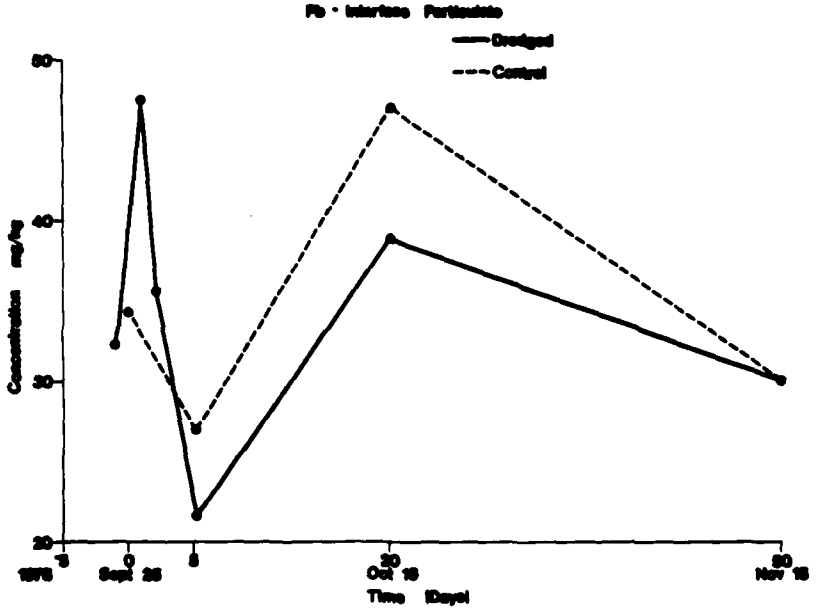


Figure 32.



Zinc

Zinc results are found in Table 16 and Figures 34-38. Dredging had no measurable effect on the concentrations of zinc in water and sediment samples.

Mid-Water Analyses

The isolated abnormally high value of 175 µg/l in filtered mid-water on day 50 following dredging must be noted and could be attributed to highly localized and temporary contamination present in the surface water at the dredging site. This value has been excluded from averaging.

In filtered mid-water, the average concentrations of zinc at the control and dredging sites were equivalent (34 µg/l) for a period of 50 days following dredging activity (Figure 34). Similarly, the average concentrations of zinc in unfiltered mid-water samples were essentially equivalent at the control and dredging site over the same period, being 23.7 and 22.8 µg/l, respectively (Figure 35).

Interface Water Analyses

The similarity between control and experimental values continued in interface water; average concentrations in filtered interface water were 36 µg/l at the dredged site over 50 days and 40.2 µg/l at the control site (Figure 36). For the same period, the average zinc content of the interface particulate material was 70.1 mg/Kg at the dredging site and 72.0 mg/Kg at the control site (Figure 37).

Sediment Analyses

When results for the zinc content of interface particulate material were substituted for sediment values, the average zinc content of sediments at the dredging site was 65.3 mg/Kg, compared with 70.4 at the control site (Figure 38).

Copper

The results of the analyses for copper in water and sediment samples are given in Table 17 and Figures 39-43.

As was the case with zinc, there was no indication of a dredging effect on the concentrations of copper in the water and sediments at the dredging site. Average concentrations of copper in control and experimental samples varied little, if at all. In filtered mid-water, the average control and experimental concentrations of copper over 50 days were 10.5 µg/l and 12.3 µg/l, respectively; in unfiltered mid-water, 12.7 µg/l and 11.0 µg/l; in filtered interface water, 13.0 µg/l and 13.7 µg/l; in interface particulate samples, 25.9 mg/Kg and 23.6 mg/Kg; in sediment samples, the values were 22.8 mg/Kg and 24.1 mg/Kg, respectively.

Table 16. Concentrations of zinc in water and sediment samples.

		ZINC - DREDGED SAMPLES					ZINC - CONTROL SAMPLES				
Date	Day	Filtered Mid-water µg/l	Unfiltered Mid-water µg/l	Filtered Interface µg/l	Interface Particulate mg/Kg	Sediment mg/kg	Filtered Mid-water µg/l	Unfiltered Mid-water µg/l	Filtered Interface µg/l	Interface Particulate mg/kg	Sediment mg/kg
1978											
Sept 25	-1	30	24	40	67	72.6					
Sept 26	0 0 hr	35.2			71		40	24	40	67	70
Sept 26	0 1 hr	34.4	14.7								
Sept 27	1	29	19.5	20	69						
Sept 28	2	30	22.4	30	72						
Oct 1	5	30	25	35	68	53	20	20	30	71	68
Oct 16	20	49.6	34	42	62	67.5	30	32	29	86	73
Nov 15	50	175	20	49	82	68	46	19	62	64	74
1979											
Apr 19	205	---	---	---	---	55	---	---	---	---	67
Sampling variation			19.3%		9.5%	10.3%		19.3%		9.5%	10.3%

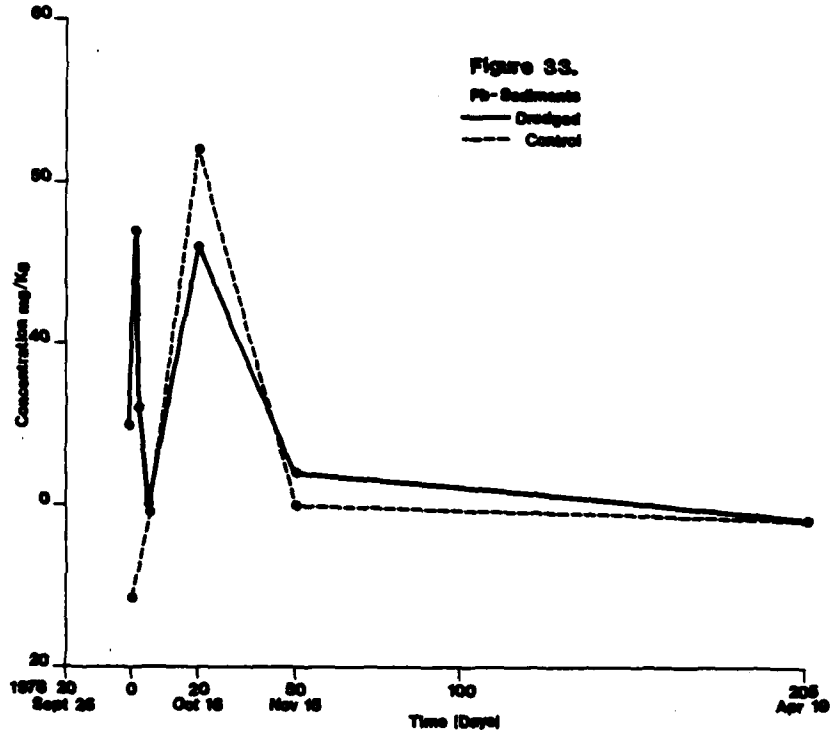


Figure 34.
Zn - Filtered Midwater

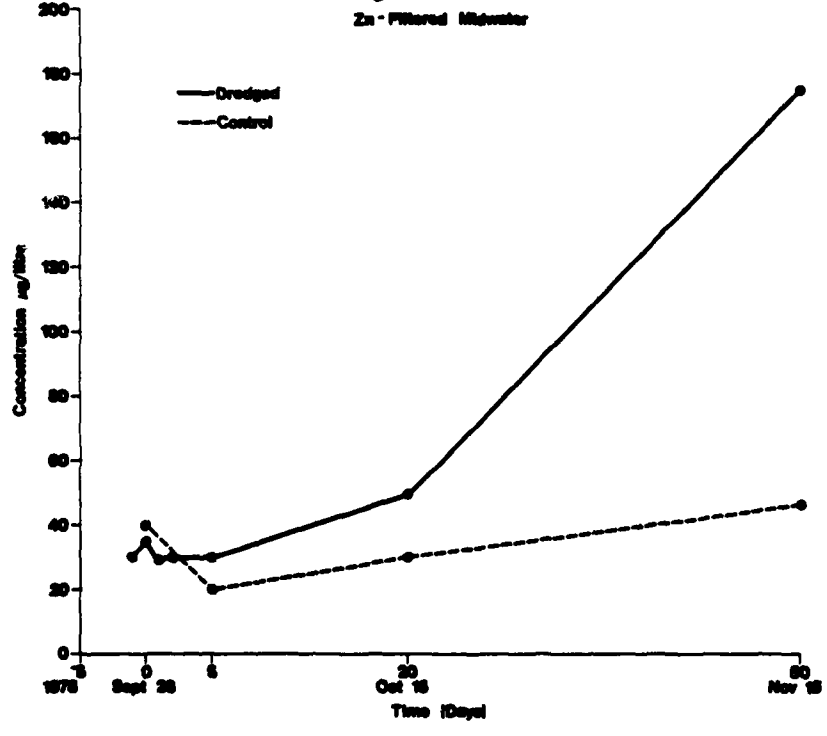


Figure 35.

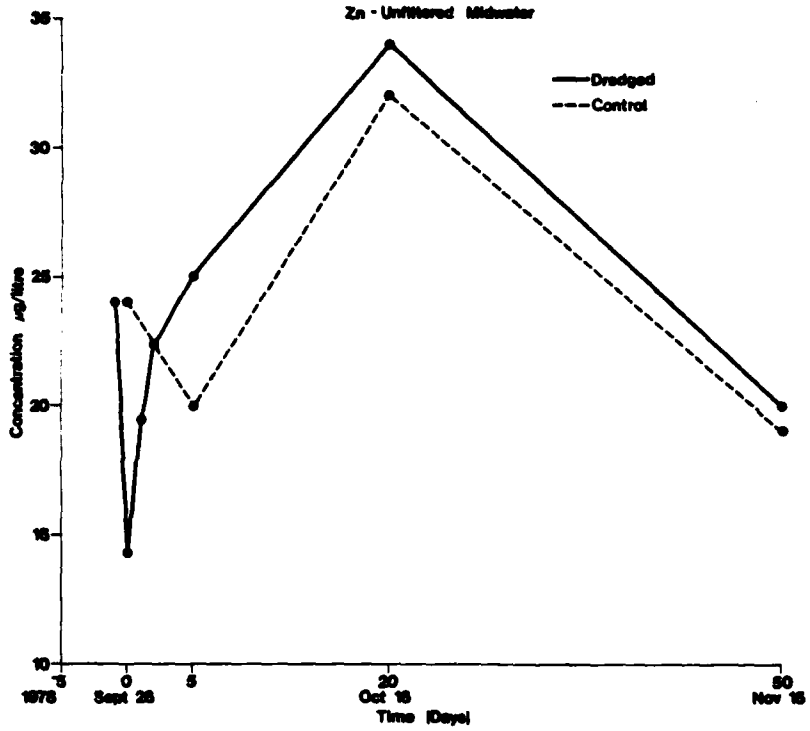


Figure 36.

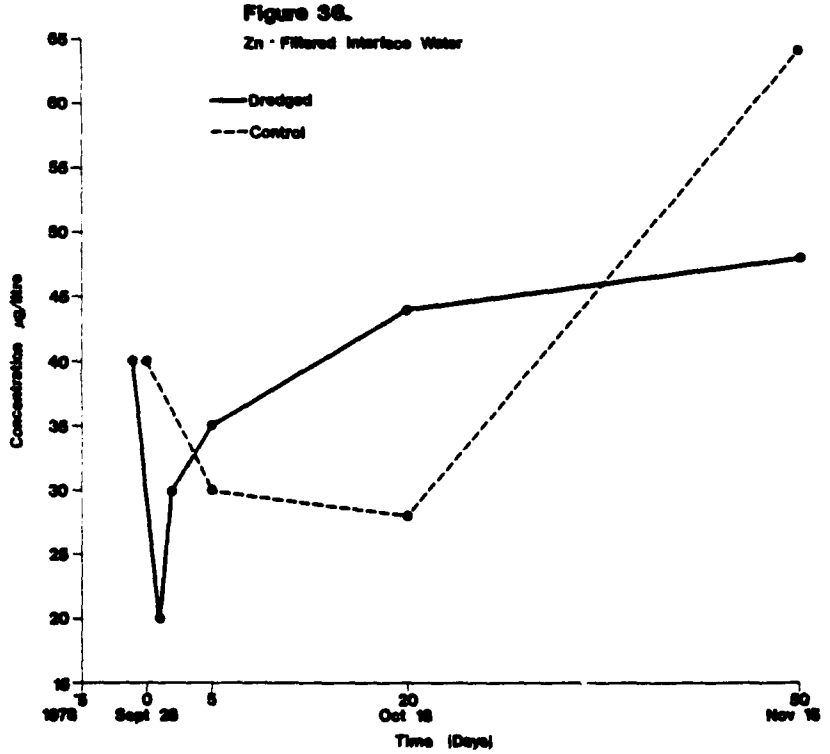


Figure 37.

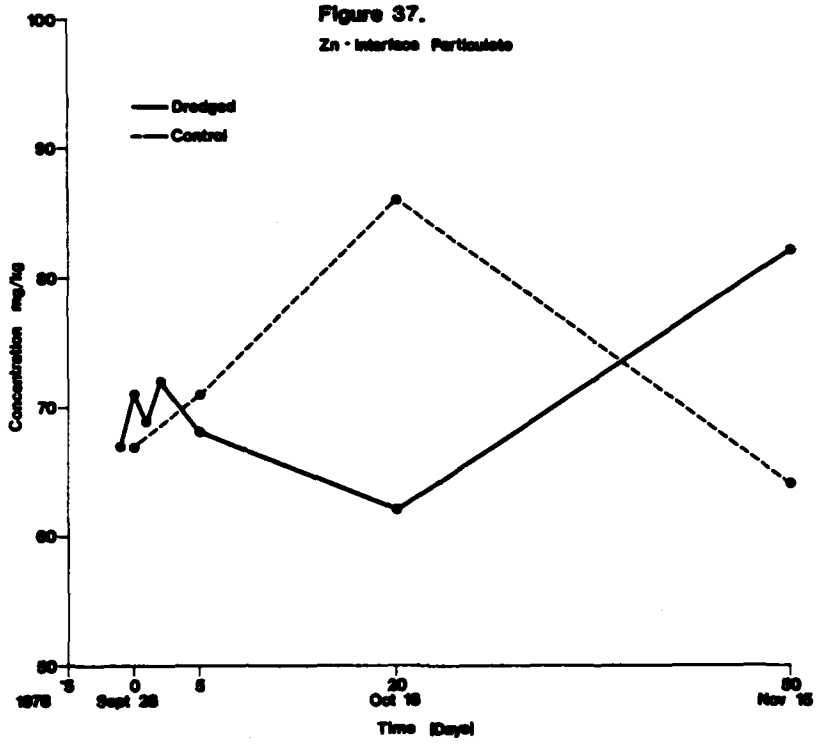


Figure 38.

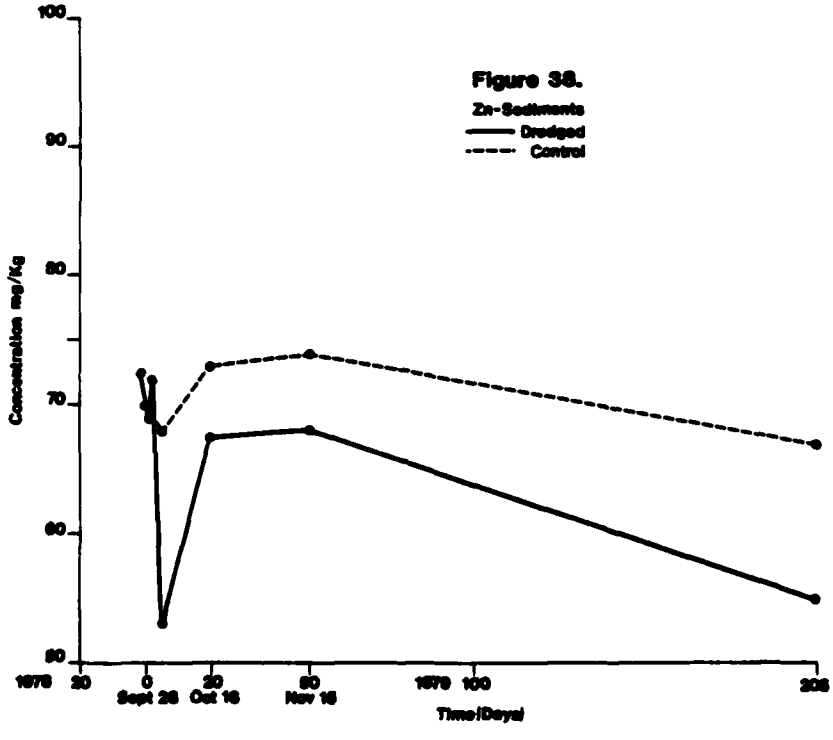


Table 17. Concentrations of copper in water and sediment samples.

		COPPER - DREDGED SAMPLES					COPPER - CONTROL SAMPLES				
Date	Day	Filtered Mid-water µg/l	Unfiltered Mid-water µg/l	Filtered Interface µg/l	Interface Particulate mg/Kg	Sediment mg/Kg	Filtered Mid-water µg/l	Unfiltered Mid-water µg/l	Filtered Interface µg/l	Interface Particulate mg/Kg	Sediment mg/Kg
1978 Sept 25	-1	8	9.7	11.5	21.6	23					
Sept 26	0 0 hr	12			21.2		10	9	15	22.6	19
Sept 26	0 1 hr	14	11								
Sept 27	1	12	7	10	21.6						
Sept 28	2	18	13.4	13	22.3						
Oct 1	5	11	14	20	21	17	15	18	10	22	19
Oct 16	20	12	13	14	18.3	18	10	18.9	14	20	19
Nov 15	50	12	9	7	39	43	7	5	9	39	34
1979 Apr 19	205	---	---	---	---	39	---	---	---	---	30.8
Sampling variation			13.4%		7.3%	10.5%		13.4%		7.3%	10.4%

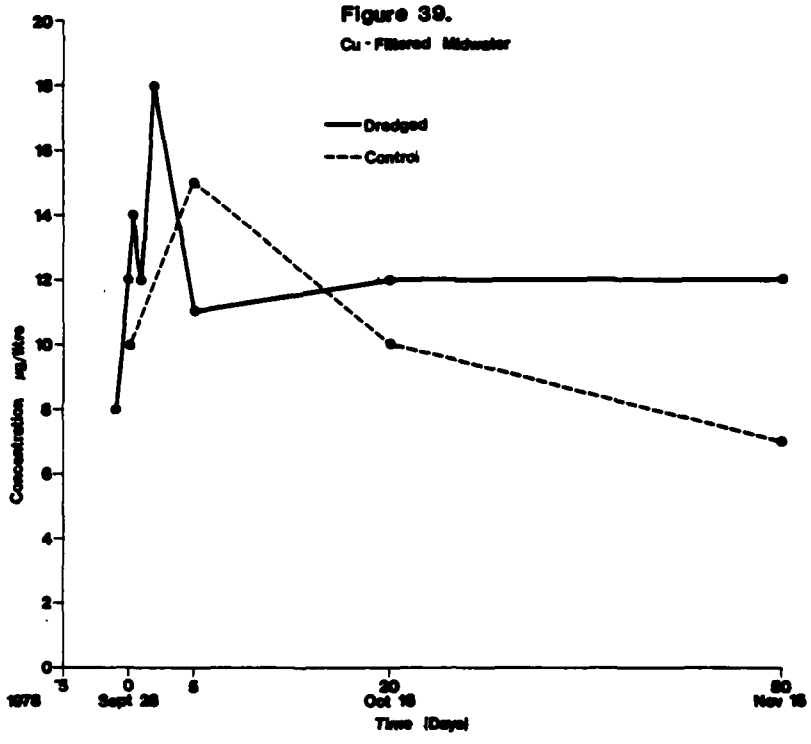


Figure 39.
Cu - Filtered Midwater
Dredged
Control
Concentration µg/litre
Time (Days)
1978
Sept 28
Oct 18
Nov 16

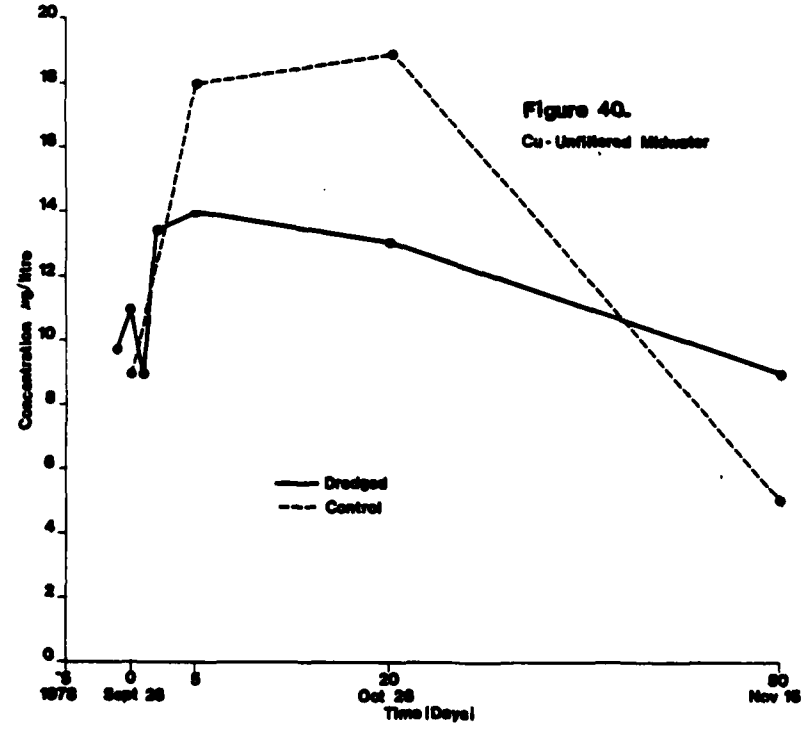


Figure 40.
Cu - Unfiltered Midwater
Dredged
Control
Concentration µg/litre
Time (Days)
1978
Sept 28
Oct 28
Nov 16

Figure 41.

Cu - Filtered Interface Water

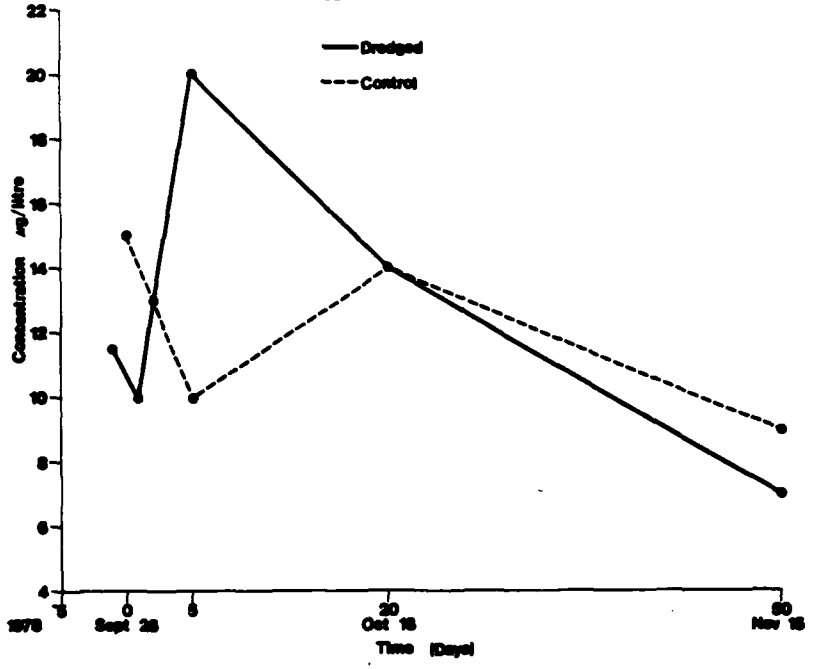
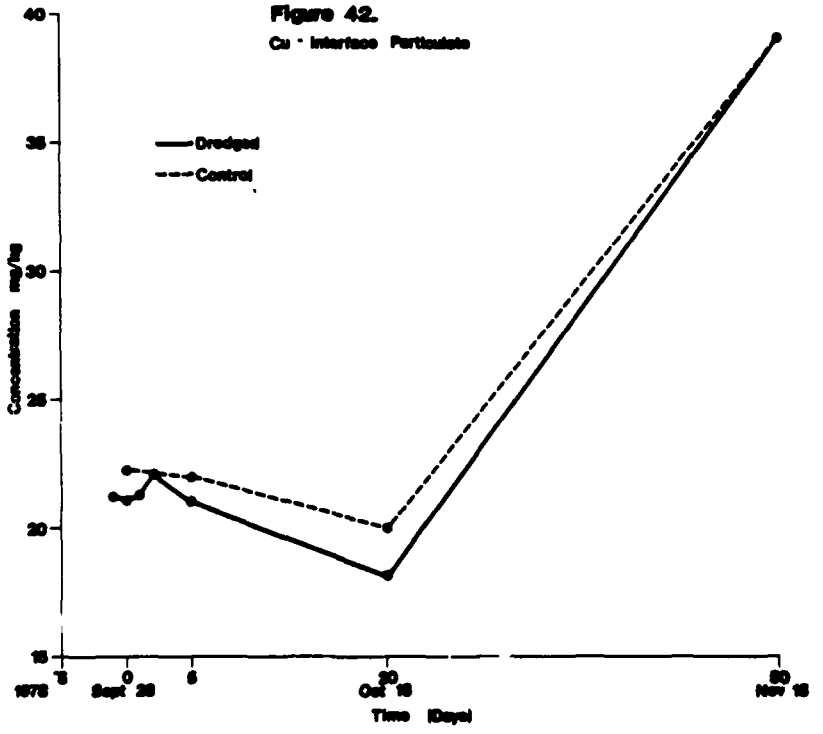


Figure 42.

Cu - Interface Particulate



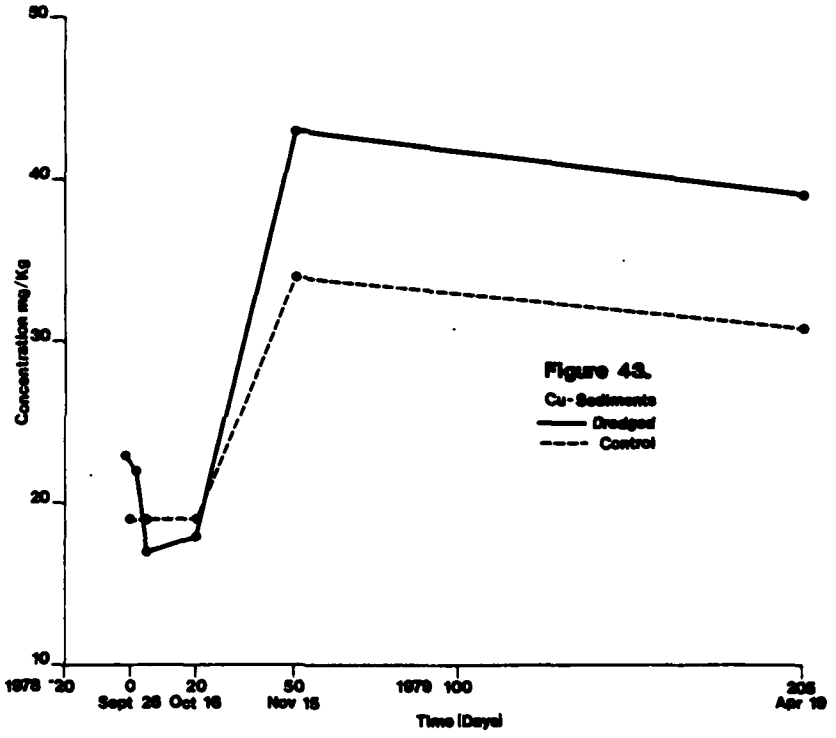

50
40
30
20
10
Concentration mg/kg
1978 -20
0
Sept 28
20
Oct 16
50
Nov 15
1979 100
205
Apr 19
Time (Days)
Cu-Sediments
Dredged
Control

Figure 43.

Discussion

Nutrients

It would seem more realistic to comment only on the behavior of (NH_4^+)-N in mid-water following dredging for a 50 day period because of the time gap between days 50 and 259.

Examination of Figure 8 shows that, apart from the immediate rise in concentration of (NH_4^+)-N in response to dredging, the variations in the concentration in mid-water at the dredged site were parallel to similar behavior at the control station, but at much lower levels. This would suggest that the sediments that were redeposited following dredging responded more easily than control sediments to environmental factors promoting the release of (NH_4^+)-N.

Day 20, October 16, 1978, together with October 14 and 15, experienced the highest winds (15-17 mph) of the entire 50-day period. The high wind values were associated with increases in the (NH_4^+)-N concentrations in dredged and control mid-water samples, but the dredged concentration were ~20 µg/l higher than the controls.

During hydraulic shell dredging, large volumes of anaerobic sediments from depth are gouged from the lake bottom and redeposited in the immediate vicinity as a dispersed slurry. The deposition of anaerobic sediments containing inorganic nitrogen in the reduced form as (NH_4^+)-N on the lake bottom can account for the higher (NH_4^+)-N concentrations in the mid-water at the dredged site.

Dredging alters the physical characteristics of the sediments. They are redeposited with a fluffy consistency and are much less consolidated than control sediments for a lengthy period following their deposition. As can be seen from other sections of this report, low bulk density dredged sediments were deposited over a wide area at the dredged station. The lack of consolidation and the decrease in bulk density of anaerobic sediments redeposited after dredging would facilitate the diffusion of (NH_4^+)-N through the interstitial water, making it available to the overlying mid-water.

The total inorganic nitrogen values, found to be significantly higher at the dredging site over 50 days, can be explained similarly. The oxidized nitrogen fraction $(NO_3^- + NO_2^-)$ of the total inorganic nitrogen content of mid-water did not alter in response to dredging over the period from day -1 to day 50. In samples from the dredging site, the $(NO_3^- + NO_2^-)$ component of the total inorganic nitrogen was 33.4%, the average value over 50 days. In control samples, the $(NO_3^- + NO_2^-)$ content averaged 34.1% of the total inorganic nitrogen. The significantly higher values of total inorganic nitrogen in mid-water for 50 days following dredging must then be related directly to the significant increases in reduced (NH_4^+) nitrogen.

The initial increases in orthophosphate and total phosphorus in mid-water samples for 2-5 days following dredging are considered to be

attributable to the interaction of the reduced sediments with the water column. Orthophosphate was released into solution from the reduced sediments in contact with the water column during disposal and resedimentation. The increased amounts of total phosphorus in mid-water samples over 2-5 days following dredging were associated with increased amounts of particulate material in resuspension in the water.

When the results obtained for (NH_4^+)-N concentrations in interface water are compared with mid-water values, trends are apparent. The (NH_4^+)-N concentrations in filtered interface water increased initially upon dredging and then decreased to levels comparable with those of the control, but always 10-20 µg/l higher. Again, it would seem that the changes in the physical characteristics of the dredged sediments facilitated diffusion of (NH_4^+)-N through the less consolidated anaerobic sediments, thus increasing the (NH_4^+)-N content of interface water.

As happened with mid-water, the $(NO_3^- + NO_2^-)$ fraction of total inorganic nitrogen in filtered interface was roughly equivalent in dredged and control samples, being 39% and 35%, respectively. It can be assumed that the higher values of total inorganic nitrogen found in dredged samples over 20 days following dredging were caused by increases in (NH_4^+)-N.

At the interface, significantly higher amounts of orthophosphate in filtered water were associated with decreases in the total phosphorus content of the particulate fraction. It would seem, therefore, that the removal of reduced sediments from depth by dredging and their superficial redeposition on the lake bottom promoted the release of soluble orthophosphate from the particulate fraction into the water. The decrease in the total phosphorus content of the particulate fraction was probably due in part to the removal of orthophosphate into solution. The deep sediments redistributed at the interface would be expected to contain less organic matter and hence less measurable total phosphorus.

Lower amounts of organic matter would also be associated with lower values of total Kjeldahl nitrogen in the particulate material of the interface water.

Samples of the slurry from the dredge outlet showed a 33% reduction in total phosphorus, reflecting the lower organic content of the deeper sediments redistributed at the interface by dredging activity.

The slurry from the dredge contained only 5% of the total Kjeldahl nitrogen of undredged sediments. There was, however, a gradual increase in the total Kjeldahl nitrogen of the redeposited sediments for a period of 9 months following dredging. The increase accompanies the presumed accumulation of organic plant and animal debris at the water/sediment interface.

Summary

Nutrients

It can be concluded that hydraulic shell dredging had significant effects on the nutrient chemistry of the waters and sediments of Lake Pontchartrain.

The response of nitrogen nutrients to dredging was demonstrated over a lengthy period of 9 months. The ammonia nitrogen content of shallow water maintained a significant increase over control samples. The total Kjeldahl nitrogen content of the dredge slurry was only 5% of the sediments before disturbance by dredging. Thereafter, the organic nitrogen content recovered with time.

For a period of 50 days following dredging, the balance of phosphorus exhibited changes. At the water/sediment interface, soluble inorganic phosphate in filtered water was present in increased amounts. The total phosphorus content of the particulate fraction of interface water showed a decrease.

It is thought also that the dredging activity altered the physical characteristics of the sediments, rendering them more sensitive to environmental stresses. The diffusion of (NH_4^+)-N through the redeposited sediments was facilitated. Also, the redeposited material that originated deep within the sediments had lower values of total phosphorus and total Kjeldahl nitrogen than did sediment samples undisturbed by dredging.

Metals

The analytical results indicate that copper and zinc concentrations in water and sediments are not affected by dredging. This non-effect is in keeping with Gorham and Swaine (1965) and May (1973). However, filtered interface water concentrations of lead, zinc, and copper appear to be elevated after dredging.

The observed variations in the levels of iron and manganese in unfiltered mid-water suggest that dredging activity has a substantial and lasting effect on the concentration of these two metals. That such effects are not significant statistically in interface water or sediments indicates that the iron and manganese concentrations are related to artificial suspension induced by dredging.

The initial, immediate, high values of iron and manganese in mid-water may be due to the suspension of reduced hydrous oxides of iron and manganese derived from the bottom sediments and redischarged into the water column. In unfiltered mid-water samples, the decrease that occurs over 5 days following the initial increase can be attributed to slow settlement of fine particulate material.

The fast recovery of iron and manganese concentrations in filtered mid-water within 24 hours following dredging is noteworthy. It is thought that the fast recovery from the immediate high values may be due to the chemical reactions of these metals in response to the temporary aeration of the dredging activity.

Many workers (Mortimer 1971, Lee 1975, Gotoh and Patrick 1972, Patrick et al. 1977) have found that the oxidation-reduction potential of surface waters and sediments regulates the concentrations of iron and manganese and their bioavailability in water-sediment systems. In anaerobic conditions, soluble-reduced forms of these metals predominate;

in the presence of dissolved oxygen, insoluble oxidized compounds of iron and manganese are formed.

The sudden change in oxidation-reduction conditions accompanying hydraulic shell dredging in Lake Pontchartrain means that reduced iron and manganese compounds may be converted to insoluble oxidized forms. The immediate increase and fast recovery of the concentrations of these two metals in filtered mid-water could be accounted for in these terms. Soluble, reduced iron and manganese oxides, which are immediately detectable in filtered water, will remain as fine suspensions of insoluble ferric and manganese precipitates, which settle more slowly, and are only measurable in unfiltered samples.

The secondary increases in the iron and manganese levels in unfiltered mid-water in association with 15-17 mph winds that occurred just before and during day 20 can be attributed to two factors. Wave-induced reoxygenation of the water column combines with the greater susceptibility of redeposited dredged sediments to entrainment by wave-induced forces. The concentration distribution through the 50 day sampling period, therefore, reflects both the initial effect of dredging and subsequent storm effects.

Conclusions

It can be concluded that hydraulic shell dredging had significant effects on the nutrient and heavy metal chemistry of the water and sediments of Lake Pontchartrain.

Dredging augmented significantly the amounts of ammonia nitrogen and soluble phosphate in the water column at the dredging site. Also, the total phosphorus content of interface particulate material decreased significantly in response to dredging, as did the total Kjeldahl nitrogen content of the dredged and redeposited sediments.

The iron content of unfiltered mid-water at the dredging site remained significantly higher than that of control samples (.05 level) for a 50-day period. It is suggested that these higher iron values are in part due to the artificial resuspension of sediments and to the slow settlement of very fine particulate material following dredging. The change in the physical properties of the redeposited sediments also increased their sensitivity to subsequent wind and wave effects.

The manganese content of unfiltered mid-water reacted to the dredging similarly to iron. Significantly higher values of manganese (.025 level) were in evidence at the dredged site over a period of 50 days.

No significant dredging effects were measured on the concentrations of lead, zinc, and copper.

EFFECTS OF SHELL DREDGING ON THE BENTHIC FAUNA OF LAKE PONTCHARTRAIN

Introduction

Since Lake Pontchartrain is quite shallow, averaging 4 m over most of the central portion of the lake, there is a much smaller ratio of water volume to bottom area than in other large lakes or in, for instance, the open gulf. Consequently, the benthic food web is considerably more important than in many other aquatic systems. Many of the dominant fish species such as the spot, Leiostomus xanthurus Lacepede, or the Atlantic croaker, Micropogonius undulatus (Linnaeus), feed on the benthic invertebrates (Levine 1980) as do many of the migrant waterfowl of the area, for instance the Lesser scaup, Aythya affinis (Eyton), and the Greater scaup, Aythya marila (Linnaeus). Thus any assessment of the impact of man's activities on the lake ecosystem should include a careful analysis of the impact on the benthos.

Methods

To quantitatively assess the impact of hydraulic shell dredging on the benthic fauna of Lake Pontchartrain, a series of replicate quantitative samples at the experimentally dredged site and the nearby control site were taken with a modified J&O box corer (Jonassen and Olausson 1966) 0.09 m^2 and subsampled with acrylic core tubes (4.9 cm^2) for meiofauna samples. Any box corer sample that appeared to be disturbed (such as when rough water would cause the boat to move enough to cause the corer to enter the sediment at an angle instead of straight down), that appeared to have entered the sediment too deeply so that organisms could be lost through the top, or that appeared to be leaking in any amount because of a piece of shell preventing full closure of the jaws of the device, were discarded. The contents of the box corer were sieved first through a one-half inch (1.27 cm) mesh, then one-eighth inch (0.32 cm) mesh to remove large shells and organisms, then through a 0.5 mm screen to retain the remainder of the macrofauna (animals > 500 µm). All fractions were preserved onboard ship in a buffered formalin solution with Rose Bengal stain. Meiofauna samples (animals between 500 and 44 µm) were preserved unsieved. All samples were returned to the laboratory for further sieving, enumeration, identification, and archiving. Organisms were identified to the lowest practical taxon, with certain forms being sent to specialists for confirmation, or further identification.

Meiofauna samples were sieved in the laboratory through a series of sieves. Animals retained on the 500 µm sieve were returned to the macrofauna sample from the box core from which the meiofauna core was removed. Animals retained on the 63 µm and 44 µm sieves were counted as meiofauna. As animals were identified, they were placed in appropriately labelled vials. All vials were recounted by another person, and the residue from which the animals had been removed was searched for animals that might have been overlooked. No sorter was retained whose accuracy had not improved to better than 95% by the third sample. After a sorter

had improved to better than 97%, only every third sample was checked. By recounting all vials of new sorters, and rechecking residues, total sorter error for the laboratory for meiofauna was held to under 3%. Sample vials have been archived for further use by taxonomic specialists if requested.

All macrofauna sorted to major taxa by student workers were identified to species by the more highly trained technical staff. Some specimens were sent to consulting taxonomic specialists for confirmation of identification. Some undescribed species are listed as such. Since macrofauna are rechecked as species identifications are made, the additional recounting to maintain laboratory efficiency in meiofauna sorting is not necessary for macrofauna sorting.

Samples at the experimentally dredged site were taken initially, before dredging, and on days 1, 2, 5, 20, 50, 205, 235, and 331. The control site was sampled initially and then on days 5, 20, 50, 205, 235, and 331. After the first year, samples have been taken at the two study sites quarterly. A total of 104 meiofauna and 78 macrofauna samples have been completed and analyzed to this date.

Standard statistical analyses were performed using standard programs; certain special programs for classification and ordination of benthic data were provided by Dr. Stephen A. Bloom of the University of Florida (Bloom et al. 1977). No transformations of the data were used. To prevent the classification from being unduly biased by large numbers, (dominant species) or small numbers (rare species), a simultaneous double standardization was used (Boesch 1973). The Canberra metric was used as a measure of dissimilarity, with a flexible sorting strategy ($\beta = -0.25$), to produce the clusters for the analysis (Boesch 1977).

Results

Abundance of Meiofauna

Examination of the meiofauna community (Table 18, Table B1, and Figure 44) shows first that there were fewer animals at the experimental site than at the control site originally, before the experimental dredging was done. The numbers of true meiofauna, such as nematodes and copepods, were not significantly different. The major differences occurred in forms such as ostracods and bivalves that could settle out of the plankton. This patchiness in species settling is a common phenomenon, accounting for a great deal of the natural fluctuations in both the macrofaunal and meiofaunal components of soft-bottom benthic communities (Eagle 1975). The day after dredging, the abundance at the dredging site had dropped to approximately half its previous value. This is accounted for by noting that although very few nematodes came through the dredge outlet in recognizable form, dredging at the experimental site in a typical "soft-bottom," or light silt-clay, left a series of troughs and ridges that, through slumping, coalesced, and thus redistributed the animals remaining in the ridge of undredged areas of the experimental site over the entire area. The second day after dredging, the abundance of total

Table 18. Meiofauna, major components

	Experimental		Control	
	N/10 cm^2	SE	N/10 cm^2	SE
	---------- Initial ----------			
Nematodes	228.7	20.9	288.8	21.4
Copepods	68.8	10.9	94.2	7.0
Others	242.9		279.6	
TOTAL	540.4	10.9	662.6	43.4
	---------- Day 1 ----------			
Nematodes	104.9	49.4		
Copepods	33.6	6.7		
Others	124.8			
TOTAL	263.3	47.1		
	---------- Day 2 ----------			
Nematodes	167.6	41.8		
Copepods	19.4	4.6		
Others	252.0			
TOTAL	439.0	81.8		
	---------- Day 5 ----------			
Nematodes	151.3	12.9	196.1	55.2
Copepods	13.2	3.6	26.5	9.0
OTHERS	165.0		205.2	
TOTAL	329.5	30.0	427.8	119.9
	---------- Day 20 ----------			
Nematodes	47.4	12.9	173.2	31.6
Copepods	26.5	8.6	34.6	6.6
Others	78.9		198.1	
TOTAL	152.8	42.4	405.9	71.2
	---------- Day 50 ----------			
Nematodes	234.3	20.3	457.4	77.6
Copepods	37.2	5.5	68.3	6.7
Other	276.5		278.5	
TOTAL	548.0	38.5	803.7	161.0
	---------- Day 205 ----------			
Nematodes	264.9	52.8	592.6	44.0
Copepods	108.0	25.2	58.8	13.9
Others	246.4		167.3	
TOTAL	609.6	76.9	720.7	131.5

Table 18. (Continued)

	Experimental		Control	
	N/10 cm^2	SE	N/10 cm^2	SE
	---------- Day 235 ----------			
Nematodes	324.4	60.2	712.0	43.7
Copepods	40.7	7.3	81.5	5.3
Others	152.3		301.5	
TOTAL	517.4	83.4	1,094.1	41.3
	---------- Aug 79 ----------			
Nematodes	202.7	30.4	238.0	94.9
Copepods	151.8	21.0	56.9	13.0
Others	198.1		626.4	
TOTAL	552.6	53.8	921.3	85.4
	---------- Nov 79 ----------			
Nematodes	352.9	36.3	266.4	37.2
Copepods	83.5	8.2	108.5	20.6
Others	215.5		131.9	
TOTAL	651.9	87.7	506.8	73.6
	---------- Feb 80 ----------			
Nematodes	480.3	81.5	648.4	126.0
Copepods	174.2	61.7	112.1	9.7
Others	169.0		331.4	
TOTAL	823.5	64.1	1,091.9	115.6
	---------- May 80 ----------			
Nematodes	644.3	86.0	857.7	129.7
Copepods	65.2	19.9	35.1	7.7
Others	125.8		192.0	
TOTAL	835.3	85.9	1,084.8	95.8
	---------- Aug 80 ----------			
Nematodes	529.2	115.9	377.4	20.9
Copepods	70.3	19.1	58.1	6.5
Others	442.0		381.5	
TOTAL	1,041.5	173.4	817.0	52.8
	---------- Dec 80 ----------			
Nematodes	1,242.2	163.9	1,278.3	163.1
Copepods	94.2	10.9	60.6	8.5
Others	357.0		284.2	
TOTAL	1,693.4	197.1	1,623.1	199.0

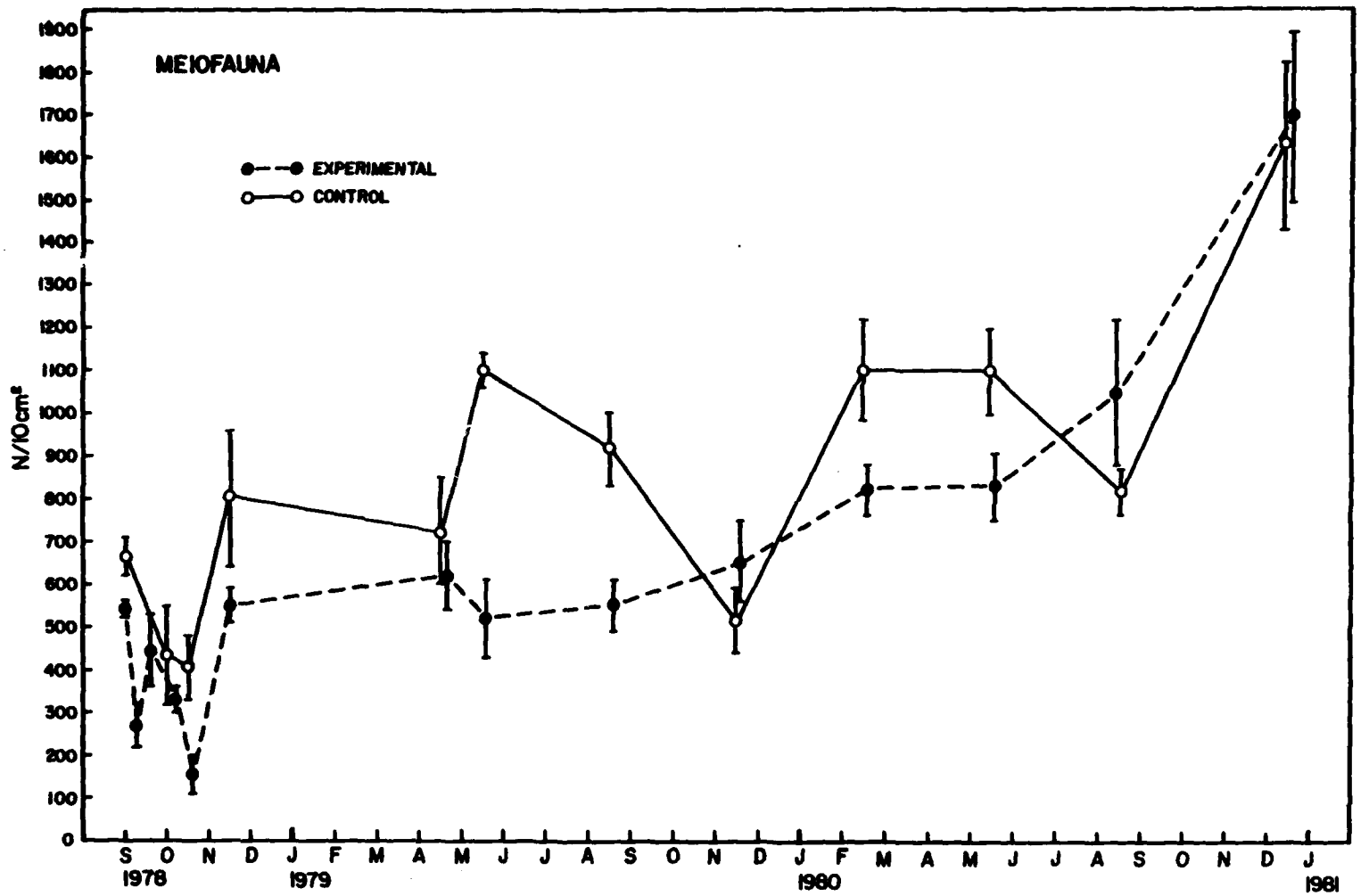


Figure 44. Meiofauna abundance at the control site and experimental site. Circles indicate the mean of 4 samples; bars represent the associated standard error.

meiofauna at the experimental site was higher because of another settlement of very small (< 500 μm) bivalves and influx of ostracods, in about the same numbers that the control site had received them a few days previously.

By the 20th day, abundance at both sites fell, possibly a seasonal effect, with the numbers at the experimental site becoming significantly lower than at the control site. This difference was maintained through the 50th and 200th day samples, becoming more pronounced by the 235th day samples, when the control site had twice as many animals as the experimental, a quite significant difference. Analysis of the August data shows that this difference was maintained throughout the summer of the first year.

During the second year of the study, abundance of meiofauna declined during the fall months at the control site and increased at the experimental site, with the result that the November 1979 samples are not significantly different at the two study sites. Meiofauna at the control site increased more rapidly during the winter, with the differences between the two sites reappearing. Both the February 1980 and the May 1980 samples from the experimental site are about 75% of the density of the control site. These differences are not as great as the previous year, when the experimental site was only 47% of the control (in May 1979). In August 1980, concurrent with the sharp decrease in the density of animals at several of the benthic characterization study sites, a reduction occurred in the density of the meiofauna at the control site, with an increase at the experimental site.

Integration over time of the average daily standing stock of meiofauna at the two study sites to obtain a mean daily value for the first year showed the experimental site to be significantly lower, at 528.67/10 cm^2, than the control site, at 841.01/10 cm^2.

Integrating the second year's data similarly, the experimental site remains somewhat lower, with 774.83/10 cm^2, than the control site, with 885.75/10 cm^2. This difference in the average daily standing stock for the second year is not significant ($P \leq 0.05$).

Samples taken during December 1980 at the beginning of the third year of the study show very high densities of meiofauna at both the study sites. The abundance of 1693.4/10 cm^2 at the experimental site is not significantly different from the abundance of 1623.1/10 cm^2 at the control site. It is, however, significantly higher than 97% of the meiofauna samples taken in Lake Pontchartrain during the course of the two concurrent studies. Fluctuations of this magnitude are not uncommon in benthic studies. Coull and Bell (1979) describe a station where over the course of 63 months of sampling, the density varied by over an order of magnitude.

It is possible that if the study sites were sampled in May 1981 (the time of the year when differences have been most pronounced), they may not be significantly different. Because of their very small size and the importance of the size to volume ratio, the meiofauna have the best chance of surviving in the sediments that have been altered by dredging.

An examination of the patterns of abundance in the nematodes (Figure 45) shows more clearly the actual dynamics at the two sites. There are no significant differences at the experimental and the control sites before the experimental dredging. After the fifth day, the differences are significant and remain so until the end of the first year. Patterns for the second year are quite similar to that of the total meiofauna.

Abundance of Macrofauna

An examination of the changes in the abundance of macrofauna found at the two study sites (Figure 46, Table 19, and Table B2) reveals a complex pattern. Initial macrofaunal numbers were not significantly different at the control site and the experimentally dredged site. By the second day after the dredging, the numbers of animals at the experimental site were reduced to 54% of the initial numbers. A settlement of the two small gastropods Probythinella louisianae and Texadina sphinctostoma caused a considerable increase in the numbers by the fifth day. These animals did not appear to survive well in the altered sediments and by day 20, the numbers had fallen again. Further settlement of small molluscs at day 50 again increased the total numbers, but by day 205, the experimental site was reduced to 54% of the numbers at the control site.

After the opening and subsequent closing of the Bonnet Carré Spillway into Lake Pontchartrain, the two study sites were sampled again (day 235, May 1979), and total population at the experimental site had dropped further, to 38% of the control site. We can only speculate about this additional drop in benthic community numbers. It is possible that the additional stress of the rapid change in salinity and water temperatures at the study site resulting from the opening of the Bonnet Carré Spillway caused the change in the numbers at the experimental site, which was already stressed by the effects of the experimental dredging. It is also possible that the numbers would have dropped whether the Spillway was opened or not.

In August 1979 at the end of the first year after dredging, the experimental site had only 40.7% of the total numbers of animals that were present initially. The control site had more than twice as many animals present as the experimental site and was not significantly different from initial values.

In November of 1979 a settlement of small gastropods occurred again, with the total numbers at the two study sites again not significantly different. By May 1980 the two sites were again different; the experimental site was reduced to 50.3% of the control site. Figure 47 shows the macrofauna divided into two fractions: gastropods and non-gastropods. The gastropod fraction ranged in percent of total animals from 62 to 98%, with the overall mean for 78 observations being 86.1%. The non-gastropod fraction includes both the bivalves, which make up 11.51%, and the non-mollusc fauna, which make up only 2.42%, of the total, with the chironomid larvae accounting for 1.16% of the total, or almost half of the non-mollusc fauna. Figure 47 illustrates one of the

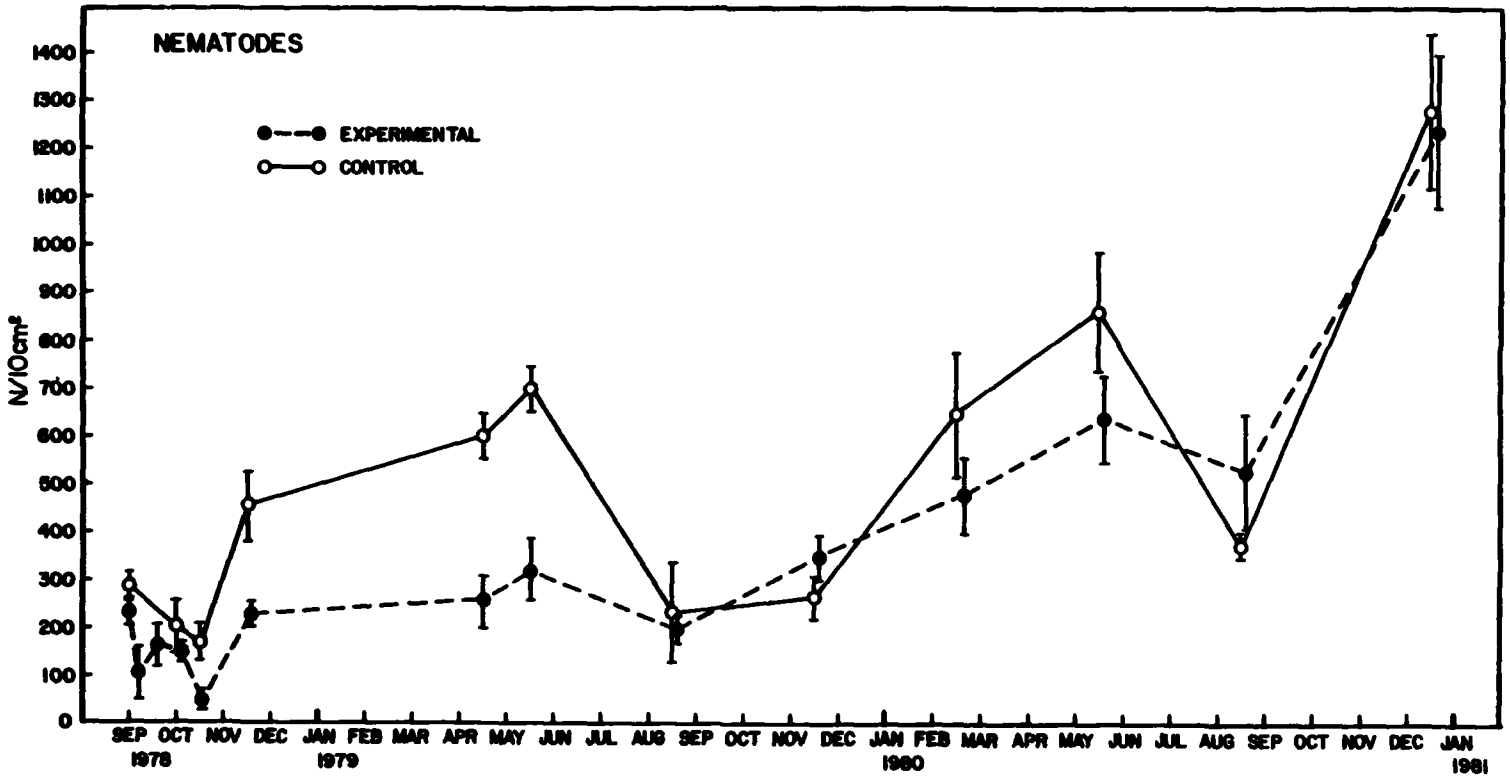


Figure 45. Nematode abundance at the control site and experimental site. Circles indicate the mean of 4 samples; bars represent the associated standard error.

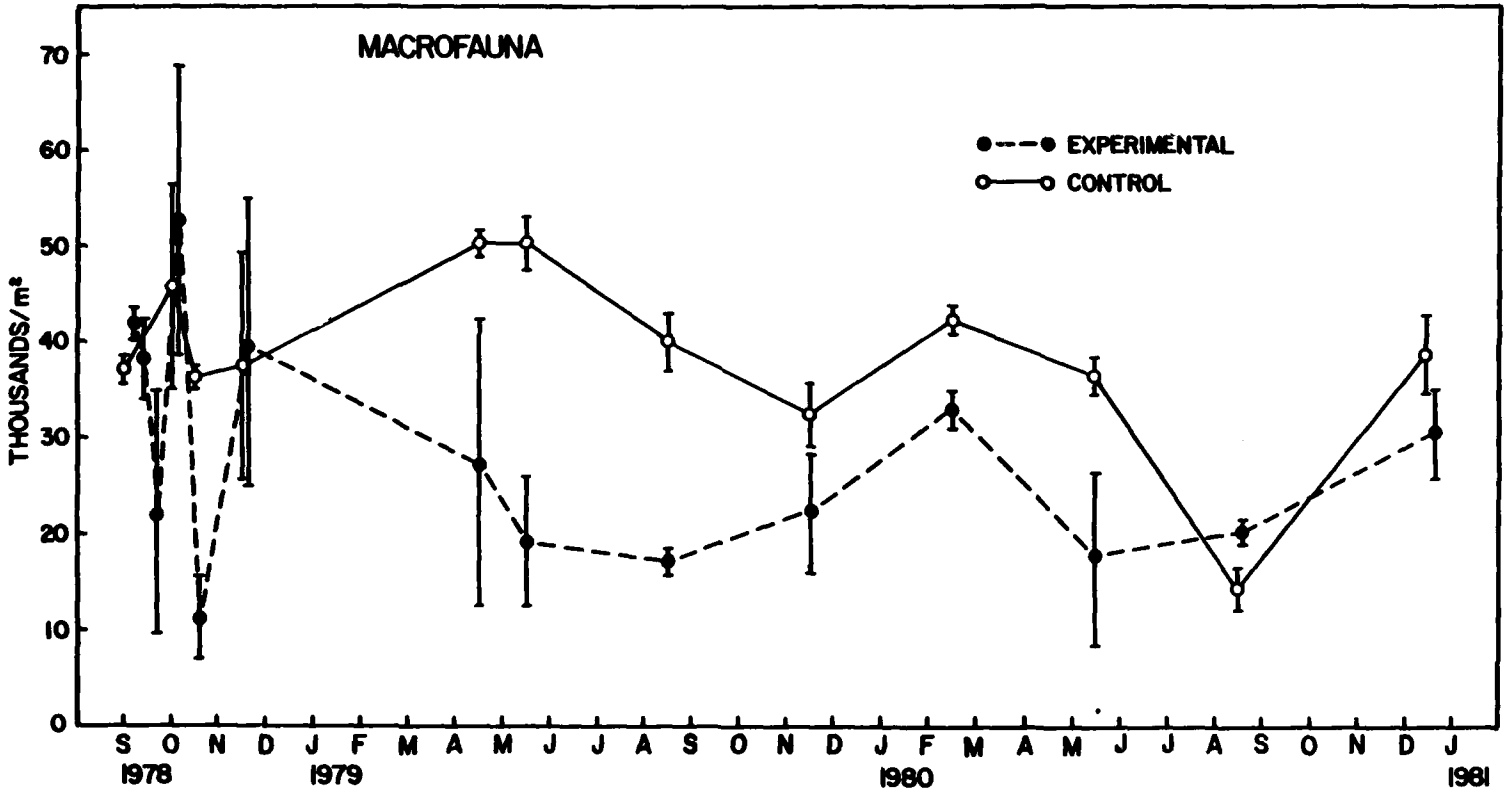


Figure 46. Macrofauna abundance at the control site and experimental site. Circles indicate the mean of 3 samples; bars represent the associated standard error.

Table 19. Macrofauna, major components

	Experimental			Control		
	N/M^2	SE	%	N/M^2	SE	%
			Initial			
Bivalves	4,766	173	11.4	3,743	105	9.9
Gastropods	36,534	746	87.0	33,284	581	88.4
Non-mollusc	683	43	1.6	664	85	1.7
TOTAL	42,013	607		37,660	477	
			Day 1			
Bivalves	4,825	1,355	12.5			
Gastropods	33,223	2,617	86.5			
Non-mollusc	378	155	1.0			
TOTAL	38,426	3,811				
			Day 2			
Bivalves	1,569	460	6.9			
Gastropods	20,616	12,769	90.9			
Non-mollusc	505	79	2.2			
TOTAL	22,689	13,310				
			Day 5			
Bivalves	2,197	767	4.1	4,274	774	9.3
Gastropods	50,633	14,524	94.2	40,927	9,606	89.2
Non-mollusc	911	237	1.7	72	168	1.5
TOTAL	53,741	15,211		45,873	10,318	
			Day 20			
Bivalves	1,009	868	8.4	2,988	388	8.2
Gastropods	10,559	3,073	87.8	32,765	50	89.9
Non-mollusc	457	137	3.8	686	64	1.9
TOTAL	12,026	3,873		36,440	369	
			Day 50			
Bivalves	4,749	1,938	11.8	2,701	252	7.2
Gastropds	33,698	12,816	84.1	33,419	11,083	89.5
Non-mollusc	1,641	266	4.1	1,227	93	3.3
TOTAL	40,089	14,730		37,347	11,125	
			Day 200			
Bivalves	8,413	3,848	31.0	9,101	997	18.0
Gastropods	17,461	10,866	64.2	40,586	436	80.4
Non-mollusc	1,307	384	4.8	790	125	1.6
TOTAL	27,181	15,021		50,477	1,557	

Table 19. (Continued)

	Experimental			Control		
	N/M^2	SE	%	N/M^2	SE	%
	---------- Day 235 ----------					
Bivalves	4,629	543	24.5	9,201	269	18.4
Gastropods	14,008	5,861	73.9	40,281	2,402	80.5
Non-mollusc	308	71	1.6	581	106	1.1
TOTAL	18,946	6,456		50,063	2,555	
	---------- Aug 79 ----------					
Bivalves	3,042	102	17,8	4,023	258	10.0
Gastropods	13,390	1,490	78.3	35,568	3,303	88.3
Non-mollusc	672	230	4.0	672	127	1.7
TOTAL	17,105	1,295		40,263	3,651	
	---------- Nov 79 ----------					
Bivalves	1,892	443	7.5	2,008	481	6.2
Gastropods	19,879	5,164	87.8	28,975	3,136	89.4
Non-mollusc	1,387	321	5.5	1,420	189	4.4
TOTAL	23,158	5,880		32,403	3,394	
	---------- Feb 80 ----------					
Bivalves	2,959	765	8.7	2,272	227	5.4
Gastropods	30,017	1,986	88.4	38,492	722	91.7
Non-mollusc	988	76	2.9	1,161	162	2.8
TOTAL	33,963			41,955	1,131	
	---------- May 80 ----------					
Bivalves	679	232	3.7	1,434	113	3.9
Gastropods	17,374	8,750	94.4	34,925	1,482	95.4
Non-mollusc	352	20	1.9	254	45	0.7
TOTAL	18,405	8,800		36,614	1,552	
	---------- Aug 80 ----------					
Bivalves	232	51	1.2	225	22	1.6
Gastropods	19,502	704	97.8	13,601	1,794	95.9
Non-mollusc	207	19	1.0	356	119	2.5
TOTAL	19,941	651		14,182	1,726	
	---------- Dec 80 ----------					
Bivalves	7,919	1,662	25.2	8,533	389	22.2
Gastropods	22,319	3,915	62.0	28,386	3,781	73.8
Non-mollusc	1,180	142	3.8	1,525	225	4.0
TOTAL	31,418	5,425		38,444	3,975	

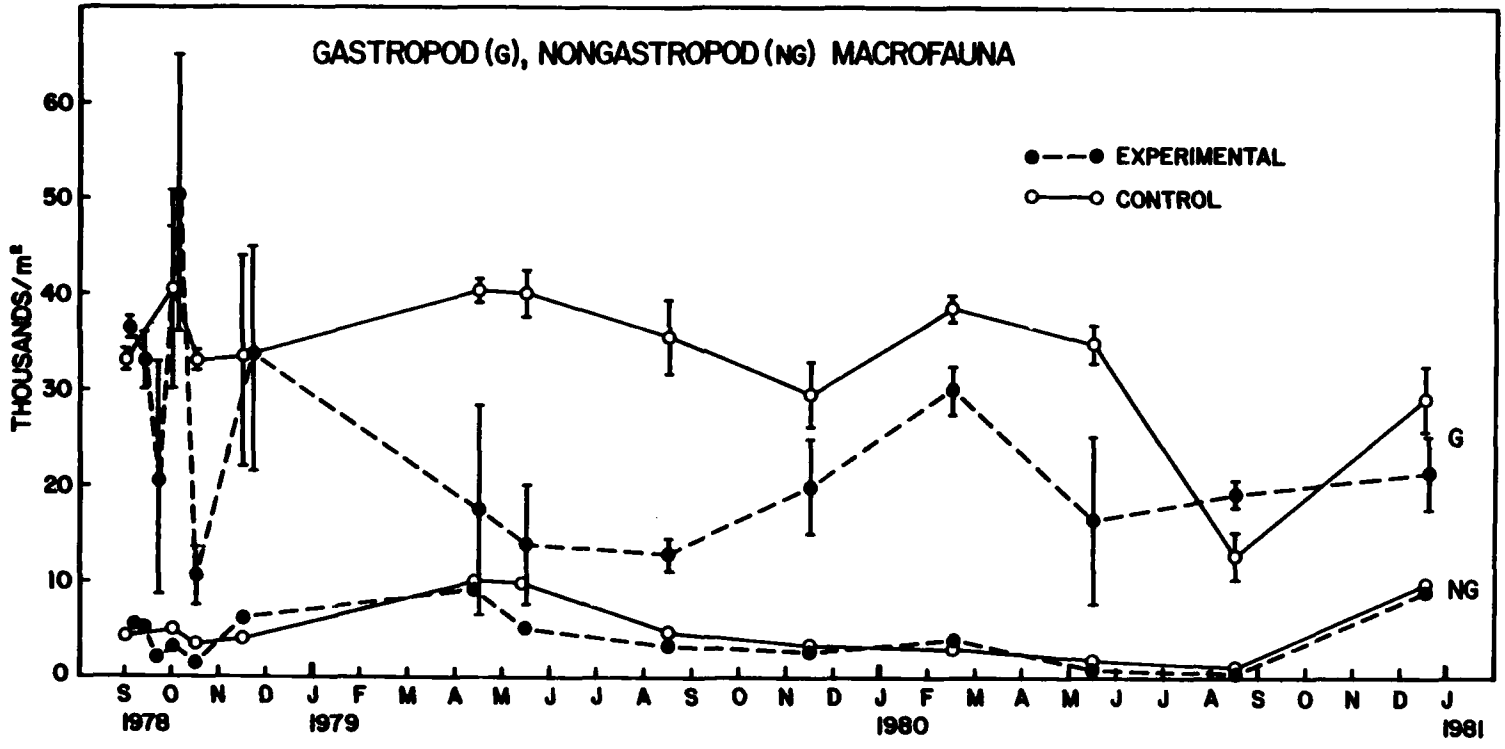


Figure 47. Gastropod abundance at the control site and experimental site. Upper series of circles indicate the mean of 3 samples; bars represent the associated standard error. Nongastropod Macrofauna abundance is indicated by the lower series of circles.

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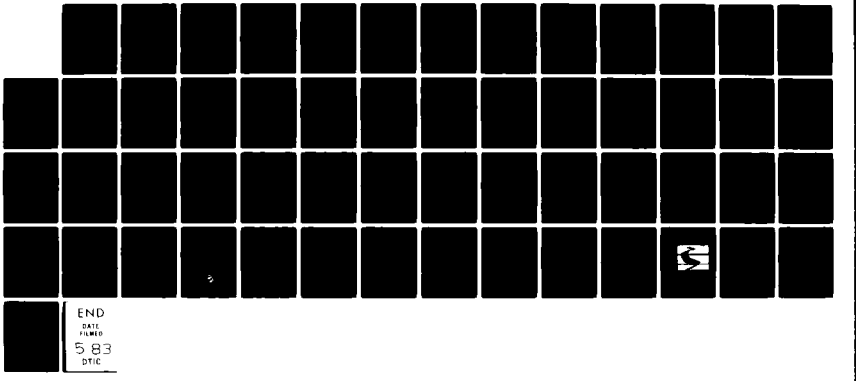

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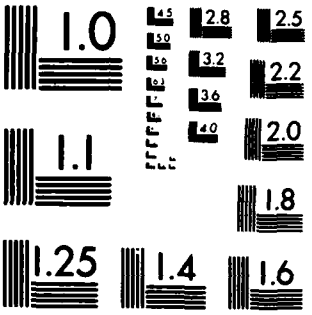

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unique features of the Lake Pontchartrain benthic fauna: the very high dominance of the gastropod fraction. The non-gastropod fauna were not significantly different at the two study sites except immediately following the dredging of the experimental site and again in May 1979, following the opening and closing of the Bonnet Carré Spillway.

In May 1980 the non-gastropod fauna were not significantly different at the two sites, but they comprised only 5% of the total numbers. In August 1980 the control site numbers were significantly lowered, to 35.2% of the total numbers for the previous August. At no time during the course of the study preceding that month had numbers at the control site ever fallen so low. This occurred at the same time that stations a few miles to the south and east of the control site had been found, during the course of the concurrent benthic characterization study, to have had their populations decimated. This finding would lead to the conclusion that this sudden decrease is not a normal seasonal event but the result of some as yet undiscovered stress to the lake, possibly an introduction of halogenated hydrocarbons, which were found in significant amounts at the stations south and east of the control site. By December 1980 the control site had regained its previous density of organisms and again, like in November 1978 and November 1979, the two study sites were not significantly different.

Integration over time of the average daily standing stock to obtain a mean value for the year, i.e., including the time between sampling dates, showed the two study sites to be significantly different the first year of the study, with a mean for the control site of 44,059 organisms, and for the experimental site, a mean of 27,480. Integration of the average daily standing stock during the second year of the study showed a lowered mean for the control site, 34,274, and also a lowered mean for experimentally dredged site, 23,512. The second year showed only a 6.2% improvement in the dredged site over the first year. It will not be possible to assess the degree to which the experimentally dredged site will recover until after May 1981, because this is the time when the two study sites have shown the greatest differences. If at that time the two sites are shown to be not significantly different, then the experimental site could be considered to have recovered.

Species Diversity

In addition to measures of abundance, certain aspects of community structure were measured. Diversity of all collections was measured using the Shannon-Wiener index, $H' = -\Sigma p_i \log_e p_i$. The diversity of a community depends on two things: the number of species and the evenness with which the individuals are distributed among them (Pielou 1975). A community with a few, evenly distributed species can have the same diversity as one with many, unevenly represented species. It is therefore useful to examine not only the diversity of the collections at the two study sites but also the evenness and species numbers. Evenness was measured using $J' = H'_{max}$, where $H_{max} = \log_e S$, or $J' = H'/\log_e S$.

Several groups were not identified to species. Chironomids are lumped, for instance, and treated as one species, although an examination

by an expert of all specimens collected in the two concurrent studies showed more than 99% of Ablabesmyia sp. and less than 1% of Cryptochironomus sp. and Coelotanypus sp. The nemerteans all belong to one undescribed species. The number of species of oligochaetes and ostracods has not yet been completely determined. Since these groups each contribute only a tenth of one percent to the total numbers and occur in less than a fourth of the samples, it is not likely that the lumping will bias the results to any significant degree.

Each value given for diversity, evenness, and species number (Table 20) is expressed as the mean and standard error of the measurements made on the samples collected at each study site on each date.

The initial samples taken at the control site and the experimentally dredged site were not significantly different ($P \leq 0.05$) in diversity, in evenness, or in species number. The mean values for diversity, for evenness, and for species numbers at the two study sties are not significantly different ($P \leq 0.05$). Differences do arise during the course of the study. At day 205, for instance, the macrofauna abundance at the control site is almost double that at the experimental site, yet the diversity at the experimental site is significantly higher. This is principally caused by the higher evenness at the experimental site, which in turn is caused by the lower numbers of the two dominant species, the gastropods (Figure 47, Table 19).

Numerical Analysis

The results of the numerical analysis (Bloom et al. 1977) using the classification produced by the normal analysis of the macrofauna data (Figure 48) are influenced by the fact that the Canberra Metric dissimilarity measure was chosen specifically to lessen the effects of the wide ranges of numbers of individuals of certain species. We then can look for evidence of groupings due to community structure as well as species abundance. At a very low level of similarity, two groups are formed. The upper group is characterized by more "high" attributes (high diversities or high numbers, or high species numbers), the second group by more "low" ones. Starting at the top, 26 and 12 are the numbers designating the samples from the two study sites in December 1980. They both had significantly higher species numbers and high diversity. Number 23, the February 1980 collection from the experimental site, and 7, 9, 10, the August 1979, February, and May 1980 samples from the control site exhibit relatively high evenness. Numbers 19 and 18, the April 1979 and November 1978 experimental site samples, and 5 and 6, April and May of 1979 samples are all characterized by high diversity and high species numbers, with 5 and 6 also having the highest total abundance of the collections. The lower group, beginning with 11 and 25 the August 1980 samples from the two study sites, is characterized by low diversity, low total abundance, low species numbers, and low evenness. Numbers 15 and 17 are the day 2 and day 20 collections at the experimentally dredged site and are possibly the two days exhibiting the greatest effects from dredging, with low diversity, low total abundance, low species numbers, and low evenness. These four collections form a small group separated from the others at a significant level. The next group, 20 and 24, are

Table 20. Macrofauna; species numbers, evenness, and diversity

	Control			Experimental		
	Species N	Evenness	Diversity	Species N	Evenness	Diversity
Initial	12.33 ± 1.20	0.343 ± 0.005	0.858 ± 0.035	12.33 ± 1.20	0.337 ± 0.010	0.838 ± 0.014
Day 1				8.67 ± 0.88	0.398 ± 0.025	0.859 ± 0.094
Day 2				9.67 ± 0.33	0.399 ± 0.082	0.898 ± 0.175
Day 5	13.00 ± 2.30	0.308 ± 0.018	0.771 ± 0.049	12.33 ± 1.76	0.239 ± 0.019	0.597 ± 0.063
Day 20	13.67 ± 0.88	0.273 ± 0.014	0.711 ± 0.028	12.33 ± 0.88	0.280 ± 0.044	0.707 ± 0.124
Day 50	15.00 ± 0.00	0.366 ± 0.063	0.992 ± 0.171	15.67 ± 0.88	0.439 ± 0.039	1.207 ± 0.113
Day 205	13.33 ± 0.33	0.430 ± 0.003	1.114 ± 0.020	16.33 ± 0.88	0.558 ± 0.032	1.557 ± 0.085
Day 235	13.00 ± 1.00	0.448 ± 0.011	1.147 ± 0.027	9.00 ± 0.57	0.615 ± 0.056	1.341 ± 0.082
Aug 79	10.00 ± 1.58	0.453 ± 0.029	1.022 ± 0.022	9.00 ± 1.73	0.579 ± 0.055	1.230 ± 0.071
Nov 79	12.33 ± 0.33	0.399 ± 0.019	1.001 ± 0.039	12.00 ± 1.15	0.435 ± 0.021	1.077 ± 0.063
Feb 80	12.67 ± 1.45	0.407 ± 0.015	1.026 ± 0.027	10.67 ± 1.20	0.466 ± 0.003	1.141 ± 0.040
May 80	9.33 ± 0.33	0.399 ± 0.002	0.892 ± 0.016	10.33 ± 0.67	0.515 ± 0.127	1.208 ± 0.318
Aug 80	9.00 ± 0.57	0.399 ± 0.010	0.877 ± 0.045	8.67 ± 0.80	0.320 ± 0.004	0.691 ± 0.023
Dec 80	16.67 ± 0.33	0.430 ± 0.011	1.210 ± 0.033	18.33 ± 0.67	0.421 ± 0.020	1.224 ± 0.043
Means	12.53 ± 0.44	0.388 ± 0.016	0.968 ± 0.044	11.81 ± 0.52	0.430 ± 0.020	1.041 ± 0.075

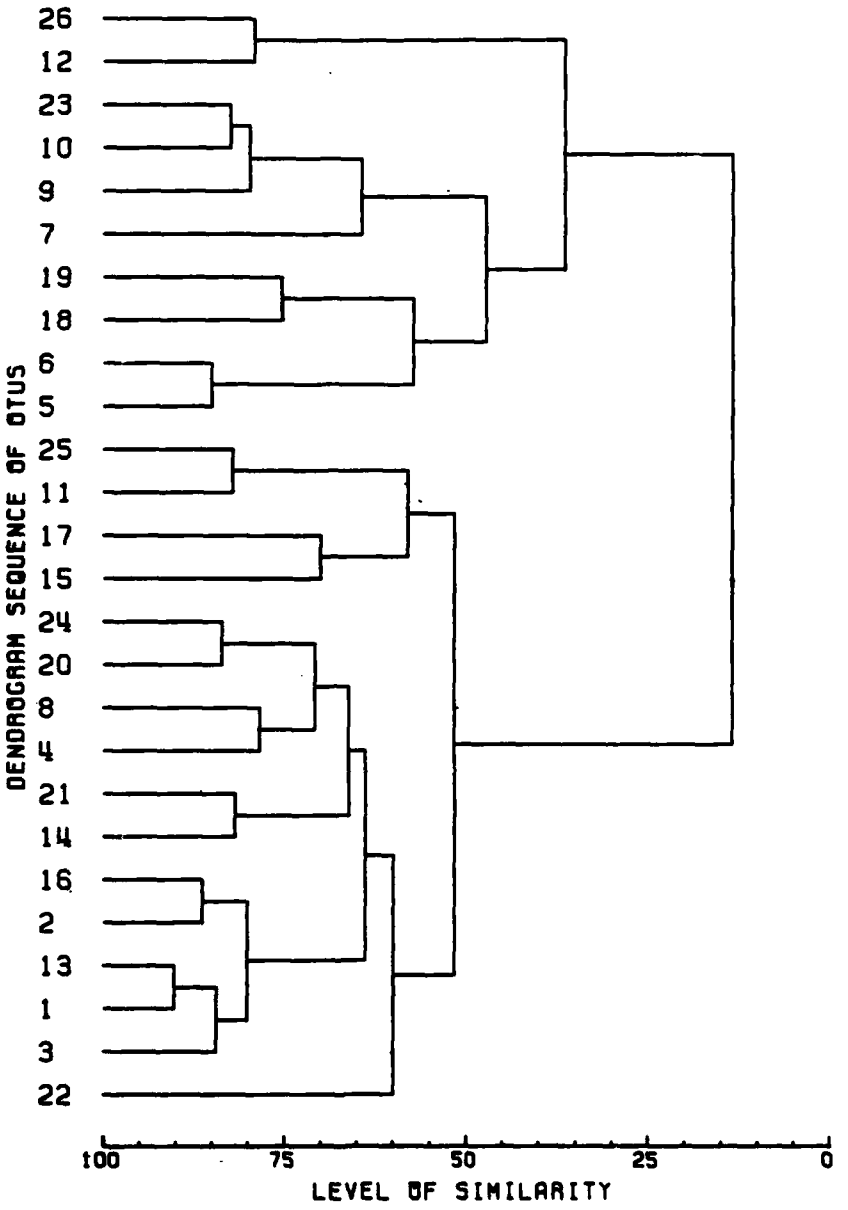

Figure 48. Dendrogram illustrating classification of collections from study sites. Numbers 1 through 12 represent initial to final collections from control site; number 13 through 26 represent initial to final collections from experimental site.

the May 1979 and May 1980 collections from the experimental site with low total abundance and low species numbers. The November 1978 and 1979 collections from the control site, 4 and 8, are the lowest total numbers which occurred at that site, except for the extremely low numbers which occurred in August 1980. Numbers 21 and 14, are the August 1979 and day 1 collections at the experimental site and have very low species numbers, with, for instance, no amphipods. The next group, 16, 2, 13, 1, and 3, includes the first three samples at the control site and the Initial and day 5 samples at the experimental site. All are characterized by low diversities and low evenness. Little seasonality is evidenced, and the scattering of the collections from both study sites through both groups after the effects of abundance were suppressed indicates that the samples were obtained from essentially the same community. The last collection, 22, has total abundance almost as low as the group 25, 11, 17, and 15, but higher diversity; almost as low as 20 and 24, but lower diversity. It is separated from both these groups at a significant level of dissimilarity.

Ordination by principal coordinate analysis using the dissimilarity matrices generated by Canberra metric (Figure 49) makes the groups more obvious perhaps, at least the extreme ones. High on the left are the very low diversity, low number collections. On the extreme right the high diversity, high number collections. Number labels are the same as on the dendrogram.

Biomass

Biomass determinations were done on all identified organisms. No size differences were seen in organisms from the two study sites. The very small size of the species common to the two study sites and the high dominance of two species in particular, the small gastropods Probythinella louisianae and Texadina sphinctostoma, created a very even distribution of biomass with reference to the distribution of abundance. If one sample, for instance, had a total of twice as many animals as another, the biomass of that sample would be twice that of the other. Since all weights involved were ash-free dry weights, the biomass will be proportional to carbon content also or to energy content. When we are discussing a change in the numbers, the same proportional change occurred in the biomass. In May 1979 the mean abundance of macrofauna at the control site was 50,063/m^2, and the biomass was 11.61 g/m^2. At the experimental site, the mean abundance was 19,113/m^2, and biomass, 4.83 g/m^2. Abundance at the experimental site was 38% of the control site; biomass was 42%. Table 21 illustrates this correspondence in numbers and biomass. The mean numbers of macrofauna in all samples from both study sites are shown ranked by number first, and then biomass. The first six species make up 98.72% of the total abundance and 98.87% of the total biomass. The average abundance over all samples was 33,162/m^2, and the average biomass was 8.13g/m^2.

The meiofauna show very similar patterns. The mean abundance at the control site in May 1979 was 1094.1/10cm^2, and biomass was 1.1975mg/10cm^2. Mean abundance at the experimental site was 517.4/10cm^2, and biomass was 0.4912mg/10cm^2. Abundance at the experimental site was 47% of the

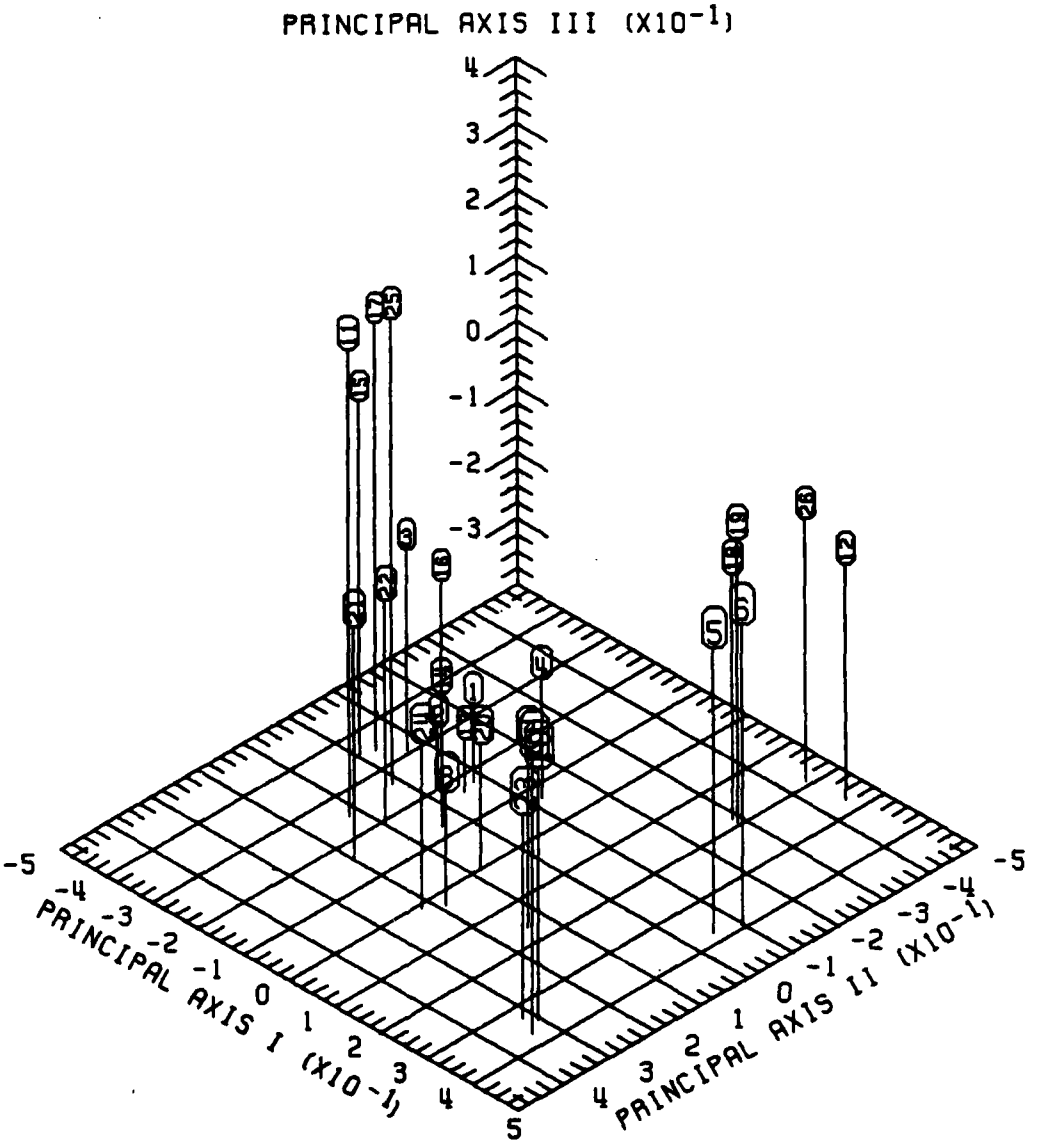

Figure 49. Ordination by principal coordinate analysis of collections from the control site (1-12) and the experimental site (13-26).

Table 21. Macrofauna; mean number and biomass per m^2, percent numbers and biomass

Rank $\bar{N}$	Biomass	Species	$\bar{N}$	$\bar{N}/m^2$	Biomass (afdw) (g/m^2)	% N	% Biomass
1	1	Texadina sphinctostoma	1993.9487	21,719	4.2917	65.49	52.80
2	2	Probythinella louisianae	626.0385	6,819	2.2380	20.56	27.53
3	3	Rangia Cuneata	209.0897	2,278	1.0251	6.87	12.61
4	4	Mulinia pontchartrainensis	78.6282	856	0.4054	2.58	4.99
5	5	Mytilopsis leucophaeta	62.6923	683	0.0434	2.06	0.53
6	6	CHIRONOMIDS	35.3718	385	0.0334	1.16	0.41
7	11	Hypaniola florida	8.6923	95	0.0034	0.29	0.04
8	14	Monoculodes edwardsi	5.2949	58	0.0022	0.17	0.03
9	18	OSTRACODS	3.7179	41	0.0002	0.12	0.01
10	13	OLIGOCHAETES	3.4872	38	0.0023	0.11	0.03
11	17	Streblospio sp.	2.9103	32	0.0005	0.10	0.01
12	7	NEMERTEANS	2.7820	30	0.0275	0.09	0.34
13	12	Corophium lacustre	2.2564	25	0.0028	0.08	0.03
14	9	Macoma mitchelli	2.0000	22	0.0150	0.07	0.18
15	15	Edotea montosa	1.9744	22	0.0015	0.07	0.02
16	8	Mysidopsis almyra	1.7692	19	0.0218	0.06	0.27
17	16	Mediomastus californiensis	1.5769	17	0.0007	0.06	0.01
18	10	Parandalia americana	0.7179	8	0.0095	0.02	0.12
		All others		15	0.0037	0.04	0.04
		TOTAL		33,162	8.1281 g/m^2		

1st 6 species; cumulative %; 98.72 numbers, 98.87 biomass.

control, while biomass was 41%. The mean number of meiofauna animals in all samples from both study sites was 737.2/10cm^2, and biomass was 0.8650mg/10cm^2.

Discussion

In order to assess the effects of hydraulic shell dredging on the benthic fauna of Lake Pontchartrain, the total abundance, relative abundance, species composition, and community structure at an experimentally dredged site and a nearby control site have been examined. A number of statistical and numerical methods have been used in an attempt to distinguish the effects of the hydraulic shell dredging on the structure of the benthic community from the effects of other environmental stresses, both natural (such as abrupt salinity fluctuations) and anthropogenic (such as toxic substances).

In addition to its relative importance in food web dynamics in shallow estuarine systems such as Lake Pontchartrain, the benthic community is also more vulnerable to any environmental stresses than any other group. The sedentary habits of benthic organisms subject them to a greater array of stresses that mobile groups could avoid. This vulnerability makes the benthos the best indicator of environmental stress in estuarine systems (Copeland and Bechtel 1971). Consequently, many published studies of the effects of various stresses on benthic communities exist, with varied evaluations of the results. A common problem with many of these "before and after" studies is the degree to which large-scale natural fluctuations in the community can mask changes due to environmental stress (Eagle 1975). The use of a control site, while doubling the number of samples needed, is necessary for examining the degree to which fluctuations are natural or maninduced. In addition, long-term monitoring, such as has been carried out along the coast of Sweden (Ankar and Elmgren 1976; Leppäkoski 1975; and Rosenberg and Möller 1979) is valuable in separating natural from induced fluctuations. Historical data from Lake Pontchartrain, while interesting, are semi-quantitative at best, so reliance on the use of control site data has been necessary in this study.

One striking feature of the benthic community of Lake Pontchartrain is its consistently low species diversity. Overall species diversity of the lake obtained during the initial survey of 86 stations was 1.37 ± 0.04 (Bahr et al. 1980). The initial and final average measures of diversity at the dredged experimental site and the undredged control site were not significantly different from each other, and while slightly lower than the mean for the lake, were not significantly different from earlier values from that area.

Values for species diversity have been shown by various investigators to have a negative correlation with environmental stress, particularly with pollution (Wilhm and Dorris 1968; Woodwell 1970; Copeland and Bechtel 1971; Goodman 1975; and Ruggiero and Merchant 1979).

Diversity measures are reported using both $\log_e$ and $\log_2$; All diversities quoted in this study have been converted to $\log_e$. The commonly accepted standards for polluted systems then become: $H < 1.443$, severe pollution; $1.443 < H < 4.328$, moderate pollution; $H > 4.328$, clean water (Wilhm and Dorris 1968). This is an empirical categorization based on results of extensive environmental sampling. For instance, in a study of the species diversity and water quality in Galveston Bay, Texas, Copeland and Bechtel (1971) found that areas receiving the greatest amount of toxic effluent exhibited the lowest mean annual diversity. Overall benthic diversity in that highly polluted system was 0.840 ± 0.374, not significantly different from Lake Pontchartrain.

Gray (1978) has pointed out that the use of Caswell's neutral model of diversity (Caswell 1976) can place diversity measures related to pollution on a more theoretical basis. Briefly, the neutral model predicts the level of diversity if no biological interactions, such as predation or competition, occur. Hutchinson (1958) on the basis of niche theory, predicts a higher diversity where interactions are strong. In testing the neutral model predictions of diversity and evenness, the table of Caswell (1976) was used. The table is abbreviated, however, and a series of curvilinear regressions were constructed using his table in order to extend the table to cover a greater range.

The diversity predicted for the experimental site by the neutral model was 1.115, not significantly different from the diversity of 1.108 ± 0.146 occurring at the end of the first year. The predicted value for the control site was 1.070, not significantly different from the diversity of 1.068 ± 0.035 occurring. This matching of the neutral model is predicted theoretically for highly polluted, or disturbed systems, where species equilibrium is altered and higher dominance and lower species diversity ensues, or wherever the influence of abiotic factors swamp the influence of biological interactions. Evenness at the control site, 0.453 ± 0.029, and the experimental site, 0.579 ± 0.055, are also not significantly different from the theoretical values of 0.464 and 0.516.

Other low salinity, polluted estuaries have similarly low diversities, for instance, the Baltic, 1.05 (Ankar and Elmgren 1976) and 1.3 (Rosenberg and Möller 1979), Hampton Roads Area "mud" stations 1.59 ± 0.26 (Boesch 1973).

In addition to measures of diversity, which say nothing about the species compositon of a sample or a collection of samples, Haedrich (1975) urges that a measure of overlap be included in all environmental impact statements. He describes data from nine estuaries where areas of apparent high pollution have low annual diversity, H, and high overlap, PS (percent similarity). The species overlap, PS, at the dredging and control sites was 92.86%, indicating the apparent affects of pollution or disturbance on both sites.

Other measures of community structure, such as abundance, show marked differences at the control and experimental sites. The lack of recovery by the end of the first year is evidenced by the reduced abundance at the experimentally dredged site. Both meiofauna and macrofauna communities show reduced abundance. The total meiofauna is reduced to

57% at the experimental site compared to the control site. The dominant group, the nematodes, are reduced to 38% of the control by this time and 42% of mean annual abundance of the control. This pattern of similar diversity measures and different annual abundance continued through the second year.

In addition to examining the usual measures of community structure such as abundance, species diversity, and overlap at the two sites, a comparison of the distributional pattern of individuals within species at each site was made. Species-abundance data from a large enough sample from a heterogeneous population can usually be fitted to a log-normal distribution (Gray 1979, May 1975). Practical limits to the size of samples and the probability of misidentifying a specimen from a rare species as a member of a commoner species results more often in the distribution's being truncated on the left side (Slocumb et al. 1977; Krebs 1972). The method of plotting species-abundance data into geometric classes is extremely robust in that any errors arising from limited samples, or overly cautious identification, results only in the first point in a series (Figure 50) being below the line (Gray and Mirza 1979). After plotting data from a series of benthic surveys, Gray and Mirza have demonstrated the use of log-normal plots as an indicator of pollution-induced disturbance on estuarine benthic communities (1979). They postulate that a slope of over 50° would indicate an unpolluted or undisturbed community, whereas a slope of under 35° would indicate a community affected by pollution. Plotting the species-abundance data for the August 1979 collections at the dredging control site and the dredging experimental site (Figure 50) shows an angle of 19°, which would indicate a community that has reached an equilibrium under polluted or disturbed conditions, since transitional communities very often show a break in the line (Gray and Mirza 1979).

Information on mean abundance and mean biomass of several other benthic communities has been summarized in Table 22. The average organism in other benthic communities is 32 times larger than the average for Lake Pontchartrain macrofauna. Not only are the species which occur in the lake smaller representatives of their genera or families, but the size of the individuals are smaller than the average for collections of some of the same species from other areas in Louisiana and from other states. This condition has been noted by taxonomic experts who have confirmed species identifications for us.

In addition to small species and to small sizes within species, there were an unusually low total number of species present. In contrast to the mean of 12.01 ± 0.36 species present in the present study collections, Boesch (1973) found 23.00 ± 3.73 in a similar low diversity habitat. Conner and Simon (1979), in a study on the effects of shell dredging on a soft-bottom community in Tampa Bay, Florida, identified a mean of 79.5 benthic species before dredging and 47.5 after dredging. Even with a loss of 41% of the species, Tampa Bay still retained four times as many species as the two study sites in Lake Pontchartrain have.

Darnell, during the course of a survey of benthic invertebrates in Lake Pontchartrain in 1953-55 (unpublished manuscript), found the same limited number of benthic species. He did, however, find a somewhat

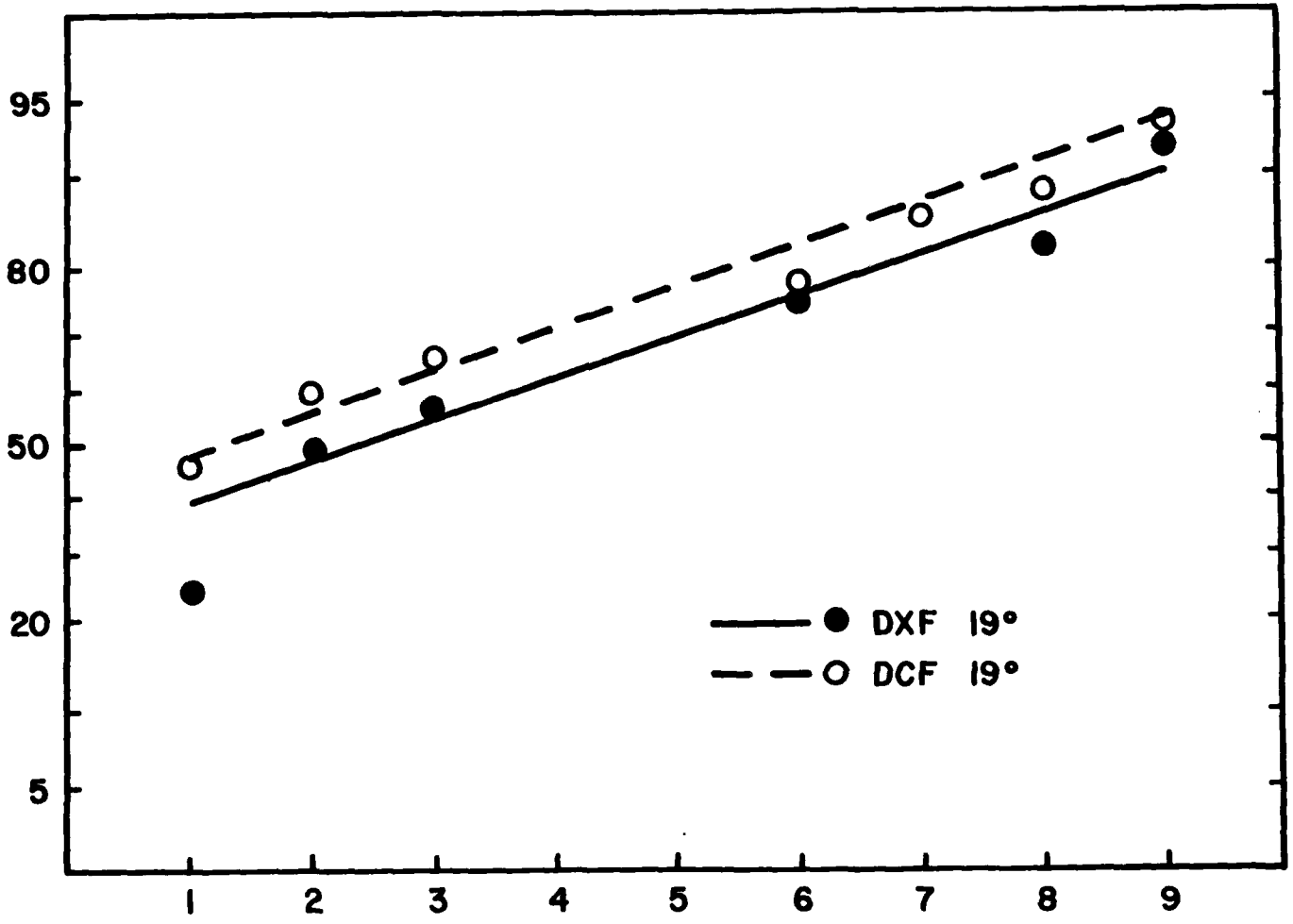

Figure 50. Log-normal plots; cumulative percents (vertical axis) of species are plotted against geometric class (horizontal axis) of individuals per species. Circles represent pooled samples for each study site.

Table 22. Macrofauna, numbers and biomass from the literature

Study site	Sieve size, mm	$\tilde{N}/m^2$	Biomass g/m^2	Biomass mg/animal
Long Island Sound (N. Y.) (Sanders 1956)	1.0	16,466	54.627	3.32
Martha's Vineyard (Mass.) (Wigley and McIntyre 1964)	1.0	2,477	10.362	4.18
Goose Creek (N. Y.) (Kaplan et al. 1974)	1.4	1,201	29.460	24.53
Lynher Estuary (V. K.) (Warwick and Price 1975)	0.5	1,436	13.240	9.22
Baltic (Sweden) (Ankar and Elmgren 1976)	1.0	3,547	10.480	2.95
Tampa Bay (Florida (Conner and Simons 1979)	0.5	18,550	27.505	1.48
Lake Pontchartrain (This study)	0.5	33,618	8.128	0.24

different distribution of organisms within the species. His reported numbers of small gastropods were an order of magnitude lower than found in the present study. Additionally, Rangia greater than 20 mm were found, as mentioned earlier in this report (p. 23), in numbers far exceeding those found during this study. If biomass determinations had been done on Darnell's collections, his total average biomass would probably have been similar to that found in this study. The difference would have been in the proportion of biomass found in each species. Rangia would have probably been ranked higher in percent biomass than either of the gastropods. Since Rangia are a preferred food of the blue crab (Callinectes sapidus), with 64% of the crabs' total stomach volume made up of Rangia (Darnell 1958), it is probable that the documented decrease in number of larger (> 20 mm) Rangia has had a significant effect on food available to blue crabs.

These very high numbers of small organisms are a result of a typical ecological strategy in ecosystems where predation is a major, unpredictable source of mortality for all life history stages of all species. We can think of the dredge as a very large, non-selective benthic predator, periodically removing every organism from an area of mud. In addition, a large percentage of the pelagic larvae present in the area are "consumed" by or pumped through the dredge, which pumps 83.23 m^3/min while actively dredging. There is, then, an advantage to be gained by any species that matures rapidly and reproduces frequently. This is typical "r-strategy," and usually is found in small organisms with rapid population growth (Miller 1979, Southwood 1977).

Many small organisms would fit this description, thus the high dominance by the two small hydrobiid gastropods requires additional examination of ecological factors. The altered bulk densities of the sediments would affect species equally. Any small species remaining in the area after the 20 years of dredging that occurred before 1953 would probably have had a somewhat more evenly distributed abundance if dredging were the only factor. It is probable that increasing levels of pollution in the last 20 years have also had an effect. In order to ensure that different levels of pollution at the two study sites did not mask or confuse the effects of dredging in the study by causing a differential mortality due to pollution stress, several sets of sediment samples were analyzed for PCB content. The two study sites had sediment levels that were not significantly different ($P \leq 0.05$) from each other. The mean of 0.32 ± 0.04 µg/g PCB's found at the two study sites during the first months of the study is significantly higher than levels of 0.17 ± 0.13 µg/g quoted for levels in the Great Lakes (Eisenreich, Hollod, and Johnson 1979). Amphipods are, as a group, more sensitive to toxic substances than molluscs or polychaetes (Reish and Barnard 1979). Polychaetes vary in sensitivity. Species known to be tolerant to many toxic substances, which, in fact, are used as indicators of polluted conditions, include Capitella capitata, Nereis succinea, Streblospio benedicti, and Polydora ligni (Reish 1979). Molluscs seem somewhat less sensitive to certain toxic substances (Menzel 1979). Hydrobiid gastropods are particularly tolerant of chlorinated hydrocarbons (Brown 1980). Differential response to the increasing levels of toxic substances would appear to be an important factor in the high dominance of the two hydrobiid gastropods, Probythinella louisianae and Texadina sphinctostoma.

Summary

We have examined the effects of hydraulic clam shell dredging on the benthic fauna of Lake Pontchartrain. We have found that it is likely that species numbers were reduced by dredging that occurred before 1954 as shown by the low species number in Darnell's unpublished work. Further changes in benthic community structure, such as lowered diversity and lowered evenness caused by the increased dominance of two species, may be attributed to increasing levels of toxic substances in the sediments. Present day dredging in this low diversity benthic community has significant effects on the abundance of both macrofauna and meiofauna through the first year after the experimental dredging at the study site and significant effects through the second year on the macrofauna. We cannot yet say whether the area has recovered to the original, initial abundance.

OVERVIEW

Lake Pontchartrain is a large, shallow, semi-enclosed estuarine system that has sustained a clam dredging industry since 1933. To understand the effects of shell dredging in this system, it is necessary to understand the difference between the nature of clam shell dredging and oyster shell dredging. In the latter, disturbances are concentrated over reef deposits and, although they may be severe, they are somewhat localized, resulting in a "pothole" effect. Large areas of a system may be left unaffected by this type of shell dredging. In clam shell dredging, dredges move continuously, extracting Rangia shells, which are dispersed widely over most of the system. Presently in Lake Pontchartrain, 44% of the lake is open to dredging, and it has been shown that an area equivalent to this is dredged in from 1.4 to 2.3 years. Considering the 48-year span of shell dredging in Lake Pontchartrain, the inevitable conclusion is that the system as a whole is affected by shell dredging. Estimating the extent is contingent on demonstrating the magnitude of shell dredging effects. The present study has quantified physical effects to the sediments chemical effects to the sediments, and water column, and biological effects to the benthic invertebrate community.

The immediate physical effects of shell dredging on the water column were found to be temporary, with most of the spoil sinking rapidly through the water column and spreading out on the bottom, filling in trenches or cuts. Very little material contributes to the visible plume, which is of a temporary nature and has little relationship to the distribution of dredged sediments. The effects to the redeposited dredged sediments are more substantial and take the form of reduced bulk densities. Bulk densities of fresh dredge spoil were found to be 1.22 g/cm^3. Bulk densities below 1.30 g/cm^3 are considered fluid mud by Diaz and Boesch (1977), who conclude that "its unique physical properties have deleterious effects on benthic fauna." Bulk densities of less than 1.30 g/cm^3 have been found at stations in Lake Pontchartrain, including in an area that has been closed to dredging for 27 years. In a laboratory consolidation experiment, dredge spoil was found to consolidate 39% of a 75 cm high sediment column in seven months. Under field conditions in the shallow lake, consolidation provides the mechanism for maintaining low bulk densities as wave-resuspended sediments fill in depressions. Low sediment bulk densities can be detrimental to sedentary benthic organisms, particularly on larger and denser organisms, which would continually sink in the sediments.

Resuspension of sediments by wind-induced wave action was found to occur to a greater degree at the dredged site than at the control site. This was evidenced by greater increases in the water column at the dredged site of particulate iron and maganese on day 20, following three days of high winds.

The effects of shell dredging on the nutrients nitrogen and phosphorous and on the heavy metals iron, manganese, lead, zinc, and copper were investigated in the water column, in the sediment-water interface, and in the sediments. All parameters studied were affected by dredging in all three environments. Many of these effects, however, were of relatively short duration, lasting from a matter of hours to one to two

days, and then returning to conditions not significantly different from those measured concurrently at a control site 0.4 km distant. Iron and manganese levels were elevated in the water column over the dredged area; levels in the dredged sediments were depressed. While the effects on iron and manganese persisted over a 50-day period and indicate that dredging facilitates the transfer of particulate forms of these elements from the bottom sediments to the water column, an overall interpretation is unclear at this time. The filtered interface water (at sediment-water interface) showed greatly elevated levels of the heavy metals lead, zinc, and copper at the dredged station over the control station betweeen days 5 and 20 following dredging. The peak in the lead level (84 µg/l) exceeds the minimum acute toxic effluent value of 50 µg/l based on ecological effects (Luthy and Carter 1980). High metal levels alone do not indicate that adverse environmental effects will occur; it is, however, the bioavailability in any given sediment that is important (Neff, Foster, and Slowey 1978). Both macrofauna and meiofauna showed sharp declines at the dredged station during the same period that the heavy metals concentrations were elevated in the interface water. Evidence that hydrobiid gastropods are susceptible to heavy metals (Brown 1980) coupled with the dominance of the macrofauna by hydrobiids would indicate that heavy metals could be a contributing factor to benthic mortality in dredged sediments.

Levels of orthophosphate in the water were not appreciably altered except during actual dredging and at least one hour afterwards. The orthophosphate returned to the original levels the day following dredging. Nitrogen, total N, and ammonium (NH_4^+) were elevated in the water column over the dredge site for a period upwards of 50 days. Three peaks in (NH_4^+)-N were measured, the first during dredging, the second 20 days after dredging, and the third 259 days after dredging. The latter two peaks follow two precipitous declines in the meiofauna and macrofauna, in October 1978 and May 1979. Ammonium is produced in degradation of material high in amino acid content, such as proteins under anaerobic conditions (DeLaune, Patrick, and Brannon 1976). It appears that dredging can increase some nutrients in the water column at the expense of benthic organisms. The long-term effects of shell dredging on the benthic community at the experimentally dredged site have been pronounced. The importance of long-term studies and of frequent sampling immediately following dredging becomes evident from examination of the data presented in the present study. If only the samples taken on day 5 and day 50 following dredging were examined, one might erroneously conclude that the benthic community had not only recovered but had exceeded the control site in numbers of organisms. Examination of samples taken on day 20 reveals, however, that the larger settlement of organisms apparent on day 5 did not survive, nor did the increased numbers present on day 50, as evidenced from later samples. This phenomenon of large settlement of organisms from the water is believed to occur in dredged areas because they have been denuded of resident organisms by dredging and subsequent die-offs resulting in wide open areas without competition. Previous short-term studies on the effects of shell dredging have recorded this phenomenon.

Results of this study indicate pronounced seasonal effects not only in the undisturbed benthic community but also in the response of the

disturbed community at the dredged site. Peak densities of benthic organisms occur in Lake Pontchartrain during the winter months; lowest densities occur in late summer. During the winter, the densities of organisms at the dredged and control sites are not significantly different. In the spring, densities at the dredged site fall significantly below those at the control site and remain at low levels through the summer. This same pattern has occurred for two years, which indicates that the experimentally dredged site has not recovered. Macrofauna appear to be affected to a greater extent than the meiofauna. The December 1981 sampling did not show significant differences between the two sites; however, a determination of whether the dredged site has recovered could not be made unless it is sampled in May 1981. Existing data indicate the sites have not recovered.

To the ecosystem, recovery of a dredged site may be important. The effects, however, of shell dredging manifest themselves more so in the loss of biological production during the recovery period. If we use the difference in mean annual standing stock of the macrofauna and meiofauna at the two study sites as the loss attributable to dredging, convert to biomass, and multiply by an appropriate (production/biomass) P/B ratio, then the loss in benthic production can be calculated. Standing stock and biomass values from this study are used along with macrofauna P/B ratios from Robertson (1979) and meiofauna P/B ratios from Warwick, Joint, and Radford (1979). The total production lost in two years is 21.73 g/m^2 or in the 7.17×10^8 m^2 of lake available for dredging, 15.6×10^6 kg. To put it another way, 17,000 tons ash free dry weight (afdw) or an average of 8,500 tons annually is lost and thus unavailable to higher trophic levels such as benthic feeding fish and blue crabs.

Several measures of community structure have been examined to assess to what degree and at what rate the benthic community of Lake Pontchartrain can recover from the effects of hydraulic clam shell dredging. Care has been taken to try to separate the effects of increasing levels of pollution from the effects of the dredging. It appears that species numbers were reduced originally by dredging before Darnell's 1954 study (unpublished manuscript). Reduction of species numbers by dredging is well documented (Cronin et al. 1970, Taylor et al. 1970, Sykes and Hall 1970, Diaz and Boesch 1973, Kaplan et al. 1974, Conner and Simon 1979). The fact that no reduction in species numbers took place in this study does not contradict other studies. The other studies lead us to believe, rather, that the reduction in species numbers took place between 1933 when dredging began, and Darnell's 1954 survey.

Loss of the large (> 20 mm) Rangia that occurred in substantial numbers some time after 1954 can be attributed to the consequent widespread alteration in sediment bulk densities in the central lake to a level equivalent to fluid mud.

Peddicord (1980), in an extensive review of the direct effects of suspended sediments on aquatic organisms, points out that contaminated fluid muds can have substantial, acute impacts on benthic fauna. These impacts vary not only with the types and concentrations of contaminants in the fluid mud but also with the species exposed. Shrimp experienced less mortality in tests with contaminated sediments than did young

crabs, which died during molting (Peddicord et al. 1975). In addition to direct effects on adult macrofauna, fluid muds can create indirect effects where they form a blanket over bottom areas critical to the juvenile life stages of aquatic organisms (Hirsch et al. 1978).

An additional effect of the destabilization of contaminated sediments is an increase in the availability of the toxic substances bound to microparticulates to filter feeders or to other organisms in the water column. Recently, Harding and Phillips (1978) found that diatoms can absorb (or adsorb) low concentrations of PCB's bound to particles, which effectively reduces photosynthetic activity. It has been assumed that PCB's were extracted from the sediment by benthic organisms and reintroduced to the food chain in the overlying water column (Eisenreich et al. 1980), since PCB's are not very soluble, and once introduced into the water, are rapidly sedimented. Introduction of PCB's into the planktonic food web by resuspension of contaminated sediments is one of the mechanisms by which fish in the Great Lakes have reached FDA actionable levels while the water concentrations remain below detection levels (Eisenreich et al. 1979).

Species diversity in a brackish system is never as high as in a freshwater or marine system. It is probable, however, that in the past Lake Pontchartrain would have had a species diversity of 2.4 to 2.8, similar to other healthy brackish systems (Rosenberg and Möller 1979). It is difficult to separate the effects of dredging from the effects of pollution on species diversity. For example, small gastropods have a greater tolerance for certain toxic substances than other small organisms such as amphipods and polychaetes (Menzel 1979, Reisch and Barnard 1979). They are thus able to use the resources made available by the loss of the larger Rangia (attributable to sediment destabilization) and the decreased competition from more sensitive organisms.

In addition to the lowering of species numbers, any concomitant lowering of species diversity by dredging had probably begun before Darnell's study. Diversity has been further lowered by pollution to the point where the only species remaining are opportunistic, or "weed," species. These species are small; individuals within the species are small, and appear to breed frequently, with large fluctuations in population densities. An examination of the meiofauna in the data appendix (Appendix B1) will show no sampling period when small, juvenile gastropods did not appear in the samples. Occasionally, numbers of meiofauna-size gastropods would equal the number of gastropods in the macrofauna on an equal area basis. This typical pattern of "r-selected" species assemblages is caused by the dredging in the sense that dredging is acting as a "non-selective" predator, and only those species that breed more frequently than an average of than once every 1.4 to 2.3 years will remain in the system.

We have discussed the adaptations within the macrofaunal components of the benthic community. The meiofaunal components show the same adaptations. If we consider copepods as representative of the meiofaunal groups present, we find the number of species present very low (Fleeger, Appendix D). Species diversity (H') of the meiobenthic copepods at the experimental site was not significantly different ($P \leq 0.05$) from the

species diversity at the control site, nor from the initial species diversity of the macrofauna. At the end of the first year of the study, August 1979, the measured diversity at the experimental site continued to not be significantly different from the control site and the macrofauna species diversity at the two study sites. This was also true in August 1980.

In summary, Lake Pontchartrain is a perturbed system, the result, historically, of continuous perturbation from shell dredging operations and from pollution. Shell dredging produces fluid mud that is deleterious to benthic fauna. Contaminated fluid mud is potentially even more harmful to benthic organisms. To what extent this potential is actually realized in a particular system will depend on the occurrence and areal distribution of the fluid mud (Peddicord 1980). In Lake Pontchartrain, shell dredging has spread fluid mud over nearly all of the open lake.

Sediments at the dredged and control sites in central Lake Pontchartrain were found to contain high levels (0.32 μg/g) of PCB's. Fluid muds have the potential for producing high suspended sediment concentrations (Peddicord 1980), and wave action in Lake Pontchartrain has the potential for resuspending sediments frequently (Swenson 1980). Transfer of PCB's from microparticulates to marine phytoplankton has been demonstrated as has the fact that PCB's inhibit phytoplankton photosynthesis and cell division (Harding and Phillips 1978) and thus, ultimately, primary production. Reduced primary production means less food eventually reaches the bottom. The combined effects of contaminated fluid mud on the bottom and reduced food availability to benthic organisms have resulted in a benthic community in the open lake that is characterized by: (1) a low number of species; (2) a low species diversity; (3) small individuals; and (4) a low biomass. These conditions have developed over many years, and it is not possible to predict, even if shell dredging operations were to cease, whether, or to what extent, the lake system would recover.

CONCLUSIONS

The results of the experimental shell dredging, assessed in the light of the literature reviewed for this study, lead to the following conclusions:

(1) Hydraulic shell dredging in Lake Pontchartrain produces fluid mud of a bulk density of 1.22 g/cm^3 from silty clay sediments. Low bulk density sediments persist for long periods in the lake.

(2) Shell dredging at a site in the central region of the lake produced a detrimental effect on the macrofauna and the meiofauna, resulting in lower densities in spring and summer at the dredged site compared to the control site. These effects have persisted for two years.

(3) Shell dredging resulted in a loss of production by the benthic community during the two years of the study. Biomass production was reduced by 38% the first year and by 25% the second year after dredging. Over the two-year period, a total of 32% of the biological production was lost at the dredged site compared to the control site.

(4) Seasonal patterns of settlement from the water column of benthic larvae and juveniles during late fall produce similar abundances at the two study sites. Differential survival at the two sites resulted in a loss in abundance and in biomass at the dredged site and a gain in these two parameters at the control site by late spring. Consequently, the dredged site appears not to have fully recovered from the effects of hydraulic clam shell dredging.

(5) Shell dredging did not produce a change in species diversity between the dredged site and the control site. Lake Pontchartrain has a low species diversity compared to other low salinity systems. Diversity has probably been reduced in the past by pollution and by the dredging of contaminated sediments. Presently, shell dredging is affecting the abundance of the animals in the benthic community.

(6) The benthic community of Lake Pontchartrain has reached an equilibrium with present levels of pollution and consists of a low number of species characterized as being tolerant of pollution.

(7) On a system-wide basis, the production of fluid mud, lowered sediment bulk densities, resuspension of contaminated sediments, and lowered primary production in the water column together have the potential to maintain a benthic community of low biomass in the open lake. Less food is thus available to benthic-feeding organisms such as fish and blue crabs.

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APPENDIX A
BENTHIC MACROFAUNA; LAKE PONTCHARTRAIN CONTROL
AND EXPERIMENTAL DREDGING EFFECTS STATIONS

Phylum Platyhelminthes
Class Turbellaria
Phylum Rhynocoela
1 unidentified species
Phylum Annelida
Class Oligochaeta
Class Polychaeta
Hypaniola florida Hartman
Laeonereis culveri (Webster)
Nereis succinea (Frey and Leukart)
Parandalia americana (Hartman)
Mediomastus californiensis Hartman
Streblospio sp. cf. *benedicti* Webster
Capitella capitata (Fabricius)
Polydora sp.
Phylum Mollusca
Class Gastropoda
Probythinella louisianae (Morrison)*
Texadina sphinctostoma (Abbott and Ladd)
Class Bivalvia
Mytilopsis leucophaeta (Conrad)
Macoma mitchelli Dall
Mulinia pontchartrainensis Morrison
Rangia cuneata (Gray)
Phylum Athropoda
Class Crustacea
Subclass Ostracoda
Subclass Malacostraca
Order Isopoda
Cassidinidea lunifrons (Richardson)
Edotea montosa (Stimpson)
Cyathura polita (Stimpson)
Order Amphipoda
Cerapus benthophilus Thomas and Heard
Corophium lacustre Vanhoffen
Grandidierella bonnieroides Stephenson
Melita nitida Smith
Monoculodes edwardsi Holmes
Order Mysidacea
Mysidopsis almyra Bowman
Order Decapoda
Callinectes sapidus Rathbun
Rhithropanopeus harrisii (Gould)
Class Insecta
Order Diptera
Family Chironomidae
Ablabesmyia sp.

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*In this report *P. louisianae* = *P. protera*. See Heard (1979) for complete taxonomic discussion.

APPENDIX B
Table B1
Meiofauna

Experimental Station Meiofauna ($\bar{N}$/10cm^2 ± SE)	Initial	Day 1	Day 2
NEMATODES	228.7 ± 20.9	104.9 ± 49.4	167.6 ± 41.8
COPEPODS	68.8 ± 10.9	33.6 ± 6.7	19.4 ± 4.6
COPEPOD NAUPLII	50.4 ± 11.2	44.3 ± 4.1	51.4 ± 13.4
OSTRACODS	30.1 ± 6.1	4.6 ± 1.5	2.0 ± 1.4
ROTIFERS	27.0 ± 6.2	27.0 ± 7.0	68.3 ± 21.7
TURBELLARIANS	8.2 ± 1.4	1.0 ± 0.6	8.2 ± 3.7
POLYCHAETES	13.3 ± 3.8	3.1 ± 1.8	2.0 ± 0.8
OLIGOCHAETES	0.5 ± 0.5	0.5 ± 0.5	
BIVALVES	37.7 ± 3.2	21.4 ± 6.3	87.1 ± 11.8
GASTROPODS	73.3 ± 8.7	17.8 ± 2.1	32.1 ± 12.3
OTHERS	1.5 ± 1.0		0.5 ± 0.5
TOTAL	540.4 ± 10.9	263.3 ± 47.1	439.0 ± 81.8

Control Station Meiofauna ($\bar{N}$/10cm^2 ± SE)	Initial
NEMATODES	288.8 ± 21.4
COPEPODS	94.2 ± 7.0
COPEPOD NAUPLII	66.7 ± 12.0
OSTRACODS	49.4 ± 4.7
ROTIFERS	13.3 ± 1.3
TURBELLARIANS	9.2 ± 1.3
POLYCHAETES	15.8 ± 5.1
OLIGOCHAETES	1.5 ± 1.0
BIVALVES	68.8 ± 10.1
GASTROPODS	50.9 ± 6.9
OTHERS	3.6 ± 0.5
TOTAL	662.6 ± 43.4

Table B1. (Continued)

Experimental Station Meiofauna ($\bar{N}$/10cm^2 ± SE)	Day 5	Day 20	Day 50
NEMATODES	151.3 ± 12.9	47.4 ± 12.9	234.3 ± 20.3
COPEPODS	13.2 ± 3.6	26.5 ± 8.6	37.2 ± 5.5
COPEPOD NAUPLII	58.0 ± 12.7	33.1 ± 9.4	46.9 ± 10.9
OSTRACODS	3.6 ± 1.6	6.1 ± 4.2	31.1 ± 2.3
ROTIFERS	11.7 ± 3.9	13.2 ± 5.0	33.6 ± 8.5
TURBELLARIANS	11.2 ± 2.9	2.0 ± 0.8	15.8 ± 7.4
POLYCHAETES	6.1 ± 1.9	5.6 ± 1.9	22.9 ± 4.0
OLIGOCHAETES	1.0 ± 0.6		0.5 ± 0.5
BIVALVES	33.6 ± 6.3	14.3 ± 4.3	56.5 ± 9.0
GASTROPODS	35.1 5.7	4.1 ± 2.2	67.8 ± 29.1
OTHERS	1.0 ± 0.6	0.5 ± 0.5	1.5 ± 1.0
TOTAL	329.5 ± 30.0	152.8 ± 42.4	548.0 ± 38.5

Control Station Meiofauna ($\bar{N}$/10cm^2 ± SE)	Day 5	Day 20	Day 50
NEMATODES	196.1 ± 55.2	173.2 ± 31.6	457.4 ± 77.6
COPEPODS	26.5 ± 9.0	34.6 ± 6.6	68.3 ± 6.7
COPEPOD NAUPLII	41.8 ± 13.4	68.3 ± 13.3	71.8 ± 4.2
OSTRACODS	17.3 ± 9.8	16.8 ± 4.3	27.0 ± 11.6
ROTIFERS	13.8 ± 2.8	13.3 ± 4.5	36.2 ± 13.1
TURBELLARIANS	5.6 ± 3.0	4.1 ± 1.5	15.3 ± 6.0
POLYCHAETES	6.9 ± 2.9	5.6 ± 2.1	29.6 ± 8.7
OLIGOCHAETES		1.5 ± 1.0	2.6 ± 1.5
BIVALVES	45.3 ± 12.3	38.2 ± 10.8	36.2 ± 12.1
GASTROPODS	70.8 ± 21.4	44.8 ± 16.3	56.6 ± 31.6
OTHERS	1.5 ± 0.5	4.1 ± 2.8	3.1 ± 2.4
TOTAL	427.8 ± 119.9	405.9 ± 71.2	803.7 ± 161.0

Table B1. (Continued)

Experimental Station Meiofauna ($\bar{N}$/10cm^2 ± SE)	Day 205	Day 235	Aug 79
NEMATODES	264.9 ± 52.8	324.4 ± 60.2	202.7 ± 30.4
COPEPODS	108.0 ± 25.2	40.7 ± 7.3	151.8 ± 21.0
COPEPOD NAUPLII	118.2 ± 25.7	32.1 ± 6.2	38.4 ± 1.9
OSTRACODS	18.8 ± 3.1	53.0 ± 9.6	45.1 ± 23.7
ROTIFERS	28.5 ± 7.4	44.8 ± 11.0	31.9 ± 4.3
TURBELLARIANS	43.3 ± 14.0	13.8 ± 1.7	3.1 ± 2.4
POLYCHAETES	12.2 ± 5.0	3.6 ± 1.8	
OLIGOCHAETES		0.5 ± 0.5	
BIVALVES	7.1 ± 3.9	0.5 ± 0.5	33.5 ± 8.1
GASTROPODS	7.7 ± 5.0	1.0 ± 0.6	47.4 ± 8.0
OTHERS	1.5 ± 1.0	2.5 ± 1.0	1.0 ± 0.6
TOTAL	609.6 ± 76.9	517.4 ± 83.4	552.6 ± 53.8

Control Station Meiofauna ($\bar{N}$/10cm^2 ± SE)	Day 205	Day 235	Aug 79
NEMATODES	592.6 ± 44.0	712.0 ± 43.7	238.0 ± 94.9
COPEPODS	58.8 ± 13.9	81.5 ± 5.3	56.9 ± 13.0
COPEPOD NAUPLII	69.3 ± 15.1	70.8 ± 12.9	58.0 ± 14.6
OSTRACODS	28.5 ± 8.6	88.6 ± 16.6	236.9 ± 52.3
ROTIFERS	8.7 ± 1.5	42.8 ± 17.9	93.3 ± 16.6
TURBELLARIANS	15.3 ± 7.0	79.5 ± 11.9	8.2 ± 0.8
POLYCHAETES	17.8 ± 9.0	6.1 ± 2.8	1.0 ± 1.0
OLIGOCHAETES	0.5 ± 0.5		
BIVALVES	2.1 ± 1.2	3.6 ± 1.0	46.0 ± 7.4
GASTROPODS	13.6 ± 1.6	8.7 ± 1.7	41.8 ± 14.1
OTHERS	1.5 ± 1.0	0.5 ± 0.5	36.2 ± 13.8
TOTAL	720.7 ± 131.5	1094.1 ± 41.3	921.3 ± 85.4

Table B1. (Continued)

Experimental Station Meiofauna ($\bar{N}$/10cm^2 ± SE)	Aug 80	Dec 80
NEMATODES	529.2 ± 115.9	1242.2 ± 163.9
COPEPODS	70.3 ± 19.1	94.2 ± 10.9
COPEPOD NAUPLII	147.7 ± 39.4	25.0 ± 5.1
OSTRACODS	5.1 ± 1.0	22.9 ± 2.9
ROTIFERS	86.1 ± 23.3	87.1 ± 27.4
TURBELLARIANS	8.2 ± 1.4	44.8 ± 7.3
POLYCHAETES	73.4 ± 18.0	84.0 ± 8.6
OLIGOCHAETES	2.5 ± 1.3	7.1 ± 2.1
BIVALVES	16.3 ± 6.8	48.4 ± 9.7
GASTROPODS	101.4 ± 28.5	32.1 ± 10.9
OTHERS	1.5 ± 0.5	5.6 ± 1.0
TOTAL	1041.5 ± 173.4	1693.4 ± 197.1
Control Station Meiofauna ($\bar{N}$/10cm^2 ± SE)	Aug 80	Dec 80
NEMATODES	377.4 ± 20.9	1278.3 ± 163.1
COPEPODS	58.1 ± 6.5	60.6 ± 8.5
COPEPOD NAUPLII	170.1 ± 21.3	32.1 ± 2.5
OSTRACODS	4.1 ± 2.0	24.4 ± 4.3
ROTIFERS	28.6 ± 7.6	85.1 ± 4.6
TURBELLARIANS	4.6 ± 1.8	38.2 ± 17.8
POLYCHAETES	65.2 ± 6.4	64.2 ± 5.1
OLIGOCHAETES	23.9 ± 7.1	3.6 ± 1.7
BIVALVES	26.5 ± 12.5	14.8 ± 2.8
GASTROPODS	53.5 ± 12.5	28.5 ± 10.7
OTHERS	3.1 ± 2.0	5.6 ± 3.2
TOTAL	817.0 ± 52.8	1623.1 ± 199.0

Table B2
Macrofauna

Control Station Macrofauna ($\bar{N}/m^2$ ± SE)	mm	Initial	Day 5	Day 20
BIVALVES	mm			
Clams	0- 2	1068 ± 165	596 ± 172	141 ± 27
Rangia cuneata	2-10	472 ± 32	490 ± 73	324 ± 51
	10-20	4 ± 4		7 ± 4
	20-30			
	> 30			
Mulinia pontchartrainensis	2-10	1032 ± 80	1402 ± 226	1071 ± 129
Macoma mitchelli	0- 2		4 ± 4	4 ± 4
	2-10	7 ± 7		4 ± 4
	10-20	7 ± 7	15 ± 7	11 ± 6
Mytilopsis leucophaeta	0- 2	1166 ± 371	1060 ± 203	635 ± 46
	2-10	352 ± 82	706 ± 182	792 ± 236
	10-20			
GASTROPODS				
Probythinella louisianae		3643 ± 211	3428 ± 675	2407 ± 17
Texadina sphinctostoma		29632 ± 570	37500 ± 8944	30358 ± 44
POLYCHAETES				
Hypaniola florida		55 ± 13	123 ± 50	40 ± 10
Laeonereis culveri				4 ± 4
Nereis succinea				
Parandalia americana		4 ± 4		
Mediomastus californiensis		18 ± 9	7 ± 4	11 ± 6
Streblospio sp.		18 ± 7	36 ± 22	7 ± 4
Capitella capitata				
Polydora sp.				
OLIGOCHAETES		4 ± 4	14 ± 9	4 ± 4
TURBELLARIANS				
NEMERTEANS		7 ± 7	22 ± 6	14 ± 4
CRUSTACEANS				
Edotea montosa		22 ± 6	14 ± 7	11 ± 0
Cyathura polita				
Monoculodes edwardsi		7 ± 7	7 ± 7	29 ± 10
Corophium lacustre				
Grandidierella bonnieroides				
Melita nitida				
Cerapus sp.				
Mysidopsis almyra		7 ± 7	7 ± 4	7 ± 4
Copepods		18 ± 18	4 ± 4	
Ostracods				
Rhithropanopeus harrisii			4 ± 4	
Callinectes sapidus				
CHIRONOMIDS		501 ± 50	428 ± 55	552 ± 32
TOTAL		37279 ± 1298	45860 ±10607	36432 ± 375

Table B2. (Continued)

Control Station Macrofauna ($\bar{N}/m^2$ ± SE)	mm	Day 50	Day 205	Day 235 (May 79)
BIVALVES				
Clams	0- 2	308 ± 95	8148 ± 773	7294 ± 470
Rangia cuneata	2-10	479 ± 20	349 ± 76	1129 ± 108
	10-20	33 ± 19	5 ± 5	
	20-30			
	> 30			
Mulinia pontchartrainensis	2-10	537 ± 41	512 ± 98	711 ± 122
Macoma mitchelli	0- 2	4 ± 4	60 ± 38	22 ± 12
	2-10	11 ± 6	22 ± 11	40 ± 4
	10-20	14 ± 7	5 ± 5	4 ± 4
Mytilopsis leucophaeta	0- 2	316 ± 34		
	2-10	988 ± 115		
	10-20	11 ± 11		
GASTROPODS				
Probythinella louisianae		5395 ± 1740	12842 ± 131	14970 ± 1189
Texadina sphinctostoma		28024 ± 9400	27744 ± 566	25311 ± 1231
POLYCHAETES				
Hypaniola florida		236 ± 42	104 ± 49	80 ± 36
Laeonereis culveri				
Nereis succinea				
Parandalia americana			5 ± 5	7 ± 7
Mediomastus californiensis		25 ± 10	49 ± 38	40 ± 7
Streblospio sp.		40 ± 4		7 ± 7
Capitella capitata				
Polydora sp.		4 ± 4		
OLIGOCHAETES		22 ± 7	82 ± 38	58 ± 35
TURBELLARIANS				
NEMERTEANS		18 ± 4	22 ± 0	29 ± 19
CRUSTACEANS				
Edotea montosa		11 ± 11	22 ± 22	36 ± 7
Cyathura polita				
Monoculodes edwardsi		94 ± 25	163 ± 22	109 ± 70
Corophium lacustre				7 ± 7
Grandidierella bonnieroides				
Melita nitida				7 ± 7
Cerapus sp.				
Mysidopsis almyra		18 ± 10	27 ± 5	36 ± 19
Copepods		7 ± 4		14 ± 14
Ostracods				
Rhithropanopeus harrisii				
Callinectes sapidus				
CHIRONOMIDS		744 ± 95	316 ± 22	149 ± 28
TOTAL		37348 ± 11698	50477 ± 1558	50063 ± 2555

Table B2. (Continued)

Control Station Macrofauna ($\bar{N}/m^2 \pm SE$)	mm	Aug 79	Nov 79	Feb 80
BIVALVES	mm			
Clams	0- 2	207 ± 97	716 ± 252	1569 ± 136
Rangia cuneata	2-10	1071 ± 109	493 ± 80	461 ± 31
	10-20			
	20-30			
	> 30		7 ± 7	
Mulinia pontchartrainensis	2-10	2662 ± 60	232 ± 45	47 ± 7
Macoma mitchelli	0- 2			
	2-10			
	10-20	4 ± 4		
Mytilopsis leucophaeta	0- 2	36 ± 7	203 ± 116	91 ± 25
	2-10	44 ± 11	356 ± 95	105 ± 68
	10-20			
GASTROPODS				
Probythinella louisianae		10446 ± 1148	6884 ± 307	18924 ± 165
Texadina sphinctostoma		25122 ± 2269	16778 ± 7905	19567 ± 887
POLYCHAETES				
Hypaniola florida			120 ± 11	312 ± 51
Laeonereis culveri				
Nereis succinea				
Parandalia americana				
Mediomastus californiensis				
Streblospio sp.		4 ± 4	40 ± 7	98 ± 25
Capitella capitata				
Polydora sp.				
OLIGOCHAETES		7 ± 7	4 ± 4	15 ± 14
TURBELLARIANS		4 ± 4		
NEMERTEANS		7 ± 7	7 ± 4	7 ± 7
CRUSTACEANS				
Edotea montosa		25 ± 13	22 ± 6	18 ± 13
Cyathura polita		4 ± 4		
Monoculodes edwardsi		7 ± 4	559 ± 221	
Corophium lacustre			4 ± 4	7 ± 7
Grandidierella bonnieroides				
Melita nitida				
Cerapus sp.			4 ± 4	7 ± 7
Mysidopsis almyra		33 ± 23	14 ± 10	7 ± 4
Copepods				29 ± 4
Ostracods				51 ± 13
Rhithropanopeus harrisii				7 ± 4
Callinectes sapidus			4 ± 4	
CHIRONOMIDS		581 ± 84	631 ± 53	628 ± 62
TOTAL		40263 ± 3651	27078 ± 3758	41955 ± 1130

Table B2. (Continued)

Control Station Macrofauna ($\bar{N}/m^2$ ± SE)	mm	May 80	Aug 80	Dec 80
BIVALVES				
Clams	0- 2	643 ± 241	7 ± 7	4767 ± 623
Rangia cuneata	2-10	247 ± 136	94 ± 51	835 ± 50
	10-20		44 ± 44	25 ± 20
	20-30		7 ± 4	
	> 30			
Mulinia pontchartrainensis	2-10	142 ± 58	36 ± 4	163 ± 23
Macoma mitchelli	0- 2			
	2-10			29 ± 7
	10-20			1333 ± 383
Mytilopsis leucophaeta	0- 2		33 ± 6	1380 ± 402
	2-10	76 ± 65	47 ± 10	
	10-20			
GASTROPODS				
Probythinella louisianae		17733 ± 570	5000 ± 820	4067 ± 600
Texadina sphinctostoma		17192 ± 919	8602 ± 1187	24320 ± 3361
POLYCHAETES				
Hypaniola florida		47 ± 26	65 ± 7	25 ± 13
Laeonereis culveri				
Nereis succinea				
Parandalia americana				51 ± 13
Mediomastus californiensis			7 ± 7	54 ± 6
Streblospio sp.			73 ± 51	18 ± 10
Capitella capitata				7 ± 4
Polydora sp.				4 ± 4
OLIGOCHAETES		7 ± 4	117 ± 54	145 ± 16
TURBELLARIANS				44 ± 13
NEMERTEANS		4 ± 4		211 ± 32
CRUSTACEANS				
Edotea montosa		22 ± 11		
Cyathura polita				
Monoculodes edwardsi				4 ± 4
Corophium lacustre				4 ± 4
Grandidierella bonnieroides				
Melita nitida				
Cerapus sp.				
Mysidopsis almyra		7 ± 4	36 ± 20	11 ± 6
Copepods				25 ± 14
Ostracods				683 ± 211
Rhithropanopeus harrisii				
Callinectes sapidus				
CHIRONOMIDS		167 ± 18	69 ± 7	240 ± 44
TOTAL		36614 ± 1552	14204 ± 1739	38444 ± 3975

Table B2. (Continued)

Experimental Station Macrofauna ($\bar{N}/m^2 \pm SE$)		Initial	Day 1	Day 2
BIVALVES	mm			
Clams	0- 2	1707 ± 123	504 ± 63	131 ± 60
Rangia cuneata	2-10	548 ± 20	918 ± 313	385 ± 143
	10-20	4 ± 4		7 ± 7
	20-30			
	> 30	4 ± 4		4 ± 4
Mulinia pontchartrainensis	2-10	1038 ± 100	1536 ± 240	795 ± 231
Macoma mitchelli	0- 2			
	2-10	7 ± 4		4 ± 4
	10-20	18 ± 7	4 ± 4	
Mytilopsis leucophaeta	0- 2	1024 ± 73	853 ± 212	116 ± 16
	2-10	447 ± 99	520 ± 208	131 ± 45
	10-20			
GASTROPODS				
Probythinella louisianae		3330 ± 227	3969 ± 802	1863 ± 848
Texadina sphinctostoma		33208 ± 531	29254 ± 2371	18753 ± 11990
POLYCHAETES				
Hypaniola florida		40 ± 22		4 ± 4
Laeonereis culveri				
Nereis succinea				
Parandalia americana			4 ± 4	4 ± 4
Mediomastus californiensis		8 ± 7	7 ± 7	4 ± 4
Streblospio sp.		25 ± 16		
Capitella capitata				
Polydora sp.				
OLIGOCHAETES		4 ± 4		4 ± 4
TURBELLARIANS				4 ± 4
NEMERTEANS		14 ± 4	11 ± 7	14 ± 10
CRUSTACEANS				
Edotea montosa		36 ± 7	11 ± 7	14 ± 10
Cyathura polita				
Monoculodes edwardsi		25 ± 7		
Corophium lacustre				
Grandidierella bonnieroides				
Melita nitida				
Cerapus sp.				
Mysidopsis almyra		4 ± 4		4 ± 4
Copepods			76 ± 76	
Ostracods				
Rhithropanopeus harrisii			4 ± 4	
Callinectes sapidus		526 ± 25	232 ± 90	450 ± 81
CHIRONOMIDS				
TOTAL		42011 ± 1248	379[illegible] ± 4360	22689 ± 13428

Table B2. (Continued)

Experimental Station Macrofauna ($\tilde{N}/m^2$ ± SE)	mm	Day 5	Day 20	Day 50
BIVALVES				
Clams	0- 2	251 ± 107	47 ± 25	3145 ± 1533
Rangia cuneata	2-10	432 ± 182	265 ± 199	163 ± 54
	10-20	4 ± 4	7 ± 7	4 ± 4
	20-30			
	> 30			737 ± 114
Mulinia pontchartrainensis	2-10	1202 ± 445	29 ± 15	
Macoma mitchelli	0- 2			
	2-10	4 ± 4	4 ± 4	
	10-20			11 ± 11
Mytilopsis leucophaeta	0- 2	160 ± 57	22 ± 13	138 ± 62
	2-10	141 ± 22	40 ± 40	552 ± 226
	10-20			
GASTROPODS				
Probythinella louisianae		4139 ± 987	1155 ± 464	8209 ± 3189
Texadina sphinctostoma		46494 ± 14323	9404 ± 2689	25489 ± 9634
POLYCHAETES				
Hypaniola florida		29 ± 16	7 ± 7	331 ± 144
Laeonereis culveri				
Nereis succinea				
Parandalia americana				4 ± 4
Mediomastus californiensis		25 ± 10	7 ± 7	44 ± 23
Streblospio sp.		36 ± 36	36 ± 14	211 ± 174
Capitella capitata				4 ± 4
Polydora sp.				7 ± 7
OLIGOCHAETES		55 ± 39	7 ± 7	11 ± 7
TURBELLARIANS				
NEMERTEANS		22 ± 0	14 ± 10	25 ± 4
CRUSTACEANS				
Edotea montosa		14 ± 10	22 ± 17	138 ± 16
Cyathura polita				
Monoculodes edwardsi		22 ± 13	22 ± 17	44 ± 17
Corophium lacustre		4 ± 4	14 ± 14	40 ± 18
Grandidierella bonnieroides				
Melita nitida				
Cerapus sp.				
Mysidopsis almyra		22 ± 17	14 ± 10	44 ± 19
Copepods			156 ± 156	44 ± 31
Ostracods				
Rhithropanopeus harrisii			7 ± 7	
Callinectes sapidus				
CHIRONOMIDS		679 ± 107	149 ± 58	516 ± 195
TOTAL		53741 ± 15211	11441 ± 3786	39842 ± 15532

Table B2. (Continued)

Experimental Station Macrofauna ($\bar{N}/m^2$ ± SE)		Day 205	Day 235	Aug 79
BIVALVES	mm			
Clams	0- 2	3351 ± 1953	2796 ± 523	94 ± 20
Rangia cuneata	2-10	599 ± 381	396 ± 47	519 ± 118
	10-20	14 ± 10		
	20-30			
	> 30			
Mulinia pontchartrainensis	2-10	3083 ± 1718	1209 ± 164	2404 ± 79
Macoma mitchelli	0- 2		73 ± 38	
	2-10	7 ± 4	131 ± 58	
	10-20		7 ± 4	
Mytilopsis leucophaeta	0- 2		18 ± 18	18 ± 10
	2-10	193 ± 193	14 ± 14	7 ± 7
	10-20	76 ± 76		
GASTROPODS				
Probythinella louisianae		4495 ± 1741	5432 ± 2081	4168 ± 252
Texadina sphinctostoma		12966 ± 9155	8576 ± 3785	9222 ± 1360
POLYCHAETES				
Hypaniola florida		138 ± 51		
Laeonereis culveri				
Nereis succinea				
Parandalia americana		14 ± 7		4 ± 4
Mediomastus californiensis		47 ± 13	7 ± 7	
Streblospio sp.		18 ± 7		
Capitella capitata				
Polydora sp.		14 ± 14		
OLIGOCHAETES		179 ± 156	7 ± 7	29 ± 14
TURBELLARIANS				
NEMERTEANS		18 ± 4	29 ± 19	7 ± 4
CRUSTACEANS				
Edotea montosa		22 ± 22	44 ± 44	7 ± 7
Cyathura polita				
Monoculodes edwardsi		112 ± 65	51 ± 19	
Corophium lacustre		410 ± 138		
Grandidierella bonnieroides				
Melita nitida		7 ± 7		
Cerapus sp.				
Mysidopsis almyra		59 ± 31	7 ± 7	62 ± 32
Copepods		4 ± 4	36 ± 19	
Ostracods				
Rhithropanopeus harrisii				4 ± 4
Callinectes sapidus				
CHIRONOMIDS		207 ± 38	127 ± 32	559 ± 195
TOTAL		27125 ± 15365	19113 ± 6923	17105 ± 1296

Table B2. (Continued)

Experimental Station Macrofauna ($\bar{N}/m^2$ ± SE)		Nov 79	Feb 80	May 80
BIVALVES	mm			
Clams	0- 2	265 ± 84	1964 ± 541	479 ± 183
Rangia cuneata	2-10	879 ± 216	530 ± 51	123 ± 40
	10-20		14 ± 10	7 ± 4
	20-30			
	> 30			
Mulinia pontchartrainensis	2-10	341 ± 81	145 ± 58	58 ± 20
Macoma mitchelli	0- 2			
	2-10			
	10-20			
Mytilopsis leucophaeta	0- 2	243 ± 100	98 ± 47	
	2-10	160 ± 35	207 ± 117	11 ± 6
	10-20			
GASTROPODS				
Probythinella louisianae		3842 ± 1179	14001 ± 927	9204 ± 4643
Texadina sphinctostoma		16038 ± 3999	16016 ± 1495	8170 ± 4132
POLYCHAETES				
Hypaniola florida		196 ± 11	265 ± 75	84 ± 13
Laeonereis culveri				
Nereis succinea				
Parandalia americana				4 ± 4
Mediomastus californiensis		4 ± 4		
Streblospio sp.		94 ± 36	40 ± 22	
Capitella capitata				
Polydora sp.				
OLIGOCHAETES		4 ± 4	4 ± 4	18 ± 13
TURBELLARIANS				
NEMERTEANS		4 ± 4	4 ± 4	22 ± 13
CRUSTACEANS				
Edotea montosa		7 ± 7	22 ± 11	7 ± 4
Cyathura polita				
Monoculodes edwardsi		14 ± 14		4 ± 4
Corophium lacustre		276 ± 195	14 ± 7	
Grandidierella bonnieroides				
Melita nitida				
Cerapus sp.				
Mysidopsis almyra		11 ± 11	4 ± 4	7 ± 4
Copepods		4 ± 4		4 ± 4
Ostracods		4 ± 4		
Rhithropanopeus harrisii			7 ± 7	14 ± 14
Callinectes sapidus			4 ± 4	
CHIRONOMIDS		472 ± 254	625 ± 36	178 ± 20
TOTAL		22994 ± 5934	33963 ± 2095	18405 ± 8947

Table B2. (Continued)

Experimental Station Macrofauna ($\bar{N}/m^2$ ± SE)	mm	Aug 80	Dec 80
BIVALVES			
Clams	0- 2	7 ± 7	4201 ± 563
Rangia cuneata	2-10	101 ± 42	853 ± 32
	10-20	11 ± 11	25 ± 25
	20-30		
	> 30		
Mulinia pontchartrainensis	2-10		84 ± 40
Macoma mitchelli	0- 2		14 ± 7
	2-10		22 ± 13
	10-20		
Mytilopsis leucophaeta	0- 2	65 ± 17	1002 ± 414
	2-10	47 ± 22	1699 ± 934
	10-20		18 ± 18
GASTROPODS			
Probythinella louisianae		4927 ± 320	2828 ± 400
Texadina sphinctostoma		14574 ± 485	19491 ± 3520
POLYCHAETES			
Hypaniola florida		4 ± 4	29 ± 13
Laeonereis culveri			
Nereis succinea			4 ± 4
Parandalia americana			109 ± 6
Mediomastus californiensis		11 ± 11	76 ± 23
Streblospio sp.		18 ± 10	15 ± 4
Capitella capitata			33 ± 11
Polydora sp.			4 ± 4
OLIGOCHAETES		22 ± 17	138 ± 28
TURBELLARIANS			44 ± 33
NEMERTEANS		11 ± 11	243 ± 60
CRUSTACEANS			
Edotea montosa		4 ± 4	4 ± 4
Cyathura polita			
Monoculodes edwardsi			
Corophium lacustre			
Grandidierella bonnieroides			
Melita nitida			
Cerapus sp.			
Mysidopsis almyra		33 ± 0	11 ± 7
Copepods			22 ± 13
Ostracods			
Rhithropanopeus harrisii			
Callinectes sapidus			
CHIRONOMIDS		106 ± 22	131 ± 0
TOTAL		19941 ± 651	31418 ± 5425

APPENDIX C

BENTHIC COMMUNITY STRUCTURAL AND METABOLIC CHANGES ALONG A TRANSECT THROUGH DREDGED AND UNDREDGED AREAS

by David W. Roberts and Leonard M. Bahr, Jr.

The following information is derived from an M.S. thesis resulting from a project performed during the Lake Pontchartrain benthic characterization and shell dredging effects studies. This thesis was written by David W. Roberts under the direction of Leonard M. Bahr, Jr.

The overall objectives of this project were to measure total benthic community metabolism along a transect from the edge of Lake Pontchartrain to the center of the lake and to relate changes in this functional parameter to corresponding changes in community structure. Because the transect passed through a variety of lake environments, including areas never dredged and areas dredged heavily, the resulting information is complementary to the data compiled during the dredging effects study. Therefore, the information particularly pertinent to that study is summarized below.

Additional details on the techniques and analyses can be found in Roberts (1981).

Data Collection

Bottom core samples were taken along a transect from Goose Point on the north shore of Lake Pontchartrain to the Causeway near the center of the lake (Figure C1). Ten stations were occupied once during late June and early July of 1979, close to the peak annual water temperature. Stations 1-6 were evenly spaced at 0.25 mi. (0.4 km) intervals from the lake edge across the nearshore region. Stations 7-10 were spaced at larger intervals across the open lake. Stations 1-5 are within the nearshore sanctuary zone protected from shell dredging. Stations 9 and 10 are within the protected area of the one-mile-wide strip on either side of the Causeway, although station 10 has been dredged experimentally.

Three 0.03 m^2 replicate sediment cores were taken at each station with a spade corer designed to produce a sealed, intact sample of surface sediment and overlying water. The same samples were used for both functional and structural measurements.

Sediment Oxygen Uptake

Total community metabolism was estimated by measuring the rate of oxygen depletion in water overlying each core during incubation in a constant temperature bath. Measurements of dissolved oxygen (DO) were made initially and at 15-20 minute intervals for 3-5 hours with a YSI Model 57 dissolved oxygen meter equipped with a Model 5720 self-stirring polarographic electrode, calibrated by Winkler titration. The water

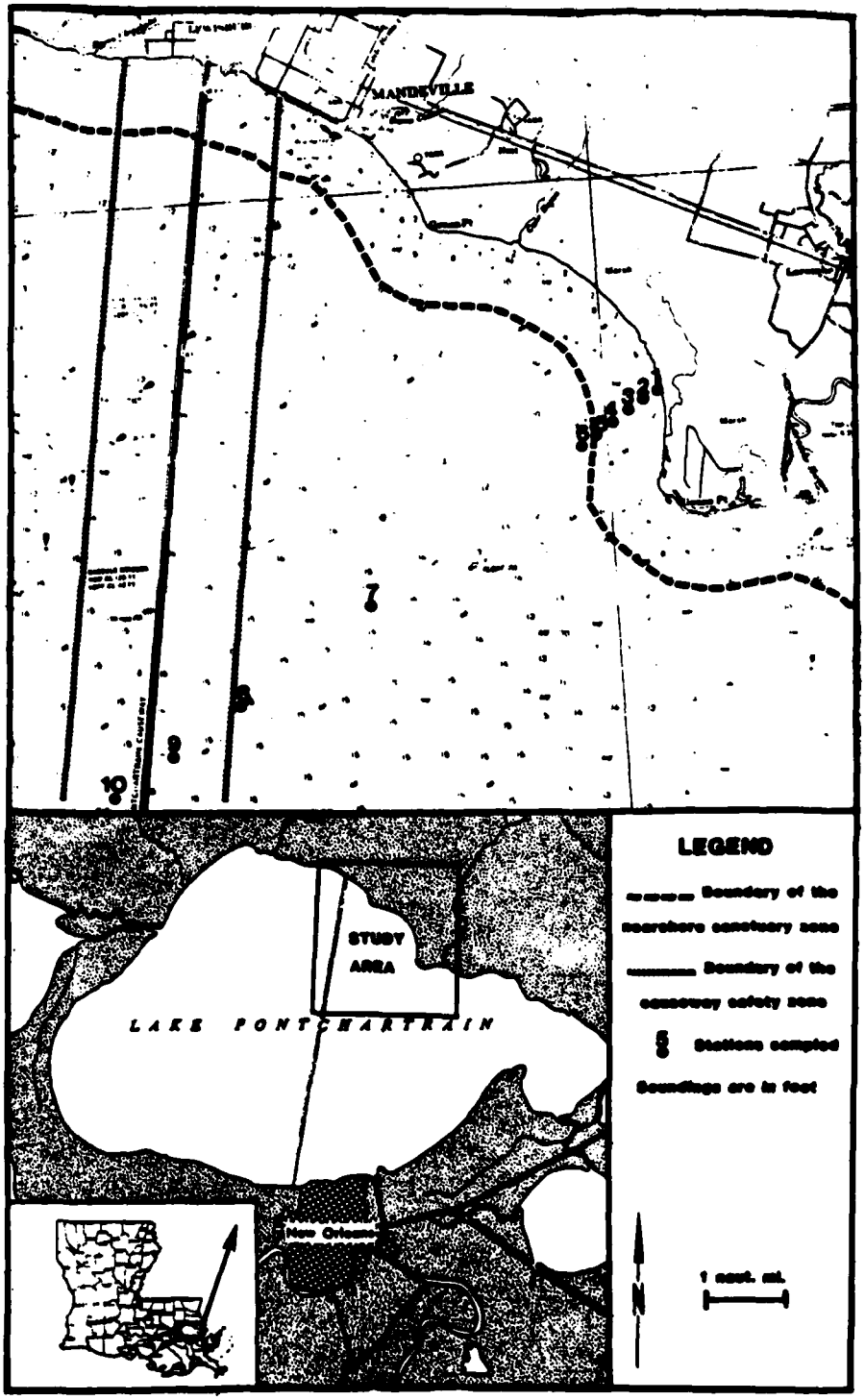

Figure C1. Location of the study area and sampling stations.

column respiration rate was calculated from change in DO of the replicate water samples, held in BOD bottles at in situ temperatures.

Measurements of DO were multiplied by the volume of overlying water to obtain total oxygen content. These values and corresponding times were subjected to least squares linear regression, which gave an integrated rate of O_2 uptake for each sample. Values for water column respiration were calculated from initial and final DO readings and total incubation time, and subtracted from total O_2 uptake to give sediment O_2 uptake.

This final value was correlated for core area to oxygen used per m^2 per hr.

Infaunal Abundance and Biomass

After measuring the community metabolism, four meiofauna subsamples were collected from the set of cores and preserved and analyzed according to the procedure described elsewhere in this report.

All remaining sediment in each core was sieved (0.5 mm mesh) for macrofauna, and sorting was performed as described elsewhere in this report. Measurements of biomass (afdw) were performed on macrofauna and meiofauna from all samples.

In addition, grain size analysis, total organic carbon content, and measurements of relative shell content were performed on samples from each station, with the exception of stations 8-10, for which carbon measurements were not obtained.

Results

Physicochemical Factors

Water depth along the transect ranged from 0.5 m at the lake edge to 5.6 m near the Causeway. The data for sediment grain-size distribution shows no distinctive pattern from the lake edge to the open lake. In general, coarse particles (sand and silt) predominate in all samples analyzed.

Organic content increased steadily from a low of 10 mg afdw·g dry wt^{-1} at station 1 to a high of 119 mg afdw·g dry wt^{-1} at one mile, before decreasing again at station 6 (Figure C3). A second peak of organic carbon occurred at station 7 in the open lake, but it cannot be fully evaluated because of the missing values for stations 8, 9, and 10.

The shell ratio (large shell to small shell) displayed an interesting pattern. Mean values decreased rapidly from station 1 to 5. The ratio remained above 1.0 at station 5, indicating that the weight of large shell material was greater than that of small shell. At station 6, the ratio dropped abruptly to 0.02, and remained below 1.0 at stations 7 and 8 in the open lake (Figure C2). At stations 9 and 10, the ratio rose again above 1.0, indicating the predominance of large shell.

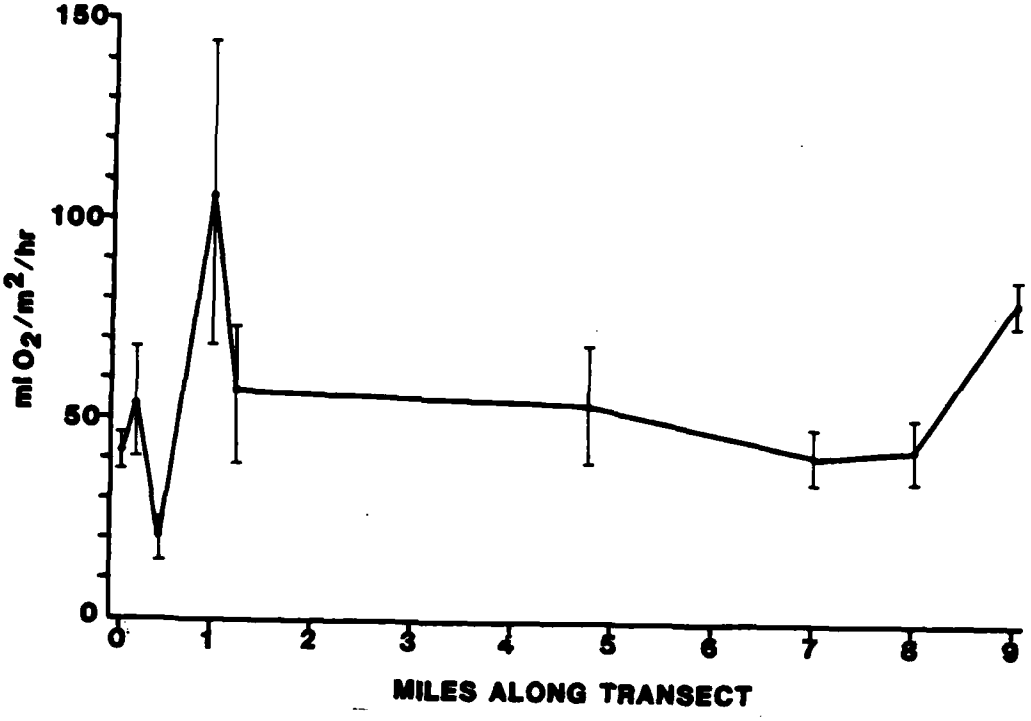

Figure C3. Total community respiration as a function of distance from shore along the transect. Vertical lines represent standard error.

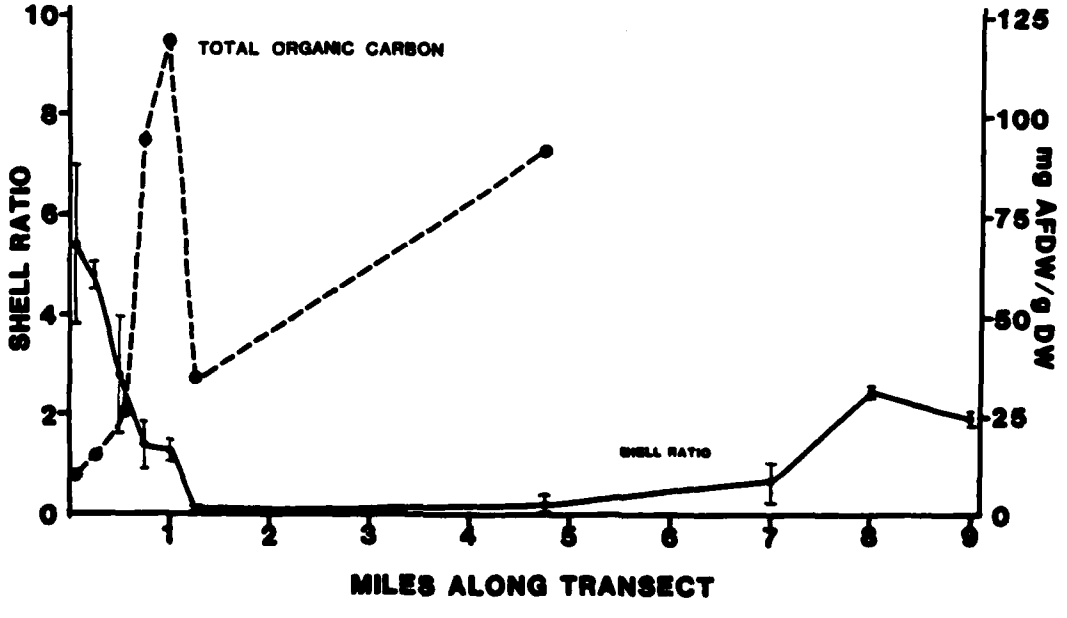

Figure C2. Graph of organic carbon content and shell ratio of the sediments as a function of distance from shore along the transect. Vertical lines represent standard error.

Benthic Community Metabolism

The rate of oxygen uptake (QO_2) from undisturbed sediment cores represents a very good estimate of total respiration of the benthic communities sampled. The QO_2 ranged from 20.0 to 98.6 ml $O_2 \cdot m^{-2} hr^{-1}$, with highest rates at station 5, one mile from shore (see Figure C3). The next highest QO_2 occurred at station 10 (80.6 ml $O_2 \cdot m^{-2} hr^{-1}$). These two values were significantly higher than at all other stations ($P > 0.05$). For comparison a QO_2 of 101 ml $O_2 \cdot m^{-2} hr^{-1}$ was reported for the marine nearshore environment off the Georgia coast at similar temperatures (Smith 1973). Respiration rates at stations 6-9 were not significantly different from each other ($P < 0.05$). Ranging from 45-60 ml $O_2 \cdot {}^{-2} hr^{-1}$. The conspicuously low value recorded at station 3 was the result of very low initial oxygen concentrations and is probably spurious. A very intense bloom of Anabena sp. was noticed during the sampling at station 3, and shading of the algae near the bottom was the most likely explanation of the oxygen depletion.

Infaunal Abundance and Biomass

A total of 23 macrofaunal taxa were distinguished from the 10 stations sampled. The largest collection of taxa (19) occurred at station 5, which was also the site of peak macrofaunal abundance and biomass (Figure C4). The lowest abundance and biomass of macrofauna was found at the lake edge (station 1) and at 1.25 miles (2.0 km) from shore (station 6). Data collected in the vicinity of station 1 indicate that macrofaunal frequency declined steadily from 4717 ind$\cdot m^{-2}$ in May to 2148 ind$\cdot m^{-2}$ in August, and then rose sharply to 5307 ind$\cdot m^{-2}$ in September (M. Crandall, pers. comm.). The August value above is comparable to the 2479 ind$\cdot m^{-2}$ measured in this study, suggesting that the macrofaunal community was at a low point at station 1 during the study.

The only macrofaunal specimens weighing over 1 g afdw were very large Rangia cuneata. The largest clam measured 46 mm and weighed 3.89 g afdw (station 10). No Rangia in the size class between 10-42 mm were collected along the transect. Individuals greater than 42 mm were only found at nearshore stations 1, 2, and 5, and at station 10 near the Causeway, and their infrequent occurrence and large biomass posed problems in statistical analysis.

Biomass of Rangia and Probythinella showed a similar pattern along the transect (Figures C5 and C6): a gradual increase at stations 2, 3, and 4, a peak at 1 mile (1.6 km), and a sharp drop at station 6 (1.25 mile, 2.0 km). Mulinia and Texadina biomass also exhibited a quite similar pattern. Biomass was lowest for both species in the nearshore area and gradually increased across the open lake to a peak at station 10 near the causeway (Figures C5 and C6).

Fourteen taxonomic groups of meiofauna were collected along the transect. Total meiofaunal abundance ranges from 2.36×10^6 ind$\cdot m^{-2}$ at station 1 to 0.197×10^6 ind$\cdot m^2$ at station 3. The highest biomass of meiofauna, which was highly significant ($P > 0.001$), was found at station 1 (6.47 g afdw$\cdot m^{-2}$). No significant differences in meiofauna

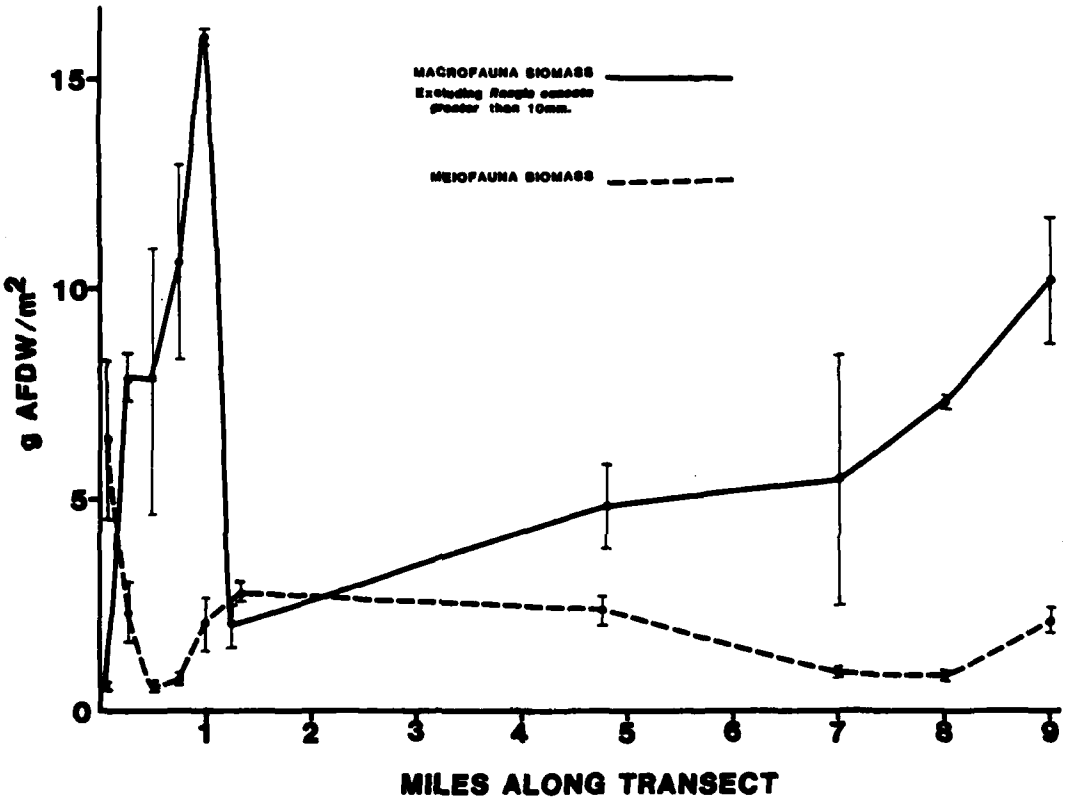

Figure C4. Total macrofauna and meiofauna biomass as a function of distance from shore along the transect. Vertical lines represent standard error.

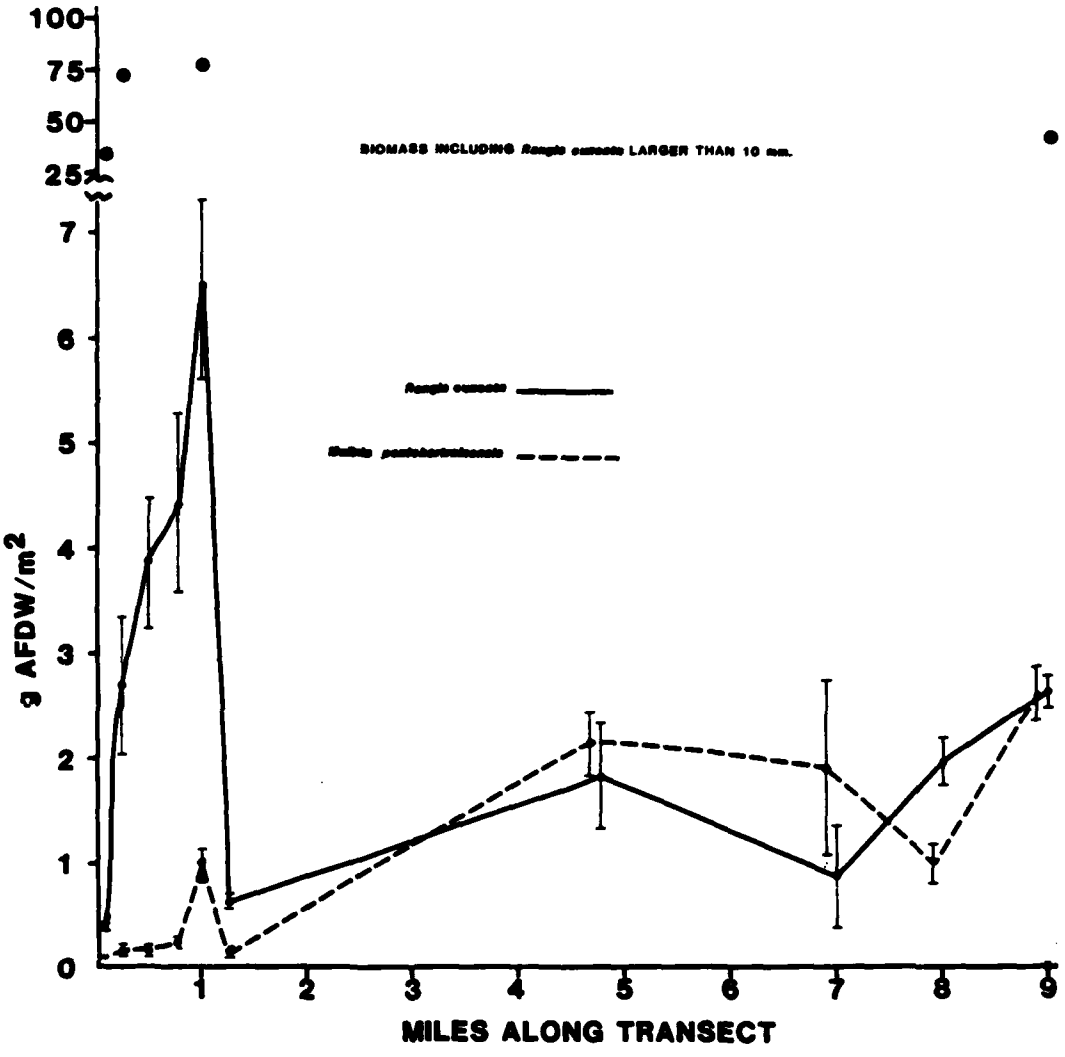

Figure C5. Biomass of the clams R. cuneata compared to M. pontchartrainensis as a function of distance from shore along the transect. Darkened circles show mean biomass of R. cuneata if large clams are included. Vertical lines represent standard error.

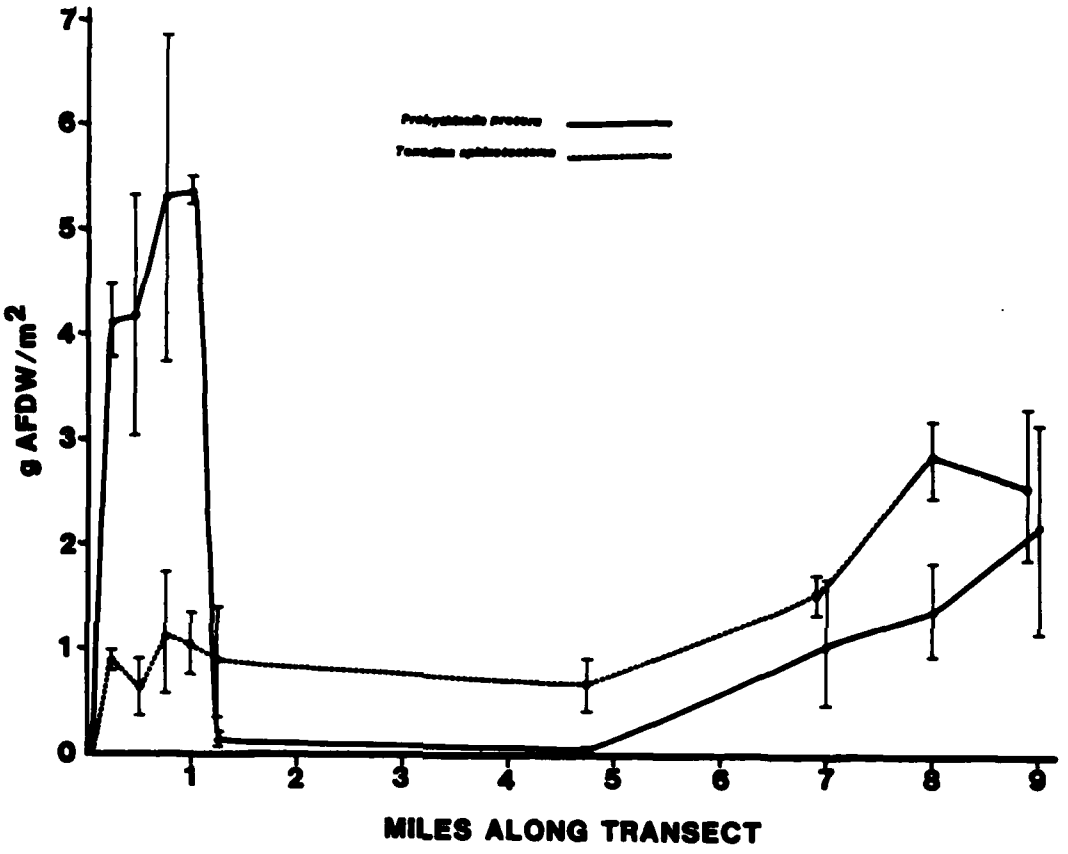

Figure C6. Biomass of the hydrobiid snails P. protera compared to T. sphinctostoma as a function of distance from shore along the transect. Vertical lines represent standard error.

biomass were noted among the other stations (Figure C4). It is not surprising that total biomass of meiofauna was significantly correlated with percent sand and shell ratio ($P > 0.001$), which were both highest at station 1.

Nematode species comrpised the dominant taxa at nearshore stations 1-5 (Figure C7). Nematodes accounted for 82% of the meiofaunal biomass at station 1, where the individuals were much larger than nematodes collected at other stations (6.0 µg vs. 2.5 µg afdw·ind^{-1}).

At stations 6, 7, and 8, nematode biomass declined and harpacticoid copepods became the dominant meiobenthic form (Figure C7). Copepod biomass was found to be significantly correlated with percent clay of the sediments ($P > 0.05$). Nematodes became dominant again at stations 9 and 10 near the Causeway.

Summary and Implications

The following conclusions about shell dredging effects can be drawn from this transect study:

(1) Peaks in total community metabolism, macrofauna biomass, and organic carbon content were found one mile from shore, at the edge of the shoreline protected zone.

(2) Macrofauna biomass, organic carbon content, and shell content decreased rapidly between one mile and 1.25 miles from shore.

(3) Structure of the benthic assemblages changed markedly from the shore to the middle of the lake. At the lake edge, a variety of relatively rare macrofaunal species occurred at low densities, along with a meiofaunal community of high biomass dominated by nematodes. From 0.25 to 1.00 miles from shore, the macrofauna were dominated by a clam, Rangia cuneata and a hydrobiid snail, Probythinella protera. Nematodes continued to predominate in the meiofauna. From 1.25 miles to 7.00 miles (corresponding to the active dredging zone), another clam and another snail species became dominant (Mulinia pontchartrainensis and Texadina sphinctostoma), and copepods became the major meiofaunal taxa. Between 7.00 miles and the end of the transect (corresponding to the Causeway protected zone) nematodes again replaced copepods as the dominant meiofaunal taxon.

(4) Total community metabolism, macrofauna biomass, and shell content increased to approximately the nearshore levels between 7.00 miles and the end of the transect (9.00 miles), corresponding to the Causeway protected zone.

From these structural and functional data, it appears that the 10 stations on the transect can be grouped into four distinct benthic communities: a lake edge community, a nearshore community, an open lake community, and a "Causeway zone" community. Estimates of maximum daily metabolism of these communities are: 1.17 ± 0.12 g C per m^2 per day

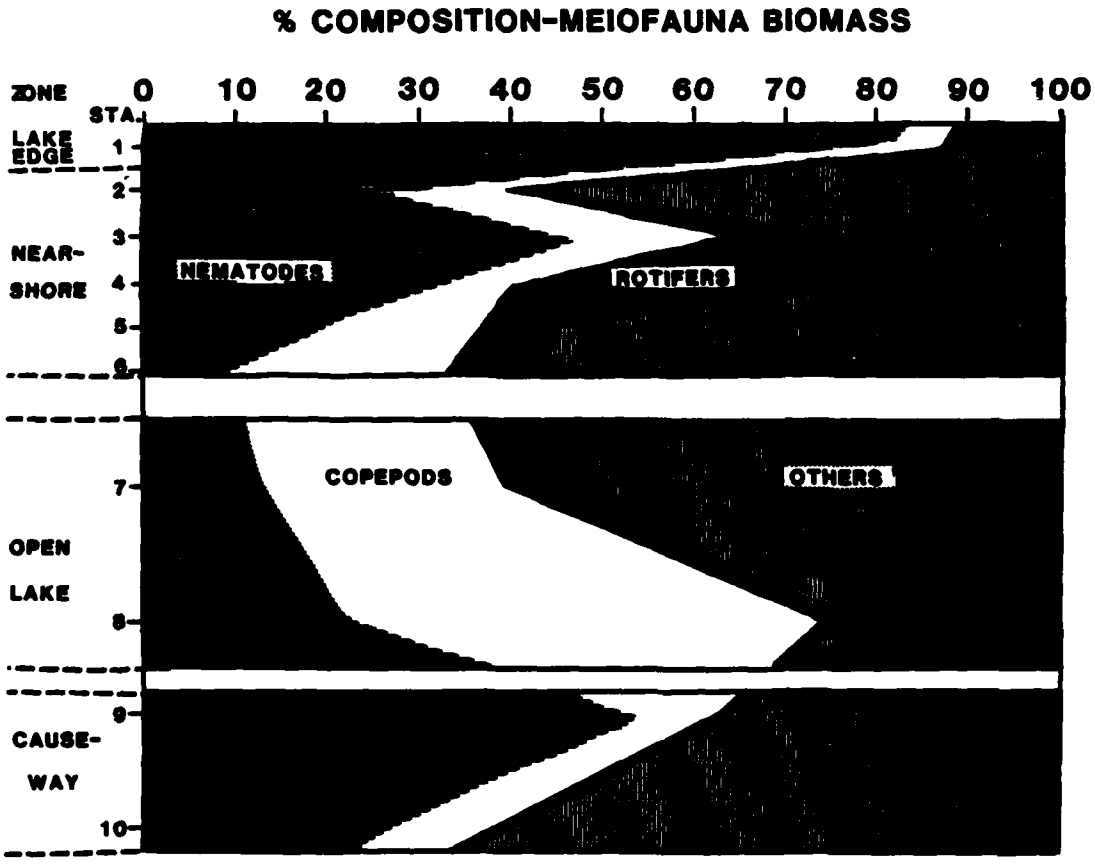

Figure C7. Percent composition of meiofaunal biomass in terms of dominant taxa and region in the lake.

(lake edge), 2.36 ± 0.65 g C per m^2 per day (nearshore), 1.47 ± 0.22 g C per m^2 per day (open lake), and 1.81 ± 0.38 g C per m^2 per day (Causeway zone).

Several explanations could account for these different zones: (1) Edge effects, including different regimes of hydrologic energy and variations in sediment grain size distribution; (2) differential predation rates, i.e., enhanced predation at the lake edge and in the open lake; (3) differences in the magnitude of cultural perturbations in the four subsystems.

The most probable explanation is a combination of 1 and 3; that is, shell dredging in the open lake, coupled with the natural enhancement of benthic diversity and productivity near the shore of the lake.

It should be pointed out that although station 10 is within the Causeway protected zone, it was experimentally dredged during the course of the dredging effects study. That no significant differences between stations 9 and 10 showed up in this study is, therefore, somewhat anomalous, but might be explained by the fact that the three subsamples collected at station 10 were taken exactly at the marker buoy in a spot that remained intact following the experimental dredging. The results of this study would not be changed if all data from station 10 were omitted.

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APPENDIX D

EFFECT OF SHELL DREDGING ON LAKE PONTCHARTRAIN MEIOBENTHIC COPEPODS

by J. W. Fleeger

The effect of dredging did not act equally on all meiobenthic copepod species, and thus analysis of total copepod abundance reveals little about dredging effects. For total copepod numbers, analysis of variance (ANOVA) shows significant temporal, i.e., seasonal, changes in density; however, no differences in abundance could be discerned between the control and dredged sites. Analysis of individual species reveals a differential response to dredging. Both short-term (within 50 days following dredging) and long-term (as long as 425 days following dredging) effects are discussed below for the four most abundant meiobenthic copepods; together these four comprise more than 98% of all copepods collected. These copepods are of the epibenthic type (see Coull 1977), living at the sediment-water interface, and all four are common from brackish water, soft sediment locations in Louisiana (see Wilson 1958, Hiegel 1969, Fleeger and Clark 1979).

ANOVA reveals that both Scottalana canadensis, a harpacticoid, and Acartia tonsa, a calanoid, respond in a very similar fashion to dredging and to seasonality. These species are well known from the plankton as well as from the benthos. Both are present in collections from each season, although S. canadensis does have a slight density peak in late fall, wjereas A. tonsa shows a similar peak in early fall. Short term effects on density are shown for S. canadensis and A. tonsa by significant density declines within 50 days following dredging. Both species, however, recovered by the spring of 1979, and no long-term impact on density could be detected.

ANOVA statistics show that Halicyclops fosteri, a cyclopoid, and Pseudobradya sp., a harpacticoid, have distinct seasonal trends in population density. Both species are also found in the plankton, but with much less regularity compared to Scottolana canadensis and Acartia tonsa. H. fosteri displayed a pronounced density peak in early August of 1978 and 1979, with a marked population decline in late August and early September. Psuedobradya sp. was prominent only in spring collections and was the dominant copepod from February-April 1979.

Long-term dredging effects are shown by ANOVA for H. fosteri and Pseudobradya sp. A dramatic seasonal (August) decline in H. fosteri abundance took place concurrently with the initial dredging, making identification of short-term dredging effects on H. fosteri impossible. H. fosteri did recover at the dredged site by the spring of 1979, and, in fact, long-term trends reveal a significantly greater abundance at the dredged site compared to the control during August 1979. During the following year, the dredged and control sites were similar in abundance for H. fosteri. Short-term dredging effects on Pseudobradya sp. are impossible to detect because of its low seasonal abundance when dredging took place in the fall of 1978. A long-term effect occurred as densities of Pseudobradya sp. were increased at the dredged site relative to the

control during the spring of 1979. During 1980, Pseudobradya sp. abundances were similar at each site.

To summarize, Scottolana canadensis and Acartia tonsa showed short-term impacts attributable to shell dredging. Both species recovered within six months, and no long-term effect was observed. For Halicyclops fosteri and Pseudobradya sp., no short-term effects of dredging were detected because of the presence of seasonal effects. Long-term effects on both species were evident. H. fosteri showed increased abundance at the dredged site one year after dredging; however, no differences were seen after two years. In Pseudobradya sp., increased abundance was observed after 6-8 months at the dredged site. Following this increase, dredged and control sites showed equal densities.

Table D1. Summary of ANOVA statistics calculated on copepod abundances.

	Short-Term		Long-Term	
Species	Seasonal Trends	Dredging vs. Control	Seasonal Trends	Dredging vs. Control
Halicyclops fosteri	+	-	+	+
Pseudobradya sp.	+	-	+	+
Scottolana canadensis	+	+	+	-
Acartia tonsa	+	+	+	-

Short-term refers to the 50 day period following dredging, long-term to the entire 425 days currently under investigation. A + indicates significance at the 0.05 level of probability; a - indicates a probability value greater than 0.05.

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